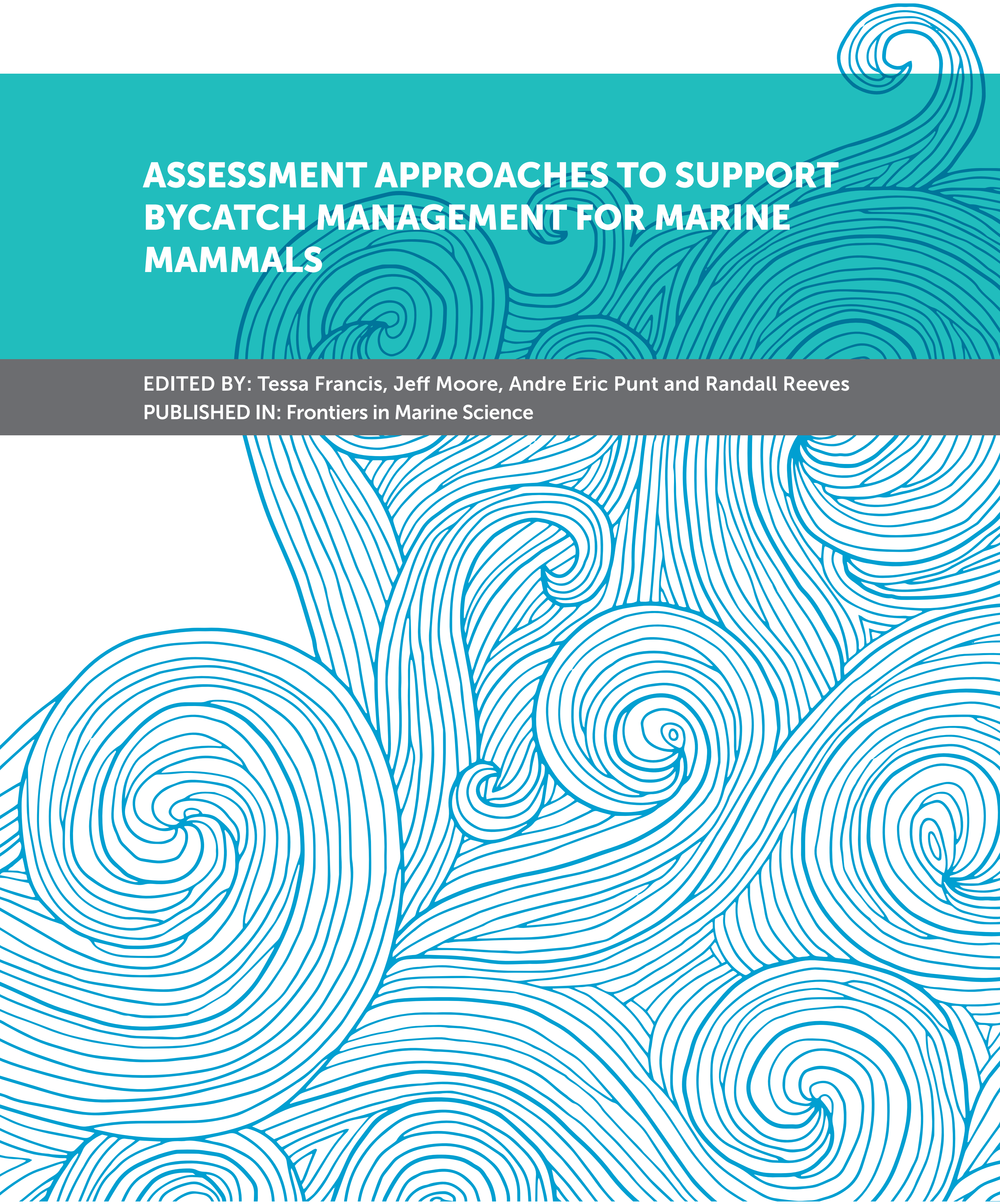
ASSESSMENT APPROACHES TO SUPPORT BYCATCH MANAGEMENT FOR MARINE MAMMALS

EDITED BY: Tessa Francis, Jeff Moore, Andre Eric Punt and Randall Reeves
PUBLISHED IN: Frontiers in Marine Science



frontiers Research Topics

ISSN 1664-8714
ISBN 978-2-88976-394-8
DOI 10.3389/978-2-88976-394-8

About Frontiers

Frontiers is more than just an open-access publisher of scholarly articles: it is a pioneering approach to the world of academia, radically improving the way scholarly research is managed. The grand vision of Frontiers is a world where all people have an equal opportunity to seek, share and generate knowledge. Frontiers provides immediate and permanent online open access to all its publications, but this alone is not enough to realize our grand goals.

Frontiers Journal Series

The Frontiers Journal Series is a multi-tier and interdisciplinary set of open-access, online journals, promising a paradigm shift from the current review, selection and dissemination processes in academic publishing. All Frontiers journals are driven by researchers for researchers; therefore, they constitute a service to the scholarly community. At the same time, the Frontiers Journal Series operates on a revolutionary invention, the tiered publishing system, initially addressing specific communities of scholars, and gradually climbing up to broader public understanding, thus serving the interests of the lay society, too.

Dedication to Quality

Each Frontiers article is a landmark of the highest quality, thanks to genuinely collaborative interactions between authors and review editors, who include some of the world's best academicians. Research must be certified by peers before entering a stream of knowledge that may eventually reach the public - and shape society; therefore, Frontiers only applies the most rigorous and unbiased reviews.
Frontiers revolutionizes research publishing by freely delivering the most outstanding research, evaluated with no bias from both the academic and social point of view. By applying the most advanced information technologies, Frontiers is catapulting scholarly publishing into a new generation.

What are Frontiers Research Topics?

Frontiers Research Topics are very popular trademarks of the Frontiers Journals Series: they are collections of at least ten articles, all centered on a particular subject. With their unique mix of varied contributions from Original Research to Review Articles, Frontiers Research Topics unify the most influential researchers, the latest key findings and historical advances in a hot research area! Find out more on how to host your own Frontiers Research Topic or contribute to one as an author by contacting the Frontiers Editorial Office: frontiersin.org/about/contact

ASSESSMENT APPROACHES TO SUPPORT BYCATCH MANAGEMENT FOR MARINE MAMMALS

Topic Editors:
Tessa Francis, University of Washington Tacoma, United States
Jeff Moore, Southwest Fisheries Science Center (NOAA), United States
Andre Eric Punt, University of Washington, United States
Randall Reeves, Okapi Wildlife Associates, Canada

Citation: Francis, T., Moore, J., Punt, A. E., Reeves, R., eds. (2022). Assessment Approaches to Support Bycatch Management for Marine Mammals. Lausanne: Frontiers Media SA. doi: 10.3389/978-2-88976-394-8

Table of Contents

05 ***Editorial: Assessment Approaches to Support Bycatch Management for Marine Mammals***
André E. Punt, Tessa B. Francis, Jeffrey E. Moore and Randall R. Reeves

08 ***Estimating the Abundance of Marine Mammal Populations***
Philip S. Hammond, Tessa B. Francis, Dennis Heinemann, Kristy J. Long, Jeffrey E. Moore, André E. Punt, Randall R. Reeves, Maritza Sepúlveda, Guðjón Már Sigurðsson, Margaret C. Siple, Gísli Víkingsson, Paul R. Wade, Rob Williams and Alexandre N. Zerbini

35 ***Estimating Cetacean Bycatch From Non-representative Samples (I): A Simulation Study With Regularized Multilevel Regression and Post-stratification***
Matthieu Authier, Etienne Rouby and Kelly Macleod

54 ***Using Bayesian Models to Estimate Humpback Whale Entanglements in the United States West Coast Sablefish Pot Fishery***
Jason E. Jannot, Eric J. Ward, Kayleigh A. Somers, Blake E. Feist, Thomas P. Good, Dan Lawson and James V. Carretta

68 ***Best Practices for Assessing and Managing Bycatch of Marine Mammals***
Paul R. Wade, Kristy J. Long, Tessa B. Francis, André E. Punt, Philip S. Hammond, Dennis Heinemann, Jeffrey E. Moore, Randall R. Reeves, Maritza Sepúlveda, Genoa Sullaway, Guðjón Már Sigurðsson, Margaret C. Siple, Gísli A. Víkingsson, Rob Williams and Alexandre N. Zerbini

87 ***Using Case Studies to Investigate Cetacean Bycatch/Interaction Under-Reporting in Countries With Reporting Legislation***
Charla Jean Basran and Guðjón Már Sigurðsson

102 ***Estimating Bycatch Mortality for Marine Mammals: Concepts and Best Practices***
Jeffrey E. Moore, Dennis Heinemann, Tessa B. Francis, Philip S. Hammond, Kristy J. Long, André E. Punt, Randall R. Reeves, Maritza Sepúlveda, Guðjón Már Sigurðsson, Margaret C. Siple, Gísli A. Víkingsson, Paul R. Wade, Rob Williams and Alexandre N. Zerbini

117 ***Participatory Risk Assessment of Humpback Whale (*****Megaptera novaeangliae*****) and Leatherback Turtle (*****Dermochelys coriacea*****) Bycatch in Northern Peru***
Anna B. Costanza, Chiara Guidino, Jeffrey C. Mangel, Joanna Alfaro-Shigueto, Gregory Verutes, Marjolaine Caillat, Aritree Samanta and Ellen Hines

140 ***Evaluating Strategies for Managing Anthropogenic Mortality on Marine Mammals: An R Implementation With the Package RLA***
Mathieu Genu, Anita Gilles, Philip S. Hammond, Kelly Macleod, Jade Paillé, Iosu Paradinas, Sophie Smout, Arliss J. Winship and Matthieu Authier

158 ***Precautionary Principle or Evidence-Based Conservation? Assessing the Information Content of Threat Data for the Yangtze Finless Porpoise***
Lisa M. W. Mogensen, Zhigang Mei, Yujiang Hao, Xavier A. Harrison, Ding Wang and Samuel T. Turvey

169 ***Estimating Bycatch From Non-representative Samples (II): A Case Study of Pair Trawlers and Common Dolphins in the Bay of Biscay***
Etienne Rouby, Laurent Dubroca, Thomas Cloâtre, Sebastien Demanèche, Mathieu Genu, Kelly Macleod, Hélène Peltier, Vincent Ridoux and Matthieu Authier

184 ***Low Resighting Rate of Entangled Humpback Whales Within the California, Oregon, and Washington Region Based on Photo-Identification and Long-Term Life History Data***
Jennifer Tackaberry, Elana Dobson, Kiirsten Flynn, Ted Cheeseman and John Calambokidis

197 ***Assessment of Australian Sea Lion Bycatch Mortality in a Gillnet Fishery, and Implementation and Evaluation of an Effective Mitigation Strategy***
Simon D. Goldsworthy, Brad Page, Derek J. Hamer, Andrew D. Lowther, Peter D. Shaughnessy, Mark A. Hindell, Paul Burch, Daniel P. Costa, Shannon L. Fowler, Kristian Peters, Rebecca R. McIntosh, Frederic Bailleul, Alice I. Mackay, Roger Kirkwood, Dirk Holman and Simon Bryars

frontiers | Frontiers in Marine Science

EDITORIAL
published: 27 May 2022
doi: 10.3389/fmars.2022.928127

Editorial: Assessment Approaches to Support Bycatch Management for Marine Mammals

André E. Punt[1*], *Tessa B. Francis*[1,2], *Jeffrey E. Moore*[3] *and Randall R. Reeves*[4]

[1] School of Aquatic and Fishery Sciences, University of Washington, Seattle, WA, United States, [2] Puget Sound Institute, University of Washington Tacoma, Tacoma, WA, United States, [3] Marine Mammal and Turtle Division, NOAA, Southwest Fisheries Science Center (SWFSC), La Jolla, CA, United States, [4] Okapi Wildlife Associates, Hudson, QC, Canada

Keywords: bycatch, abundance estimation, marine mammals, conservation, mitigation

Editorial on the Research Topic

Assessment Approaches to Support Bycatch Management for Marine Mammals

Bycatch in marine fisheries is, and long has been, one of the leading sources of human-caused mortality of marine mammals (Lewison et al., 2004; Read et al., 2005). It has contributed to declines of many populations and species, and at least one species extinction. A well-established approach for managing the impacts of fishing on marine mammal populations involves identifying the fishery or fisheries of concern, identifying the affected marine mammal populations, collecting data on marine mammal abundance and bycatch, estimating levels of mortality that populations can likely sustain, and implementing regulations or taking other mitigation approaches to reduce marine mammal bycatch as needed to achieve management goals. The Potential Biological Removal (PBR) framework established under the USA's Marine Mammal Protection Act (MMPA) identifies a level of human-caused mortality that, with high probability, will allow a given marine mammal population to remain at, or recover to, its level of maximal production (Wade, 1998), and prescribes a process for achieving this.

Interest in developing similarly robust assessment frameworks for use across a broad range of contexts is expected to increase worldwide owing to the recently-implemented seafood import provisions of the MMPA[12], which require exporting countries to implement reliable, standardized methods for collecting and analyzing data to estimate marine mammal abundance and bycatch rates. The absence of either the data to evaluate bycatch impacts or a plan to collect such data for fisheries that export to the USA can lead to the products from those fisheries being prohibited from entering the USA, with severe implications for associated fishing communities, some of which are in poor and developing countries. At the same time, the import provisions provide incentives and opportunities to reduce marine mammal bycatch.

This Research Topic was inspired by a need to increase the number of marine mammal populations around the world for which abundance and bycatch can be estimated and to help identify which fisheries are most urgently in need of mitigation. The 12 resulting contributions consist of nine original articles, one policy and practice article, and two review articles that collectively provide guidance and novel methods or examples for addressing these issues.

Wade et al. provide a comprehensive overview of the steps involved with assessment and management of marine mammal bycatch. These include collecting data on the abundance and bycatch of marine mammals and on fisheries that are known or suspected to cause bycatch, assessing the impact of bycatch in relation to reference points, and using the results of the assessments to guide

OPEN ACCESS

Edited and reviewed by:
Peter H. Dutton,
Southwest Fisheries Science Center (NOAA), United States

****Correspondence:***
André E. Punt
aepunt@uw.edu

Specialty section:
This article was submitted to Marine Megafauna, a section of the journal Frontiers in Marine Science

Received: *25 April 2022*
Accepted: *09 May 2022*
Published: *27 May 2022*

Citation:
Punt AE, Francis TB, Moore JE and Reeves RR (2022) Editorial: Assessment Approaches to Support Bycatch Management for Marine Mammals. Front. Mar. Sci. 9:928127. doi: 10.3389/fmars.2022.928127

---

[1] U.S. Federal Code: 16 U.S.C. 1371(a)(2).

bycatch mitigation and reduction. Most of the other papers in the Research Topic expand on the steps outlined by Wade et al.

Two publications provide best-practice guidance for data collection and estimation of marine mammal abundance and bycatch, applicable across species and in a variety of programmatic and data contexts. Hammond et al. review the steps for estimating abundance of marine mammal populations. Guidance is provided for the use of transect sampling from ships and aircraft, land- or ice-based counts, and mark-recapture methods. The paper summarizes data collection and practical considerations related to estimating abundance for pinnipeds, cetaceans, and sirenians. Moore et al. review methods for estimating bycatch mortality of marine mammals using data on bycatch-per-unit-effort, total fishing effort, and rate of mortality of bycaught animals. They summarize sources of error when estimating bycatch, including non-representative sampling, observer effects, and cryptic mortality.

The Research Topic also features articles exploring bycatch estimation using sub-optimal data. Basran and Már Sigurðsson quantify the under-reporting bias in estimates of bycatch when estimation methods rely on data from logbooks rather than observer programs, using data from New Zealand, the United States, and Iceland. They conclude that if reliance on logbook data is to continue, clearer regulations and simplified reporting using modern technology, in combination with electronic monitoring cameras to verify compliance, would improve accuracy. Authier et al. outline a model-based approach for estimating bycatch when the data are non-representative. This can arise when levels of observer coverage are low or certain fishing sectors or vessel types are not monitored due to logistics, costs, or security issues. Rouby et al. address the difficulties that arise from non-representative data on bycatch of common dolphins (*Delphinus delphis*) in the Bay of Biscay by proposing a regularized multilevel regression method with post-stratification implemented using Bayesian methods.

Several papers explore the implications of unobserved mortality or injury. Jannot et al. describe a model-based approach for estimating bycatch of humpback whales (*Megaptera novaeangliae*) in the fishery for sablefish (*Anoplopoma fimbria*) off the US west coast. The method is based on the Bayesian paradigm to estimate probability distributions for unobserved bycatch. Tackaberry et al. use photo-identification data to explore post-release survival of humpback whales entangled in fisheries off the US west coast, and find that entangled whales are seen (and photo-identified) less frequently than control animals, and that this is particularly true of younger individuals. Constanza et al. use a low-cost, public-participation mapping approach called Bycatch Risk Assessment (Hines et al., 2020) to generate a spatial and temporal assessment of entanglement (bycatch) risk to humpback whales off northern Peru.

Genu et al. describe an R software package to carry out management strategy evaluation of control rules for setting marine mammal removal limits, and provide an example application for harbor porpoises (*Phocoena phocoena*) in the North Sea.

Mogensen et al. use multiple analytical approaches to investigate spatial relationships between live and dead Yangtze finless porpoises (*Neophocaena asiaeorientalis asiaeorientalis*) and different threats, trends in reproduction over time, and sustainable offtake levels, finding that mortality is spatially associated with cargo traffic, that observed mortality levels are unsustainable, and that population recruitment is declining. Goldsworthy et al. describe an assessment of bycatch of Australian sea lions (*Neophoca cinerea*) off South Australia and evaluate the effectiveness of mitigation methods including closures, bycatch mortality limits, and incentives to switch to gears with lower bycatch rates.

The contributions to this Research Topic provide guidance for managers and researchers who are expected to understand, quantify, and mitigate the impacts of marine mammal–fishery interactions, including situations where few or no data are available and no system is in place to collect needed data.

AUTHOR'S NOTE

The views expressed are those of the authors and do not necessarily reflect the viewsof the Lenfest Ocean Program or the Pew Charitable Trust.

AUTHOR CONTRIBUTIONS

All authors have made a substantial, direct and intellectual contribution to the work, and approved it for publication.

FUNDING

Support for this project was provided by the Lenfest Ocean Program

REFERENCES

Hines, E., Ponnampalam, L. S., Junchompoo, C., and Peter, C. (2020). Getting to the Bottom of Bycatch: A GIS-Based Toolbox to Assess the Risk of Marine Mammal Bycatch. Endanger. *Species Res.* 42, 37—57. doi: 10.3354/esr0103

Lewison, R. L., Crowder, L. B., Read, A. J., and Freeman, S. A. (2004). Understanding Impacts of Fisheries Bycatch on Marine Megafauna. *Trends Ecol. Evol.*19 (11), 598—604.

Read, A.J., Reynolds, J. E., Perrin, W. F., Reeves, R. R., Montgomery, S., and Ragen, T. J. (2005). Bycatch and Depredation," in: Marine Mammal Research: *Conservation Beyond Crisis*. p. 5—17.

Wade, P. R. (1998). Calculating Limits to the Allowable Human-Caused Mortality of Cetaceans and Pinnipeds. *Mar. Mammal Sci.* 14, 1—37.

Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

frontiers
in Marine Science

REVIEW
published: 27 September 2021
doi: 10.3389/fmars.2021.735770

Estimating the Abundance of Marine Mammal Populations

Philip S. Hammond[1]*, Tessa B. Francis[2,3], Dennis Heinemann[4], Kristy J. Long[5], Jeffrey E. Moore[6], André E. Punt[2], Randall R. Reeves[7], Maritza Sepúlveda[8], Guðjón Már Sigurðsson[9], Margaret C. Siple[2], Gísli Víkingsson[9], Paul R. Wade[10], Rob Williams[11] and Alexandre N. Zerbini[10,12,13]

[1] Sea Mammal Research Unit, Scottish Oceans Institute, University of St Andrews, St Andrews, United Kingdom, [2] School of Aquatic and Fishery Sciences, University of Washington, Seattle, WA, United States, [3] Puget Sound Institute, University of Washington Tacoma, Tacoma, WA, United States, [4] U.S. Marine Mammal Commission, Bethesda, MD, United States, [5] Office of Protected Resources, NOAA's National Marine Fisheries Service, Silver Spring, MD, United States, [6] Protected Resources Division, NOAA SWFSC, La Jolla, CA, United States, [7] Okapi Wildlife Associates, Hudson, QC, Canada, [8] Facultad de Ciencias, Universidad de Valparaíso, Valparaíso, Chile, [9] Pelagic Division, Marine and Freshwater Research Institute, Hafnarfjörður, Iceland, [10] Marine Mammal Laboratory, NOAA AFSC, Seattle, WA, United States, [11] Oceans Initiative, Seattle, WA, United States, [12] Cascadia Research Collective, Olympia, WA, United States, [13] Marine Ecology and Telemetry Research, Seabeck, WA, United States

Edited by:
Jeremy Kiszka,
Florida International University,
United States

Reviewed by:
Luis Cardona,
University of Barcelona, Spain
Chandra Paulina Salgado Kent,
Edith Cowan University, Australia

***Correspondence:**
Philip S. Hammond
psh2@st-andrews.ac.uk

Specialty section:
This article was submitted to
Marine Megafauna,
a section of the journal
Frontiers in Marine Science

Received: *03 July 2021*
Accepted: *23 August 2021*
Published: *27 September 2021*

Citation:
Hammond PS, Francis TB, Heinemann D, Long KJ, Moore JE, Punt AE, Reeves RR, Sepúlveda M, Sigurðsson GM, Siple MC, Víkingsson G, Wade PR, Williams R and Zerbini AN (2021) Estimating the Abundance of Marine Mammal Populations. Front. Mar. Sci. 8:735770. doi: 10.3389/fmars.2021.735770

Motivated by the need to estimate the abundance of marine mammal populations to inform conservation assessments, especially relating to fishery bycatch, this paper provides background on abundance estimation and reviews the various methods available for pinnipeds, cetaceans and sirenians. We first give an "entry-level" introduction to abundance estimation, including fundamental concepts and the importance of recognizing sources of bias and obtaining a measure of precision. Each of the primary methods available to estimate abundance of marine mammals is then described, including data collection and analysis, common challenges in implementation, and the assumptions made, violation of which can lead to bias. The main method for estimating pinniped abundance is extrapolation of counts of animals (pups or all-ages) on land or ice to the whole population. Cetacean and sirenian abundance is primarily estimated from transect surveys conducted from ships, small boats or aircraft. If individuals of a species can be recognized from natural markings, mark-recapture analysis of photo-identification data can be used to estimate the number of animals using the study area. Throughout, we cite example studies that illustrate the methods described. To estimate the abundance of a marine mammal population, key issues include: defining the population to be estimated, considering candidate methods based on strengths and weaknesses in relation to a range of logistical and practical issues, being aware of the resources required to collect and analyze the data, and understanding the assumptions made. We conclude with a discussion of some practical issues, given the various challenges that arise during implementation.

Keywords: abundance, cetaceans, pinnipeds, sirenians, population size

INTRODUCTION

Ecologists use information on abundance to improve understanding of what determines how species are distributed (Begon et al., 1996). In conservation biology, practitioners use estimates of abundance to assess the conservation status of species or populations (Cardinale et al., 2019). Assessments for the IUCN Red List of Threatened Species[1] depend on abundance information in three of the five criteria used to rank risk levels (A: reduction in population; C: small and declining population size; and D: very small or range-restricted population). Population size and trend estimation are also used to inform options and mechanisms for managing human activities to reduce impacts.

One motivation for this paper is the widespread threat to marine mammal populations due to incidental catch (bycatch) in fishing gear (Read et al., 2006; Avila et al., 2018), and the need for information on abundance to put bycatch mortality into a population context. In Europe, a legal driver for this is the EU Marine Strategy Framework Directive, under which biodiversity indicator D1C1 requires that, to ensure long-term viability, incidental bycatch mortality be kept below levels that threaten a species (European Union, 2017). The implementing regulations[2] of the United States Marine Mammal Protection Act (MMPA) issued in 2016, require that imported fish and fish products be evaluated with respect to United States standards for managing marine mammal bycatch (as discussed in Williams et al., 2016a). These implementing regulations ("Seafood Import Provisions") require countries with fisheries identified as "Export Fisheries" to be comparable in effectiveness to the regulatory program for United States commercial fisheries; this typically requires conducting bycatch assessments of marine mammal populations (Wade et al., In review, Frontiers in Marine Science[3]).

Two fundamental pieces of information needed to conduct a bycatch assessment are an estimate of bycatch mortality and an estimate of the abundance of the impacted population. Best-practice methods for estimating bycatch mortality are described in Moore et al. (In review, Frontiers in Marine Science[4]). This paper describes methods for estimating the abundance of populations of marine mammals.

An additional, more general, motivation for the present work is that expertise in estimating marine mammal abundance is patchy around the world. Although methods to estimate abundance have been developed and applied to many marine mammal species, there are still numerous populations globally where necessary data and estimates of abundance are lacking. We hope that the information in this paper will be of use to managers and practitioners in places around the world where knowledge of the methods available to estimate marine mammal abundance may be limited.

Efforts to estimate the abundance of marine mammal populations began in earnest in the 1960s. One of the first such efforts for cetaceans involved systematic counts from coastal CA, United States, of migrating gray whales (*Eschrichtius robustus*) (see Laake et al., 2012). For pinnipeds, McLaren (1961) used data from at-sea surveys to estimate the abundance of ringed seals (*Pusa hispida*) and counts of gray seal (*Halichoerus grypus)* pups were initiated in the early 1960s to estimate the population around Britain (Russell et al., 2019). The use of data from seals that were physically captured, tagged, and recaptured to estimate abundance using mark-recapture methods began in the 1970s (e.g., Siniff et al., 1977).

Methods for estimating abundance from whale sightings were first considered by Doi (1974). In the 1970s, two notable cetacean survey programs to estimate abundance began. First, the United States government initiated surveys to estimate the abundance of pantropical spotted (*Stenella attenuata*) and spinner dolphins (*S. longirostris*) in the eastern tropical Pacific (Smith, 1981). Second, surveys under the International Whaling Commission's International Decade of Cetacean Research were initiated to estimate the abundance of Antarctic minke whales (*Balaenoptera bonaerensis*) (see Branch and Butterworth, 2001).

The development of photo-identification of individual whales and dolphins in the 1970s enabled the use of mark-recapture methods to estimate the abundance of cetaceans from their natural markings, including humpback (*Megaptera novaeangliae*), southern right (*Eubalaena australis*), killer (*Orcinus orca*), and sperm (*Physeter macrocephalus*) whales, and bottlenose dolphins (Bigg, 1982; Whitehead, 1982; Whitehead and Gordon, 1986; Whitehead et al., 1986; Scott et al., 1990). Whales had previously been marked using Discovery tags (Brown, 1978), but these data were neither intended nor used for estimating abundance.

The 1980s saw the publication of a seminal textbook on the estimation of animal abundance by Seber (1982), followed by the continued development of mark-recapture and transect methods tailored for estimation of marine mammal abundance (see reviews in Hammond, 1986; Hiby and Hammond, 1989; Hammond et al., 1990, and aerial survey methods for dugongs [*Dugong dugon*] in Marsh and Sinclair, 1989).

Methodological development has expanded rapidly over the last three decades and there is now an extensive literature describing studies to estimate abundance for many marine mammal species. There are also general texts that describe methods that are commonly used, or that provide practical guidance on implementation (e.g., Buckland et al., 2001, 2015; Amstrup et al., 2005; Dawson et al., 2008; Hammond, 2010; Williams et al., 2017; Buckland and York, 2018).

The aim of this paper is to provide easy access to the way in which marine mammal abundance can be estimated, highlighting the relevant literature (established and recent) so that readers can pursue further details about the various methods available for pinnipeds, cetaceans and sirenians. We start with a general introduction to abundance estimation, including an "entry-level" description of the fundamental concepts and the

---

[1] https://www.iucnredlist.org/

[2] https://www.federalregister.gov/documents/2016/08/15/2016-19158/fish-andfish-product-import-provisions-of-the-marine-mammal-protection-act

[3] Wade, P. R., Long, K. J., Francis, T. B., Punt, A. E.,Hammond, P. S., Heinemann, D., et al. (in review). Best practices for assessing and managing bycatch of marine mammals. *Front. Mar. Sci.*

[4] Moore, J. E., Heinemann, D., Francis, T. B., Hammond, P. S., Long, K. J., Punt, A. E., et al. (in review). Estimating bycatch mortality for marine mammal stock assessment: concepts and best practices. *Front. Mar. Sci.*

importance of recognizing sources of bias (systematic inaccuracy in estimates) and obtaining a measure of precision (uncertainty in estimates). Each of the primary methods available to estimate abundance of marine mammals is then described. For each method, information is summarized on data collection and analysis, including common challenges in implementation. The assumptions of the method (violation of which can lead to bias) are outlined and references to example studies that illustrate the method are given. Each method has strengths and weaknesses in relation to a range of logistical and practical issues, including relative cost. The final section discusses which methods are likely to be appropriate for pinnipeds, cetaceans and sirenians, given the various challenges that arise during implementation.

This paper does not address the estimation of trends in abundance. The challenges of detecting trends, driven by the limited power of typical abundance data, especially for cetaceans, are well-documented (e.g., Taylor et al., 2007). Some studies describing how to provide the best information on abundance trends from the available data include Jewell et al. (2012); Moore and Barlow (2014), Campbell et al. (2015); Williams et al. (2016b), Forney et al. (2021); Nachtsheim et al. (2021), and Boyd and Punt (2021).

ESTIMATING ABUNDANCE – GENERAL PRINCIPLES

Abundance and Population Size

The terms abundance and population size are sometimes used interchangeably. The common usage of the English word "abundance" indicates a large quantity of something but, in the context of estimating species abundance, it simply means the number of animals, whether large or small. Abundance may refer to the number of animals in any defined area or it may refer to a "population," which begs the question – what is a population?

As a precursor to his discussion of comparative demography, Cole (1957) defined a population as: "*a biological unit at the level of ecological integration where it is meaningful to speak of a birth rate, a death rate, a sex ratio, and an age structure in describing the properties of the unit.*" This resonates with the commonly considered definition that a population comprises individuals of a species that live and interbreed in the same space.

However, such a definition is challenging to implement for many species, perhaps particularly so for marine mammals. Krebs (1972, p139) proposed a more pragmatic definition: *"a group of organisms of the same species occupying a particular space at a particular time . . . [its] boundaries . . . both in space and time are vague and in practice are usually fixed by the investigator arbitrarily."* Indeed, studies of abundance using at-sea transect surveys (see below) need to be defined in space and time. Estimates of abundance from such surveys therefore relate to the "population" in an area, not to a wider biological population. In studies to estimate abundance using mark-recapture methods (see below), the population estimated comprises those individuals that use the study area, which may be different from the number of animals present in the area during a transect survey. Calambokidis and Barlow (2004) discuss this difference with respect to humpback and blue (*Balaenoptera musculus*) whales.

A population may also be defined for the purposes of assessment and/or management; such definitions may be general or specific. For example, the IUCN defines a population as the "*set of individuals from the same wild species that share the same habitat*" (IUCN, 2021). The United States Marine Mammal Protection Act defines a "population stock" as a group of animals "*of the same species or smaller [taxon] in a common spatial arrangement that interbreed when mature*" (MMPA Section 3, 16 United States Code 1362). The International Whaling Commission (IWC) manages whaling on the basis of defined management stocks, for which abundance estimates are required to be incorporated within a management procedure [e.g., bowhead whales (*Balaena mysticetus*) in the Bering-Chukchi-Beaufort Seas[5]]. The International Council for the Exploration of the Sea has defined "*Assessment Units*" for some species of small cetacean in the European Atlantic [e.g., harbor porpoise (*Phocoena phocoena)*; ICES, 2014].

Whatever the context, for any study reporting abundance, it is critical to define clearly the "population" to which abundance refers.

Counting and Estimation of Abundance From Samples

The simplest measure of the number of animals in a population is a *count*. On rare occasions (e.g., "southern resident" killer whales, which are all uniquely identifiable and do not disperse from their natal units), all individuals in the population can be counted (Olesiuk et al., 1990). However, a count is highly unlikely to represent true population size for most marine mammals, especially cetaceans, because of problems of access (individuals spend all or most of their time at sea and mostly underwater), behavior (individuals are highly mobile), and scale (most species occur over large areas). Nevertheless, a count is a *minimum* measure of abundance and this may be of value in the absence of an estimate of abundance. However, use of a count, in the large majority of cases, would substantially underestimate the capacity of the population to sustain human impacts, such as bycatch mortality, and would therefore lead to much lower bycatch limits than are necessary to achieve management objectives.

To obtain a realistic measure of the number of animals in a marine mammal population, abundance must typically be estimated. Estimation involves collecting a sample of data that can be extrapolated to the entire population using one or more statistical models. The principles of extrapolating sample data to estimate abundance using models can be illustrated through a simple example.

Consider the need to obtain a measure of the number of animals in a defined study area of size A = 450 km^2. It is not logistically feasible to count all the animals in the area. Instead, a smaller subarea of size a = 50 km^2 is defined, within which it is believed possible to count all the animals, and in which the density of animals is assumed to be representative of the

---

[5] https://archive.iwc.int/pages/view.php?ref=3606&k=

larger area A. The count in the subarea ($n = 35$ animals) forms the sample. Extrapolating to obtain an estimate of abundance is achieved by (i) dividing the count n by the size of the subarea a to provide an estimate of animal density, $n/a = 35/50 = 0.7$ animals per km^2, and (ii) multiplying estimated animal density by the size of the whole study area, $A = 450$ km^2 to provide the estimate of abundance, 315 animals.

The statistical model in this case is simply: $\hat{N} = \frac{n}{a} \times A$. The "hat" over the N indicates that it is an estimated quantity.

The sizes of the areas, A and a, are known quantities because they can be measured exactly, but the number of animals in the subarea is unknown and the count, n, is thus a random variable with associated statistical error. Estimating this error allows us to provide a measure of *precision* of the estimate of abundance (see below).

Bias and Precision in Estimates of Abundance

All models, whether simple or complex, make assumptions about the *accuracy* and *representativeness* of the sample data. The resulting estimates of abundance can be *biased* if these assumptions are not met. A biased estimate is not an accurate measure of true abundance.

In the example in the previous section, the fundamental assumption made is that it is valid to extrapolate the estimate of animal density in the subarea, n/a, to the whole study area. This assumption has two elements: first, that the sample count, n, accurately represents the number of animals in the subarea a; and second, that animal density in the subarea is representative of density in the study area as a whole.

The first element, accuracy, can be thought of as the capacity of the data collection protocol to provide accurate data. If it over- or under-counts the animals, animal density will similarly be over- or underestimated and extrapolation to the whole study area will result in an over- or underestimate of abundance. For example, the count will be incomplete and abundance will be underestimated if some animals are unavailable for detection at the time of the count because they are hidden from view, or if they react to the presence of the observer and move away before they are counted. These two situations can occur in surveys for marine mammals and the solution is to collect additional data so that a correction factor can be estimated to avoid under- or over-estimation of abundance.

The second element, representativeness, is determined by the sampling design. In this example, a single subarea has been selected for estimating animal density. This could be appropriate if animals were distributed randomly in space, but this is rarely the case and cannot be assumed. Counting within the subarea would lead to overestimating animal density and, therefore, abundance, if, e.g., there were a gradient in density across the study area and the small area fell within an area of high density. The solution is to modify the sampling design, for example by splitting the single subarea into several smaller subareas and distributing them randomly or systematically over the entire study area. This should ensure that estimated animal density is representative of overall density.

If assumptions about the data and methodology are met, the resulting estimate of abundance should be an *unbiased* measure of the true abundance. The concept here is that if multiple sets of sample data are collected, the resulting repeat estimates of abundance should be centered about the true abundance. In practice, because true abundance is unknown, the only way to determine that an estimate is unbiased is to ensure that the assumptions about the data and the estimation method are met.

Obtaining an unbiased estimate is ideally the aim of all studies of abundance. It is therefore important (a) to use a method of data collection that maximizes the *accuracy* of the data, and (b) to implement a sampling design that ensures the sample data are *representative*.

The point was made in the previous section that estimating statistical error enables an estimate of abundance or population size to be presented with a measure of precision. The measure of precision expresses the level of uncertainty we have in the estimate. Statistically, it indicates how much the estimate would vary if it were calculated from repeated sets of sample data. Measures of precision are primarily driven by sample size; they include the Standard Error (SE), Coefficient of Variation (CV, the SE divided by the estimate, expressed as a proportion or a percentage), and a 95% Confidence Interval (CI). An estimate with poor precision indicates that its value was driven substantially by chance, whereas an estimate with higher precision indicates that its value would be similar regardless of the sample of data collected. We should thus be less certain about an imprecise estimate, and be more cautious about its use, than a precise estimate. As a general rule, an estimate of abundance should always be accompanied by a measure of precision.

It is important to be clear about the difference between bias and precision in this context. An estimate may be based on few data and thus be imprecise but nevertheless be unbiased if assumptions about the data and estimation methods are met. In contrast, a precise estimate based on a large sample may be biased because of violation of sampling or analytical method assumptions. For informing conservation and management, the worst-case scenario is arguably a precise but biased estimate because it conveys a false sense of confidence in an inaccurate measure of abundance. For example, consider a population of 10,000 animals, for which there is an estimate of abundance of 15,000 with a CV of 0.1. The estimate is inflated because the sampled subarea contained a higher density of animals than the entire study area. The CV is small because a lot of data were collected in the sampled subarea. In this case, the 95% confidence interval for the estimate (12,000–18,000) would not include the true value. A mortality limit reference point (see Moore et al., In review, Frontiers in Marine Science) (see text footnote 4) calculated on the basis of this estimate would be considerably inflated, potentially leading to bycatch levels that threaten the population. This reinforces the importance of understanding and meeting assumptions about the data and the estimation methods.

Accompanying Information to Support an Abundance Estimate

Abundance estimates provided to management authorities should be accompanied by enough information to allow assessment of their validity. As well as the estimate (for a particular location and time) and measure(s) of its precision, it is also good practice to provide:

- A description of the survey/sampling design;
- A description of data collection methods, including survey platform, observer training and experience;
- Summary data, including spatial maps, searching effort, animals counted and/or individuals identified (depending on the method); and
- A description of analytical methods and additional information used to generate the estimate and the measure(s) of its precision.

An example of this is the specification of information that should accompany any abundance estimate intended for use in IWC management applications (e.g., the Revised Management Procedure or an Aboriginal Whaling Management Procedure; IWC, 2012, 2020).

METHODS FOR ESTIMATING ABUNDANCE

Several methods are available to estimate the abundance of marine mammal populations; each can be considered within the conceptual framework of extrapolating from counts, corrected as appropriate, to an estimate of total abundance or population size using statistical models. Which method is most appropriate and how it is implemented depends on the species and other factors such as logistics, resources and, in some cases, the purpose of estimating abundance. This section introduces the various methods available and identifies their data requirements, key features and assumptions.

Correcting and Extrapolating Counts of a Defined Portion of a Population

The primary basis for estimating the abundance of pinniped populations (seals, sea lions, fur seals, walruses) is information on counts of animals on land or ice. Cetaceans (whales, dolphins and porpoises) and sirenians (dugongs and manatees) are never similarly accessible for such counts. Counts may be of pups (or walrus calves) born during a pupping season or of non-pups hauled out at that time or at other times of the year.

Pinnipeds that haul out on land typically occur in breeding colonies (where pups can be counted) or in other types of aggregation outside the pupping season (e.g., for molting), thus facilitating the counting of animals. These counts can be corrected, if necessary, and then extrapolated to provide an estimate of population size. Ice-associated pinnipeds are typically distributed over wide areas of ice requiring information on counts to be obtained from aerial surveys (see "Transect sampling from ship or aerial surveys").

More generally, estimates using count data may be combined with those obtained using other methods such as surveys (see "Transect sampling from ship or aerial surveys" below) and mark-recapture (see "Mark-recapture" below), for example for Hawaiian monk seals (*Neomonachus schauinslandi*) (Baker et al., 2016).

Pup Counts

In species of pinniped that form breeding colonies on land, pups may be counted during the pupping season from a boat (e.g., Boyd, 1993), from land (e.g., Guinet et al., 1994; Wege et al., 2016), or often most efficiently via aerial surveys (e.g., Russell et al., 2019). Collating data from diverse counting methods may be required (e.g., Galatius et al., 2020). Pup counts or estimates of total pup production are frequently used as indices of relative abundance, especially when considering population trends over time (e.g., Guinet et al., 1994; Shaughnessy et al., 2000; Wege et al., 2016). Using pup counts to estimate population size requires extrapolation.

Depending on the method of counting, detecting all pups present may not be possible, e.g., because of obstructed views. The probability of detecting a pup can be estimated by focused comparisons of typical counts with known unbiased counts, or through a mark-recapture experiment using temporarily marked pups (e.g., Shaughnessy et al., 2000; Wege et al., 2016) to correct for negative bias in count data. Comparison of ground counts and aerial survey counts can also indicate bias in counts (e.g., Westlake et al., 1997; Lowry, 1999). The use of unmanned aerial vehicles (UAV) can markedly improve the probability of detecting a pup (e.g., Adame et al., 2017; McIntosh et al., 2018; see also "Aerial survey or ship survey?" below). An extreme situation is where ringed seal (*Pusa hispida*) pups are hidden in subnivean lairs and invisible and other methods may be more appropriate (e.g., photo-identification using camera traps; Koivuniemi et al., 2016).

Pupping seasons can extend over several weeks so conducting multiple counts over the entire pupping season may be necessary to obtain an unbiased estimate of total pup production (e.g., Russell et al., 2019). It may not be possible to conduct counts at all colonies in a population, in which case it is necessary to apply models to pup counts from a subset of colonies to obtain unbiased estimates of the total number of pups born in a season (e.g., Wege et al., 2016).

In species that give birth to pups over large areas of ice, aerial surveys are necessary to sample these areas. For example, pup production of harp seals (*Pagophilus groenlandicus*) in the northwest Atlantic is estimated using visual helicopter strip transect surveys of pupping areas previously identified using aerial reconnaissance, as well as photographic fixed-wing aerial surveys of these same areas (Stenson et al., 2020).

The simplest way to extrapolate from pups to total population is to construct a life table and compute the ratio of total population numbers to the number of pups, i.e., a "pup multiplier" (e.g., Pitcher et al., 2007; Hauksson, 2007; Russell et al., 2019). Values of demographic parameters (such as age-specific survival and fecundity rates) are required to construct the life table. To avoid bias, demographic parameters should be representative of the population to which the pup counts are being extrapolated and additional resources may be needed to estimate survival or fecundity. Alternatively, it may be necessary to use values from studies of the species, or a similar species, elsewhere (Trites and Larkin, 1996; Lowry et al., 2014; Punt et al., 2020). The Marine Mammal Bycatch Impacts Exploration tool[6] provides a range of calf/pup survival rates, adult survival rates and age at maturity for different groups of pinnipeds (Siple, 2021). Prudence is needed when using demographic parameters that are not population specific or if population size is changing and density-dependent effects have led to changes in demographic parameters over time (Frie et al., 2012). In addition, it is not easy to estimate the precision of estimates of total population size using this approach.

A more complex way to extrapolate from pups to total population size, that can overcome some of the problems associated with the simple life table approach, is to use a population model that is fitted to the count data (e.g., Thomas et al., 2019). Advantages of this approach include incorporating more detailed information on demographic parameters, modeling density-dependent effects, using datasets additional to pup counts (such as counts of adults and/or data on animals killed in hunts or fisheries) and incorporating uncertainty (e.g., Butterworth et al., 1995). The assumptions made using such population models to ensure unbiased estimation of population size depend on the models.

All-Age Haul-out Counts

Counts of animals older than pups made during the pupping season or at other times of the year, including the annual molt, can be used as a basis for estimating abundance. In species of pinniped that haul out in aggregations on land, the same counting methods can be used as described above for pups, and the same issues apply regarding failure to detect or count all animals at a colony or haul-out site, or not being able to make counts at all colonies/haul-out sites. Counts may thus need to be corrected as described above if assumed to be inaccurate or unrepresentative.

In ice-associated pinnipeds, aerial surveys are needed to obtain information on counts. For example, Bengtson et al. (2005) used fixed-wing aerial line transect surveys to estimate the abundance of ringed and bearded seal and Ver Hoef et al. (2014) used data from helicopter line transect surveys in the Bering Sea to estimate the abundance of ribbon (*Histriophoca fasciata*), bearded (*Erignathus barbatus*) and spotted (*Phoca largha*) seals. Speckman et al. (2011) used thermal images and digital photography data collected from fixed-wing aerial strip transect surveys to estimate the abundance of walrus (*Odobenus rosmarus*) in the Bering Sea. See also "Transect sampling from ships or aerial surveys" regarding strip and line transect surveys, and "Aerial survey or ship survey?" regarding the use of camera systems.

A novel method of obtaining counts that has potential for some species of pinniped is to use earth observation imagery (Moxley et al., 2017).

In some circumstances, it may be reasonable to assume that all animals are ashore when counts are made, for example mature animals on breeding colonies, and, in such cases, the count may represent this component of the population (e.g., for South American sea lions *Otaria flavescens*, Sepúlveda et al., 2011). However, typically, the number of animals hauled out varies substantially over time so, preferably, the mean of a series of counts conducted over an appropriate period should be used in place of a single count (e.g., Sepúlveda et al., 2012; Brasseur et al., 2018). This also allows variability in the count to be incorporated into the overall measure of precision of an extrapolated count (see below).

Because, typically, not all animals are ashore or on ice when counts are made, a mean count (or corrected count, if necessary) needs to be extrapolated to the entire population, and this requires estimating the proportion of the population that is hauled out during the period to which the mean count applies. This proportion needs to be estimated from data on haul-out behavior, obtained from observations of animals at haul-out sites or, preferably, from marked individuals. Animals may be marked using flipper tags or small implantable tags, by photo-identification, or, ideally, by deploying transmitters on the animals to obtain telemetry data (e.g., Thompson et al., 1997; Bengtson et al., 2005; Sharples et al., 2009; Udevitz et al., 2009; Lonergan et al., 2011; Ver Hoef et al., 2014). Either way, additional resources are required to generate data from which the proportion of the population hauled out can be estimated. To obtain an unbiased estimate of population size by simple extrapolation of a mean haul-out count using an estimate of the proportion of the population hauled out requires that the counts and the haul-out proportion data are, similarly, representative of all components of the population in terms of age, sex and life history. If this is not the case, analysis will need to account for the variation (e.g., Lonergan et al., 2011) or at least acknowledge potential bias in estimates of total population size.

Transect Sampling From Ship or Aerial Surveys

Most of the information on the abundance of cetaceans comes from surveys of defined areas conducted at sea on ships (or smaller boats), or from the air using sampling along define transect lines. Pinnipeds can also be detected during at-sea surveys, although it is unusual for these data to be used to estimate abundance (but see Brediñana-Romano et al., 2014; Williams et al., 2017). However, an aerial survey is the only option for obtaining information on abundance for some ice-associated pinnipeds. The abundance of sirenians is typically estimated from aerial surveys.

Unlike extrapolating counts or mark-recapture (see below), transect methods sample animal density within defined areas, not individual animals. Typically, a team of observers searches on either side of a set of transect lines placed across a survey area and counts the number of animals detected. Most cetaceans occur clustered in groups, so observers typically count groups of animals and determine (or estimate – see below) the size of each group. Use of camera systems instead of observers to capture data on aerial survey is discussed below.

Surveys may be conducted in so-called "passing mode" or "closing mode." In passing mode, the ship, small boat or aircraft does not divert from the transect line to investigate detected groups of animals. In closing mode, when a group of animals is detected, searching effort is suspended and the

---

[6] https://msiple.shinyapps.io/mammaltool/

group is approached by the ship or small boat or circled overhead by the aircraft. The additional time spent observing the animals, especially in close proximity from ships or small boats, may allow more accurate species identification and group size estimation, including taking photographs for this purpose. Surveys conducted in passing mode maximize searching time, and are necessary for double observer team surveys (see below), but they may be subject to greater error in species identification and group size estimation than surveys conducted in closing mode.

Abundance is estimated by first estimating the density of animals per unit area and then extrapolating density to the whole study area. Abundance is therefore defined as the estimated number of animals in a specified area during the period of time that the survey took place. It is thus conceptually similar to the example described in "Counting and estimation of abundance from samples," but the area sampled is the area around the set of transect lines placed across the survey area (see **Figure 1**), not a subarea.

Proper placement of the transect lines is critical for unbiased estimation of abundance. To guarantee that estimated animal density is representative of the entire study area, the set of transect lines must be placed according to a design that ensures that every point in the study area has the same probability of being sampled. This is known as an *equal coverage probability design*. Clearly any one set of transect lines can only sample a small proportion of the available area. However, incorporating a random starting point into an appropriate design ensures that many repeated, randomly chosen sets of lines would sample everywhere in the study area. Estimates of abundance from surveys that achieve equal coverage probability are known as *design-based estimates*.

Stratification of survey areas into a number of blocks is commonly done, often for logistical reasons. Survey design should then ensure equal coverage probability within each block. If density differs among blocks, for example because they cover different habitats, stratification may improve the precision of abundance estimates. Transect line placement can be challenging in coastal, inland, and riverine areas; Dawson et al. (2008) provide guidance for such areas. In some circumstances, data from surveys that do not achieve equal coverage probability can be used to estimate abundance in a model-based approach (see "Model-based estimation of abundance").

Transect surveys for cetaceans typically collect data visually. However, for deep-diving species, abundance estimated from visual data is likely to be negatively biased because of the long dive times of these species (see *availability bias* below). For these species, using passive acoustic data (i.e., recordings of sounds made regularly, such as for echolocation, by the animals) to estimate abundance may be possible. For example, sperm whale abundance has been estimated from a combined acoustic and visual survey (Barlow and Taylor, 2005) and solely from acoustic data collected from hydrophones towed behind a survey ship (Lewis et al., 2007). Taylor et al. (2017) combined a line transect survey with acoustic data from static recorders to estimate the abundance of vaquitas (*Phocoena sinus*). Data from static acoustic recorders have also been used to estimate the abundance of beaked whales (Marques et al., 2009) using point-transect distance sampling, a variant of line transect sampling in which the area around a series of points is searched, instead of the area on either side of a series of transect lines (Buckland et al., 2001). These methods are under development; recent work also includes the use of drifting passive acoustic recorders to estimate the density of deep-diving cetacean species (e.g., Keating et al., 2018; Barlow et al., 2021a,b,c).

Strip Transect Surveys

In strip transect sampling, it is assumed that all animals are detected within a strip of defined width on each side of the transect line (**Figure 1**). The sum of the lengths of all the transect lines multiplied by the width of the strip on both sides of the line defines the sample area searched. As noted above, this is equivalent to the subarea in the example in "Counting and estimation of abundance from samples." The number of animals detected (counted) divided by the size of the searched area gives an estimate of animal density, which is multiplied by the size of the survey area to estimate abundance. If not all animals are detected within the strip, abundance will be underestimated (see "Line transect sampling" below).

A form of strip transect is typically used to survey sirenians. In surveys of dugongs in Australia, the defined strip is narrow (200 m) and no clear evidence has been found of a decline in detection with perpendicular distance within the strip (Pollock et al., 2006). However, corrections to these counts are made for *availability bias* and *perception bias* (Marsh and Sinclair, 1989), which are described below. Other example studies of estimating sirenian abundance from such surveys include Preen (2004); Findlay et al. (2011), Martin et al. (2015); Hagihara et al. (2016) and Hostetler et al. (2018). Note also that abundance of manatees (*Trichechus manatus*) in Panama has been estimated from side-scan sonar deployed from a small boat in a river system (Guzman and Condit, 2017). Strip transect surveys have also been used to obtain data to estimate the abundance of ice-associated pinnipeds (e.g., Speckman et al., 2011; Stenson et al., 2020).

Line Transect Sampling

The probability of detecting an animal, or a group of animals, declines as the distance away from the transect line increases so, unless the strip is narrow, not all animals in the strip will be seen. Line transect sampling is a modification of strip transect sampling, in which the assumption that all animals are detected within a defined strip is relaxed; it is only required that all animals are seen on the transect line (see below for discussion of this). Instead, data are collected on the perpendicular distance from the transect line to each detected animal or group (**Figure 1**). Line transect sampling is thus often referred to as distance sampling (Buckland et al., 2001, 2015; Thomas et al., 2010).

A detection function is fitted to the perpendicular distance data (Thomas et al., 2010; **Figure 1**), which provides an estimate of the average probability of detection (of animals or groups) within the strip. Including covariates related to sighting conditions in the detection function may improve its fit. Dividing the number counted by the average probability of detection corrects the count for the animals or groups missed within the strip. In conventional analysis, however, the correction is actually made to the width of the strip, not to the number counted.

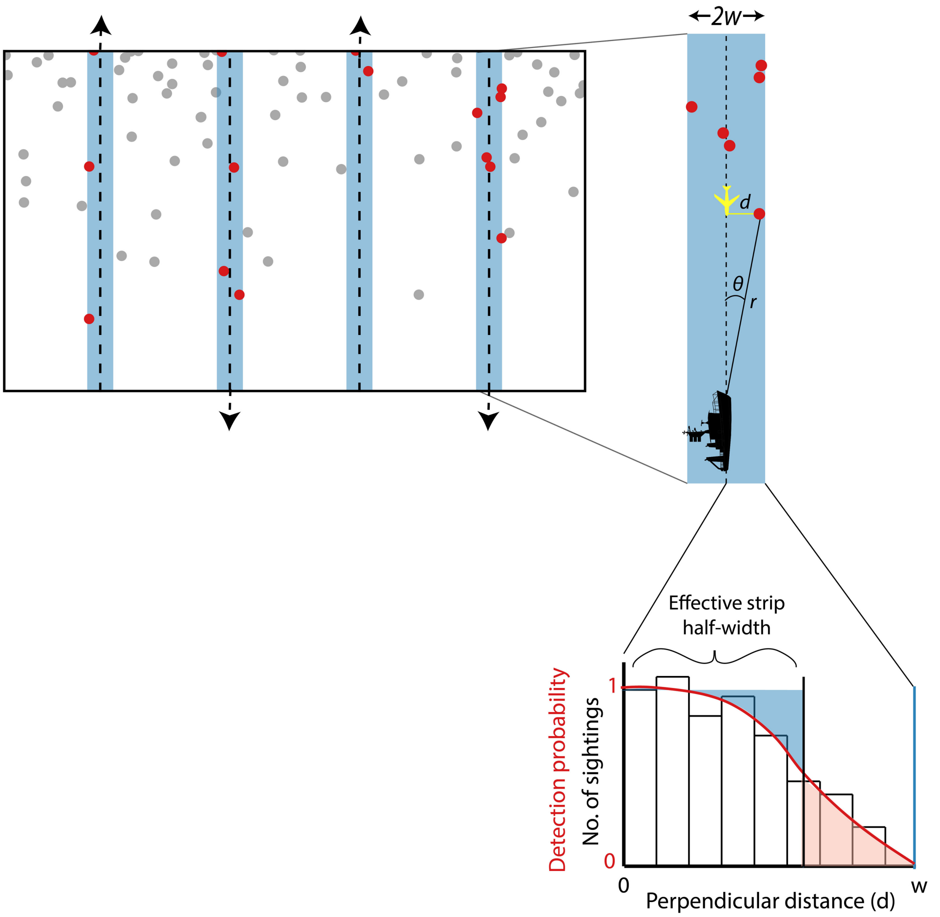


FIGURE 1 | Schematic representation of the elements of a line transect survey. Illustration of a sampling design (top left) in which parallel lines are placed equidistant from each other (other designs such as zig-zag lines could also be appropriate). The starting point for the first line is random so that the design satisfies the requirements for an equal coverage probability design (see text). The shaded areas around the lines represent the strip searched by observers as they progress in the direction of travel indicated by the arrows at the end of each line. The dots represent animals (or groups of animals); red dots are those within the searched strip. In a strip transect survey, all the animals represented by red dots are assumed to be detected. In a line transect survey, animals further from the transect line are less likely to be detected so not all of the animals represented by red dots may actually be sighted. Note that animals are not distributed randomly across the survey area – density is higher toward the top than the bottom. This design follows good practice in placing lines perpendicular to the density gradient (if known) to minimize inter-line variability in the number of animals detected and thus maximize precision in abundance estimation. To the right is an illustration of data collection on one of the transect lines. The width of the strip searched on each side of the line is defined as *w*. When an animal (or group of animals) is detected from the ship, observers measure the distance *r* and the angle θ, from which perpendicular distance from the transect line, *d*, can be calculated as *r* x sinθ. Perpendicular distance is measured directly from the aircraft when it comes abeam of the animal. On completion of a survey, the perpendicular distances measured to all the detected animals or groups can be plotted as a frequency histogram. Note that the frequency of animals detected declines as perpendicular distance increases away from the transect line (zero perpendicular distance). The red curved line represents the detection function fitted to the perpendicular distance data, with detection probability = 1 at perpendicular distance = 0, as assumed in line transect analysis (see text). The effective strip half-width, *esw* (average detection probability x *w*, see text), is the perpendicular distance at which the number of animals detected at greater distances (represented in pink) equals the number of animals present but not detected at lesser distances (represented in blue).

Multiplying the strip width by the average probability of detection gives a quantity known as the *effective strip width* (*esw*, **Figure 1**). The term "effective" signifies that it is the width of a strip within which all animals would have been counted. Note that *esw* conventionally refers to one side of the transect line only so is actually the *effective strip half-width*. The sum of the lengths of all the transect lines multiplied by the *esw* on both sides of the line defines the *effective search area*, within which all animals would have been counted.

Animal density is estimated by dividing the number of animals seen, or the number of groups seen multiplied by mean group size, by the *effective search area*. As for strip transect sampling, abundance is estimated by multiplying the estimate of animal density by the size of the survey area.

There is an extensive literature on using line transect sampling to estimate the abundance of many species of cetaceans in different habitats. **Table 1** gives references to some example studies that describe how the method can be implemented for a variety of species. In addition, line transect sampling is

TABLE 1 | Example studies using line transect (distance) sampling to estimate the abundance of cetacean species.

Survey platform	Species	Area	References
Small boat	Irrawaddy dolphin	Bangladesh, Sarawak, Malaysia	Smith et al., 2006; Minton et al., 2013; Kuit et al., 2021
Small boat	Indo-Pacific finless porpoise	Sarawak, Hong Kong, Malaysia	Minton et al., 2013; Jefferson and Moore, 2020; Kuit et al., 2021
Small boat	Indus river dolphin	Pakistan	Braulik et al., 2012
Small boat	Ganges river dolphin	Bangladesh	Smith et al., 2006
Small boat	Amazon river dolphin (boto), Tucuxi	Amazon and Orinoco Basins	Vidal et al., 1997; Gomez-Salazar et al., 2012; Williams et al., 2016b; Paschoalini et al., 2020
Small boat	Killer whale	British Columbia	Williams and Thomas, 2009
Ship	Multiple species	California Current	Barlow and Forney, 2007
Ship	Multiple species	Central North Atlantic	Pike et al., 2019
Ship	Multiple species	European Atlantic	Hammond et al., 2002, 2013
Ship	Multiple species	Alaska, British Columbia, Bering Sea	Zerbini et al., 2006; Williams and Thomas, 2007; Friday et al., 2013
Ship	Common minke whale	Northeast Atlantic	Skaug et al., 2004
Ship	Fin whale	Northeast and central Atlantic, California Current	Moore and Barlow, 2011; Víkingsson et al., 2013
Ship	Antarctic minke whale	Southern Ocean	Branch and Butterworth, 2001
Ship	Blue whale	Western United States	Calambokidis and Barlow, 2004
Ship	Blue whale	Southern Ocean	Branch, 2007
Ship	Killer whale	Alaska	Zerbini et al., 2007
Ship	Humpback whale	Brazil, Western United States, Southern Ocean	Calambokidis and Barlow, 2004; Branch, 2011; Bortolotto et al., 2016
Aerial	Multiple species	European Atlantic	Hammond et al., 2013
Aerial	Multiple species	Greenland	Hansen et al., 2018
Aerial	Common minke whale, humpback whale, white-beaked dolphin, harbor porpoise	Iceland	Borchers et al., 2009; Gilles et al., 2020; Pike et al., 2020
Aerial	Harbor porpoise	German/Dutch North Sea	Scheidat et al., 2012; Nachtsheim et al., 2021
Aerial	Harbor porpoise	California Current	Forney et al., 2021
Aerial	Franciscana dolphin	Argentina, Brazil	Crespo et al., 2010; Danilewicz et al., 2010; Sucunza et al., 2020
Aerial	Hector's dolphin	New Zealand	Slooten et al., 2004; MacKenzie and Clement, 2014

used to estimate the abundance of ice-associated seals (e.g., Bengtson et al., 2005; Ver Hoef et al., 2014).

Further Assumptions of Line Transect Sampling

An important assumption of line transect sampling is that all animals directly on the transect line are seen. Clearly, this is unlikely to be true generally for marine mammals, which spend most of their lives underwater and might thus be unavailable for detection at the surface. Undercounting as a result of this is known as *availability bias*, because animals are unavailable to be seen. Even if a group of animals on the transect line is at the surface, it may not be detected because of observation conditions or simply by chance; this is known as *perception bias*. Estimates of cetacean abundance that are not corrected for animals missed on the transect line are thus negatively biased to an unknown extent. In the literature, the probability of detection on the transect line, encompassing availability and perception, is typically known as *g(0)*.

A standard way to correct for animals missed on the transect line is to collect data from two independent observation platforms (double observer team data), identify duplicate detections, and implement so-called *mark-recapture distance sampling* methods (e.g., Burt et al., 2014), which allow correction for perception bias. *g(0)* is likely to vary with sighting conditions (Barlow, 2015), which may thus need to be taken into account in analysis (e.g., Moore and Barlow, 2013).

Double-observer team data collection is usually not possible on small boats used in coastal waters, and small aircraft may also not be able to accommodate two independent teams of observers; estimates of abundance from such surveys will therefore not be corrected for perception bias. Some recent work explores the use of passive acoustic data collected from a towed hydrophone in combination with a conventional visual observer team to estimate *g(0)* (Martin et al., 2020; Rankin et al., 2020).

There is also the potential to correct for availability bias using double observer team data. On ships, this can be implemented using the so-called "tracker mode," which involves one team of observers (tracker) searching far ahead of the vessel and tracking detected animals until after they become available to the second team (primary) searching closer to the vessel (e.g., Hammond et al., 2013). However, this method may not correct, or fully correct, for availability bias, depending on the species. In particular, as mentioned above in the context of acoustic data, long-diving species such as beaked whales or sperm whales may be underwater, and thus unavailable, for the whole time that a survey ship passes by. On aerial surveys, the circle-back or "racetrack" method (Hiby, 1999) was developed to correct for both perception and availability bias for relatively short-diving species, such as the harbor porpoise (e.g., Scheidat et al., 2012; Hammond et al., 2013). In this method, triggered by a sighting of an animal or group of animals, the aircraft circles back and re-surveys the same piece of transect line to provide the equivalent of double observer team data.

Alternatively, availability bias can be corrected for using data on the proportion of time animals or groups of animals spend on the surface; such data may come from observations of diving behavior (e.g., MacKenzie and Clement, 2014; Sucunza et al., 2018) or telemetry (e.g., Heide-Jørgensen and Laidre, 2015).

Some additional studies that estimate the proportion of animals seen on the transect line, *g(0)*, include Marsh and Sinclair (1989) and Laake et al. (1997) for aerial surveys, and Barlow (2015); Moore and Barlow (2013), and Pike et al. (2019) for ship surveys.

A further assumption of line transect sampling is that animals do not move prior to detection. Random movement only becomes important at slow survey speeds; bias in estimated abundance is positive but small unless survey speed is less than around twice the speed of movement of the animals (Buckland et al., 2001). This is typically not the case but may need to be considered if the survey vessel is a yacht, for example. However, some species may react to approaching survey ships by moving away from or toward them prior to being detected, resulting in biased measurements of perpendicular distance. Such responsive movement is typically not a problem for aerial surveys because of the relatively high speed of aircraft. From ships, responsive avoidance or attraction may lead to under- or over-estimation of abundance, respectively. Over-estimation (positive bias) as a result of attraction prior to detection has been shown to be substantial in some studies (e.g., Turnock and Quinn, 1991; Cañadas et al., 2004). One method of identifying and dealing with responsive movement prior to detection on ship surveys is described in Palka and Hammond (2001).

As with other methods, it is assumed that data are collected accurately but this can be a considerable challenge in line transect (or strip transect) surveys for cetaceans. The basic data related to sightings of animals (species identity, perpendicular distance, group size) can all be difficult to determine or measure, especially from a ship, and errors can lead to bias. Issues related to measurement error in estimates of distance and angle to groups sighted on boat surveys are discussed by Williams et al. (2007). The potential bias in estimates of abundance caused by errors in distance and angle measurements is investigated by Leaper et al. (2010). A photographic method to improve the accuracy of these measurements (Gordon, 2001; Leaper and Gordon, 2001) has been further developed and used in surveys in the European Atlantic (e.g., Hammond et al., 2013). Bias in group size estimation may have a substantial effect on abundance estimates. For example, in transect surveys of pelagic dolphins, Gerrodette et al. (2019) found that observers underestimated group sizes greater than 25 animals and that this negative bias increased with group size. Boyd et al. (2019) described methods for estimating group size from photographic or video images taken on aerial survey. Observer training in group size estimation may improve estimates by reducing variance among observers and reducing bias.

Model-Based Estimation of Abundance

An alternative to the design-based estimation methods of analysis described above is to model observed density along the transect lines as a function of environmental covariate data and to use the model to predict density over the entire survey area to obtain an estimate of abundance (e.g., Gilles et al., 2016; Panigada et al., 2017; Rogan et al., 2017; Becker et al., 2020). This *model-based estimation* of abundance is also referred to as density surface modeling (Miller et al., 2013) or species distribution modeling (Zurell et al., 2020). Note that detection probability must be estimated and incorporated in analysis for model-based estimates of abundance to be unbiased. Relevant environmental covariates could be spatial (e.g., latitude and longitude), physiographic (e.g., depth, slope), or reflect dynamic oceanography (e.g., sea surface temperature).

There are several attractive features of model-based estimation of abundance. The use of additional covariate data means that model-based estimates of abundance are potentially more precise than design-based estimates calculated from the same survey data (e.g., Rogan et al., 2017). Moreover, the data do not have to come from surveys designed to achieve equal coverage probability of the survey area, so abundance can be estimated from surveys where this has not been possible (e.g., Cañadas and Hammond, 2006, 2008; Williams et al., 2006, 2011). Data collected from platforms of opportunity such as fishery surveys, ferries, or whale-watching vessels could thus be used to estimate abundance using model-based methods, as long as the base survey data themselves are adequate. A potentially useful feature of model-based estimation is that abundance can be estimated for any defined area, rather than being restricted to the survey blocks in the design-based method.

However, there are some important issues to consider when using model-based methods. As in any model, the more covariates included, the larger the number of observations needed to support a more complex model. Model-based methods may thus not be appropriate for datasets with a small number of observations. In addition, the range of values of the environmental covariates in the area for which abundance is to be estimated (the "covariate envelope") must have been adequately sampled by the survey to ensure robust prediction of abundance. Prediction of any model outside the range of the data used by the model should be done with caution and estimation of abundance in areas or in a "covariate envelope" not covered by the data may be challenged. However, there is an increasing need to estimate abundance in unsurveyed areas and methods for extrapolating model-based density to such areas are in development (see e.g., Mannocci et al., 2017; Bouchet et al., 2020).

Perhaps the most important issue is the obvious point that model-based estimates of abundance are dependent on the model fitted to the data. Good modeling practice can be gleaned from appropriate texts and example studies (e.g., Gilles et al., 2016; Becker et al., 2020) but there are nevertheless multiple ways in which models can be applied and different models will produce different estimates of abundance. Selection of the "best" model can be challenging. Estimating the precision of model-based abundance estimates can also be more challenging than it is for design-based estimation; propagating uncertainty in all model parameters is complex and is an active area of research (Becker et al., 2020; Sigourney et al., 2020).

Land- or Ice-Based Counts of Migrating Baleen Whales

Some populations of baleen whales, namely gray, humpback and bowhead whales, migrate close to land or ice and can be counted as they pass a suitable observation point. However, even if the whole population migrates past such a point, not all animals will be observed and counted, so the counts need to be corrected and extrapolated to obtain unbiased estimates. Whales can be missed because they migrated past the observation point before or after the observation period, at distances beyond visual range, or during periods when counting could not take place, such as at night or during poor weather. These whales are thus unavailable to be counted, leading to *availability bias* described above, which needs to be corrected for with correction factors derived from additional data and/or models.

Whales may also be missed because they were available to be seen but were not detected (equivalent to *perception bias* described above). A correction factor for this bias can be estimated from data collected at two observation points in the form of a mark-recapture experiment, equivalent to *mark-recapture distance sampling*. In addition, in studies where whales are typically detected in groups, the number of whales in a group may be underestimated and need to be corrected for using data collected additional to the main study.

Studies to estimate abundance of migrating baleen whales that include correction factors such as those described above include: gray whales (Rugh et al., 2005; Laake et al., 2012), humpback whales (Noad et al., 2011) and bowhead whales (George et al., 2004; Givens et al., 2016).

The number of migrating whales can also be estimated using aerial survey, rather than land- or ice-based counts. Salgado Kent et al. (2012) employed this method to estimate the abundance of humpback whales off western Australia on both north- and southbound migration. In this study, challenges in allocating temporarily milling whales to the north- or southbound component of migration was identified as a factor that could influence estimates of abundance.

Mark-Recapture

Recognition of Individual Animals

If individual animals are naturally or artificially marked, and can subsequently be recognized in the future, mark-recapture methods can be used to estimate abundance (Hammond et al., 1990; Amstrup et al., 2005; Hammond, 2018). The conventional way to mark an animal is to capture it physically and apply an artificial mark or tag, for example, a colored band or ring attached to the leg of a bird. Studies of marine mammals in which artificial marks or tags have been applied to individuals have typically been undertaken to estimate correction factors for pinniped counts or to estimate survival rates (McIntosh et al., 2013; Greig et al., 2019; Brusa et al., 2020). Population size has been estimated from analysis of data from leopard seals (*Hydrurga leptonyx*) marked with flipper tags (Jessopp et al., 2004; Forcada and Robinson, 2006) and northern elephant seals (*Mirounga angustirostris*) marked with hair dye (García-Aguilar and Morales-Bojórquez, 2005).

In some species of marine mammal, individual animals can be recognized from natural markings on their skin or pelage, or the nicks and notches on their dorsal or tail fin. Individuals are "captured" and "marked" by taking high-quality photographs of these natural markings, avoiding the need for physical capture or artificial marking. This method of data capture is known as photo-identification, or photo-id. Some example studies estimating the abundance of species that possess natural markings using mark-recapture methods applied to photo-id data are given in **Table 2**.

Photographs may be taken on land for pinnipeds and at sea for most species of cetacean; aerial photographs can be used for some species of large whale, e.g., right and bowhead whales. Processing includes grading photographs for image quality and matching images to a catalog of previously identified individuals. Urian et al. (2015) provides a good general review of best practices for implementing photo-id methods to generate data for estimating abundance using mark-recapture. Issues that need consideration include photographic quality, distinctiveness of individuals, matching criteria, and the permanence or evolution of natural markings.

TABLE 2 | Example studies applying mark-recapture analyses to photo-identification data to estimate the abundance of cetacean and pinniped species.

Species	Area	References
Humpback whale	North Atlantic, Ecuador, North Pacific, Oceania	Smith et al., 1999; Stevick et al., 2003; Calambokidis and Barlow, 2004; Barlow et al., 2011; Felix et al., 2011; Constantine et al., 2012
Fin whale	Gulf of St Lawrence	Ramp et al., 2014
Southern right whale	Argentina	Whitehead et al., 1986
North Atlantic right whale	Western North Atlantic	Pace et al., 2017
Blue whale	Eastern North Pacific, Chile	Calambokidis and Barlow, 2004; Galletti Vernazzani et al., 2017
Killer whale	Alaska, Norway	Durban et al., 2010; Kuningas et al., 2014
Common bottlenose dolphin	Eastern United States, Scotland	Read et al., 2003; Balmer et al., 2008; Arso Civil et al., 2019b
Indo-Pacific bottlenose dolphin	Western Australia	Smith et al., 2013
Indo-Pacific humpback dolphin	Hong Kong, Malaysia	Chan and Karczmarski, 2017; Kuit et al., 2021
Irrawaddy dolphin	Sarawak	Minton et al., 2013
Gray seal	France	Gerondeau et al., 2007
Harbor seal	Scotland	Cordes and Thompson, 2015
Monk seal	Western Sahara, Hawaii	Forcada and Aguilar, 2000; Martínez-Jauregui et al., 2012; Baker et al., 2016
Leopard seal	South Georgia	Forcada and Robinson, 2006
Saimaa ringed seal	Finland	Koivuniemi et al., 2016, 2019

The process of matching individuals becomes considerably more time consuming as a catalog of individuals becomes large. Computer software to assist matching individuals of some species has been available and used for some time, e.g., for gray seals (Hiby and Lovell, 1990) and humpback whales (Mizroch et al., 1990). Two generic systems currently available are the Interactive Individual Identification System (I3S)[7]; and the Image-Based Ecological Information System (IBEIS) accessible via Wildbook[8]. Recent developments of automated matching systems for cetaceans include Bogucki et al. (2019) for right whales and Cheeseman et al. (in press) for humpback whales. Some projects encourage so-called citizen science by providing portals for members of the public to upload photographs for matching[9] [10].

In addition to physical marking/tagging and photo-id, individuals can be marked using their DNA as a genetic marker. As with photo-id, this avoids the need for physical marking and, if a remote biopsy sample is taken, physical capture (e.g., Palsbøll et al., 1997). Conventional mark-recapture models can then be applied; examples include studies of Māui dolphins (*Cephalorhynchus hectori maui*) (Hamner et al., 2014), North Pacific right whales (*Eubalaena japonica*) (Wade et al., 2011) and humpback whales in Oceania (Constantine et al., 2012). Genetic data from individuals can also be used to estimate population size in close-kin mark-recapture analysis (Bravington et al., 2016; Ruzzante et al., 2019). While the use of this method is still in its infancy, it has some advantages over traditional mark-recapture methods that may prove useful in some circumstances.

If data from more than one mark type are available, for example left-and right-side photo-id images of individuals or both photo-id and genetic markers, it may be possible to use both data sources simultaneously to estimate abundance (see Madon et al., 2011; McClintock et al., 2013; McClintock, 2015).

On rare occasions, it may be possible to identify all, or almost all, of the animals in a population over a season, in which case the data provide a (nearly) complete count and there is no need for mark-recapture analysis. This is most likely to occur only for small populations in which individuals are strongly resident in a small area, e.g., 'southern resident' killer whales, which number less than 100 (Olesiuk et al., 1990). So-called discovery curves (e.g., Balmer et al., 2008) of the cumulative number of newly identified individuals plotted over a period of time in which the population is assumed to be closed (see below) may help to ascertain this.

Mark-Recapture Estimation of Abundance

Using mark-recapture methods to estimate the size of a population is most easily illustrated by considering data from just two samples, a capture occasion and a recapture occasion.

---

[7] https://reijns.com/i3s/

[8] https://www.wildbook.org/doku.php

[9] https://happywhale.com/home

[10] https://www.norwegianorca-id.no/

Consider that a sample (number) of individuals, n_1, is captured, marked and released back into the population. After a period to allow mixing of animals, a second sample of individuals, n_2, is captured. If a number of these individuals, m_2, have been previously marked (and are thus recaptures), and if some assumptions are met (see below), the estimated proportion of marked animals in the second sample, $\hat{p}_2 = \frac{m_2}{n_2}$, should equal the proportion of marked animals in the whole population, $\frac{n_1}{N}$, where N is population size.

Equating these terms and rearranging the equation leads to an estimator of the size of the population: $\hat{N} = \frac{n_1 n_2}{m_2}$, usually known as the Petersen two-sample estimator. Alternatively, we could express estimated population size as: $\hat{N} = \frac{n_1}{\hat{p}_2}$, to illustrate that mark-recapture methods correct the count of individual animals captured in a sample by an estimate of the probability of capturing an animal. Mark-recapture analysis can also be thought of as a way to estimate the number of animals in the population that were never captured during the study.

Abundance, thus estimated, is defined as the number of individuals that were present in the study area during the period when sampling took place. Note that, in situations where animals move in and out of the study area during the sampling period (see below), this estimate of abundance may be different from that obtained from a survey over a shorter period of time (see "Mark-recapture using photo-id data or line transect sampling?").

Most mark-recapture studies, including those referred to above, use multiple samples, in which all samples except the first include recaptures of marked animals and captures of animals not previously encountered. The data used in analysis are the capture histories of each individually identified animal, typically represented by a string of 1s (captured) and 0s (not captured). Options for analysis range from relatively simple "conventional" models to those that allow relaxation of assumptions made by simpler models (see below). Mark-recapture analysis is often conducted in the purpose-specific software MARK[11] (see "Data analysis").

In some species, particularly delphinids, not all individuals in the population possess natural markings that allow them to be recognized. In these cases, mark-recapture analysis of these data only estimates the abundance of animals with adequate markings. To obtain estimates of abundance of the whole population, the proportion of unmarked animals in the population must be estimated (e.g., Urian et al., 2015).

Mark-Recapture Assumptions

Regardless of how an animal is marked, mark-recapture methods assume that individuals are uniquely marked, that marks cannot change or be lost, and that all marked animals are recognized and recorded correctly. The assumption that the data are accurate is particularly important in mark-recapture analysis because errors in recording the number of recaptures can lead to substantial bias in estimates of population size if sample sizes are small. Additional assumptions made when applying mark-recapture methods relate to the models used for data analysis (e.g., Hammond, 2018).

An important assumption made by conventional mark-recapture models is that, within a sampling occasion, all animals have the same probability of capture. In studies where individuals are physically captured and marked, capture probability may change as a result of marking. However, the most common violation of this assumption for studies of cetaceans using photo-id is that capture probabilities vary from one individual to the next – often described as heterogeneity of capture probabilities. There are several reasons for this. Individuals may have preferences for different areas leading to variation in the probability of encountering an animal if not all areas inhabited by the population are equally sampled, as is commonly the case. Individuals may also respond differently to being approached for photography, and variation in natural markings among individuals may lead to variation in ability to identify them. Capture probability may vary by sex or reproductive status as well as from individual to individual.

For these reasons, heterogeneity of capture probabilities is a feature of most cetacean photo-id datasets, even when field methods are implemented to minimize the problem. If this is not explicitly dealt with by modifying conventional mark-recapture models in analysis, estimates of population size will be negatively biased (e.g., Hammond, 2010, 2018). Animals with high capture probability will be recaptured more often than average, leading overall to more recaptures than expected and to negative bias (see equation for Petersen two-sample estimator above). Conversely, animals with very low capture probability may never be seen and thus not be included in the population estimate, also leading to negative bias.

Another key consideration is whether the population can be assumed to be demographically closed, i.e., no recruitment, death or permanent immigration/emigration takes place during the study period. If this can be assumed, *closed population models* (see Hammond, 2018) can be used in analysis. For this to be true, the study period must clearly be short relative to the life history of the study species. It is commonly accepted that data collected within a year adequately meet the assumption of demographic closure for marine mammals. For longer-lived species with lifespans of several decades, it may be acceptable to use closed models to analyze data from two or more years (e.g., Smith et al., 1999) because the bias introduced by failure of the closure assumption is relatively small (Hammond, 1986). Justifying the use of closed models is potentially important because it is straightforward to relax the assumption of equal capture probabilities, thus minimizing or avoiding the associated negative bias otherwise associated with heterogeneity (Wilson et al., 1999; Calambokidis and Barlow, 2004; Kuningas et al., 2014; Curtis et al., 2021).

Open population models (see Hammond, 2018) that allow demographic change may be required to analyze longer time series of data (i.e., several years). However, modifying open population models to fully relax assumptions about equal capture probabilities is rarely possible and they may also be difficult to fit to data (e.g., Curtis et al., 2021). Open population models are thus less commonly used for estimating the abundance of marine mammal populations.

The open population model POPAN estimates the size of a so-called "superpopulation." This is an estimate of the total number

---

[11] http://www.phidot.org/software/mark/

of animals using the study area that were alive at any point over the whole study period; for any given year it thus includes animals that have died and those that have yet to be recruited. Studies that have used this model include Constantine et al. (2012) for humpback whales in Oceania and Galletti Vernazzani et al. (2017) for blue whales (*Balaenoptera musculus*) off Chile.

Other studies have used open population models of survival to estimate annual recapture probabilities, p_t and then used them to correct the number of animals captured each year, n_t. Abundance in each year, t, is then estimated as $\widehat{N}_t = \frac{n_t}{\hat{p}_t}$, which is equivalent to the equation relating to the Petersen estimator. Studies using this approach include Taylor et al. (2009) for polar bears (*Ursus maritimus*) and Madon et al. (2013) for humpback whales. However, this approach assumes that capture and recapture probabilities are equal, which is likely not to be the case in most studies. More complex open population mark-recapture models to estimate abundance can be developed if data are available, for example for western gray whales (Cooke et al., 2005) and for North Atlantic right whales (*Eubalaena glacialis*) (Pace et al., 2017).

It may be preferable to use closed population models fitted to data within each year if multiple samples are available within each year. Alternatively, a method of analysis that combines open and closed population models, known as "robust design," can be applied to data to estimate population size within years and survival rates between years (e.g., Smith et al., 2013; Arso Civil et al., 2019a,b) or seasonal variation in population size (e.g., Cordes and Thompson, 2015). Robust design models can also estimate temporary emigration/immigration rates, the incorporation of which can reduce any bias in annual estimates of population size (and survival rate) caused by inter-annual movement of animals out of and into the study area. These models can be powerful tools but to use them appropriately requires studies that generate representative data both within and among years.

PRACTICALITIES AND RESOURCES

In some cases where the abundance of a population of marine mammals is to be estimated, the choice of a suitable method is fairly clear. For example, extrapolating counts of either pups or all-age animals is likely to be the main method considered to estimate the abundance of pinniped species that form aggregations on land because counts are typically relatively easy to conduct. For ice-associated species, however, their scattered distribution in large and difficult-to-access areas will typically require an aerial survey. Ship line transect surveys may be the only feasible method to estimate the abundance of cetacean populations inhabiting large areas far from land. However, there are circumstances in which the choice of method may not be clear.

Given the species of interest and its primary habitat, understanding the practical requirements and resources needed to collect the data to implement a given method is a good starting point. **Table 3** summarizes this information for each of the methods described above. Even if the choice of methods is clear, it should be useful to summarize what the commitments are likely to be. Some particular issues are discussed below. Whatever method is used, it is crucial that personnel have adequate expertise and receive the appropriate training.

Figure 2 summarizes the general flow of information and activities when considering the estimation of abundance of a marine mammal population.

Pinniped Counts – Pups or All-Age Animals?

Section "Correcting and extrapolating counts of a defined portion of a population" and **Table 3** describe the basis and requirements for studies to estimate the abundance of pinniped populations from counts. When considering such studies, it may be required to decide whether to count pups or animals of all ages. The practical aspects of ensuring that counts are as accurate as possible are similar, but the choice may be informed based on how available the animals are to count.

In most species, pups are available at a defined time of the year and, in many species, pups are ashore for many days or even weeks, facilitating both planning and the actual counting. If there is a period of peak pupping, pup counts can be focused at this time both to maximize the count but also to ensure consistency across years. Care may be needed, however, if the pupping period shifts over time. Counts of mature animals can also be made at pupping/breeding colonies but immature animals are unlikely to be included.

Counts of pups may not be straightforward to obtain for some species; for example, harbor seal (*Phoca vitulina*) pups can enter the water soon after birth. For such species, or where counting pups is logistically challenging, counting animals of all ages may be required or desirable. In these cases, it is important to determine an appropriate time to make counts, ideally when the largest proportion of the population is hauled out. For phocid seals, the annual molt may be a good time.

The difference in how pup counts and all-age counts are extrapolated to the whole population may also be an important factor in choice of method. For pups, life history information is needed to calculate a "pup multiplier" or as input to a population model. For all-age animals, information on haul-out behavior, preferably from telemetry, is needed to estimate the proportion of animals ashore when counts are made.

Mark-Recapture Using Photo-Id Data or Line Transect Sampling?

If individuals in a population of cetaceans possess natural markings suitable for photo-identification, e.g., humpback or blue whales, or bottlenose dolphins (*Tursiops truncatus*), viable alternative methods could be mark-recapture analysis of photo-id data or a line transect sampling using a ship, small boat or aerial survey.

For mark-recapture estimation, the distribution and/or size of the population and the behavior of the animals can make it challenging to achieve capture probabilities that are sufficiently high for abundance to be estimated with the desired level of accuracy and/or precision. In such cases, a line transect

TABLE 3 | Summary of practical requirements and resource needs for obtaining data to estimate abundance of marine mammal populations.

Taxon	Data collection method	Practical requirements and considerations	Resources (time, personnel, equipment, costs)
Pinnipeds	**Pup counts** Pups can be counted at breeding colonies on land or ice during the pupping season and these counts, corrected as necessary, can be used to estimate population size using a multiplier or a population model.	Counts from land are logistically straightforward once at the colony but an elevated observation location may be needed for effective counting. Methods may be needed to correct for animals not able to be counted. Counts from aerial survey (needed for widely distributed ice-associated seals) are logistically more challenging (safety, airports, fuel, photography). If pups cannot be counted from the whole area or all colonies, they may be extrapolated to a total count. One count may suffice but a set of counts through the pupping season may be required to estimate total pup production and quantify uncertainty. Information on birth and survival rates is needed for a life table to calculate a "pup multiplier," or to be used in a population model to extrapolate the pup count to the total population, ideally from the target population.	Time and personnel needed to conduct ground counts depends on the number and size of the colonies. Equipment needed is minimal. For aerial surveys, time needed depends on the extent of the area covered and the length of the pupping season. The cost of aircraft use is high and depends on flying time and/or study duration of sampling in remote areas. Obtaining new data on birth and survival rates may require a considerable investment in time, personnel and cost.
	All-age counts Animals other than pups of the year can be counted at breeding colonies or other haul-out sites on land or ice and these counts, corrected as necessary, can be extrapolated to the whole population using data on the proportion of animals hauled out.	The best time to conduct counts (season, time of day, stage of tide, etc.) needs to be considered. Counts from land – see pup counts, above. Counts from aerial survey – see pup counts, above. One count may suffice but multiple counts are preferable to quantify variability. Data on haul-out behavior are needed to estimate the proportion of animals hauled out on land or ice at the time of the counts. These may be observations of marked animals at haul-out sites or telemetry data from animals fitted with transmitters (see marking, below).	Ground counts – see pup counts, above. Aerial surveys – see pup counts, above. Obtaining new data on haul-out behavior may require a considerable investment in time, personnel and cost. If telemetry tags are deployed, the equipment cost will depend on the type and number of tags.
	Photo-id Individuals can be uniquely recognized from photographs of natural markings and abundance estimated using mark-recapture methods. See also genetic tagging below.	Animals in the species/population must possess natural markings suitable for recognizing individuals from photographs that last for the period of the study. The population of individuals available to be photographed should match the population for which abundance is needed. Multiple samples need to be collected in a design appropriate for analysis. Taking photographs is straightforward but the appropriate part of the animal must be able to be captured. If all animals are not uniquely recognizable, the proportion of unmarked animals must be able to be estimated.	Photographing a sufficient number of animals, in a sufficient number of colonies or haul-out sites, over multiple sampling occasions requires a considerable investment in time. A good quality camera and lens is needed. Processing and matching photographs requires experienced personnel and substantial time, depending on the number of images taken. Computer-assisted matching can reduce processing time and costs. Developing a system, if one is not already available, may require considerable resources.
	Marking (including tagging) Animals can be physically captured and marked, and abundance estimated using mark-recapture methods.	A suitable way to mark individual animals must be available (e.g., flipper tags, hair dye, telemetry). Animals must be accessible for marking/tagging. The population of individuals available to be marked should match the population for which abundance is needed. Multiple samples need to be collected in a design appropriate for analysis. Possible loss of marks over time needs to be considered.	Marking/tagging a sufficient number of animals, in a sufficient number of colonies or haul-out sites, over multiple sampling occasions may require a considerable investment in time. Equipment needed is minimal unless telemetry tags are used.

(Continued)

TABLE 3 | (Continued)

Taxon	Data collection method	Practical requirements and considerations	Resources (time, personnel, equipment, costs)
Cetaceans	**Migration counts** Animals can be systematically counted on migration from a land- or ice-based vantage point–abundance may be estimated from corrected counts.	Animals must migrate close enough to land or ice to be counted and there must be a suitable place for a fixed observation platform to count from. The counting period should aim to cover the whole migration, which could be several weeks. Counts need to be corrected for animals missed either because they passed too far offshore to be counted or for periods when counting cannot take place, such as during poor weather or at night. Counts should be corrected for animals missed by chance (perception bias). Bias in observed group size may need to be corrected.	Maintaining an observation platform for the duration of the migration period requires a considerable investment in time and personnel. Costs depend on the length of the counting period. Equipment needed includes binoculars and theodolites. Correction of animals missed by chance may require two observation platforms, with associated increased personnel, equipment and cost.
	Photo-id and genetic tagging Individuals can be uniquely recognized from photographs of natural markings (or genetically) and abundance estimated using mark-recapture methods.	For photo-id, animals in the species/population must possess natural markings suitable for recognizing individuals from photographs that last for the period of the study. For genetic tagging, animals must be accessible for biopsy sampling. The population of individuals available to be photographed/biopsied should match the population for which abundance is needed. Multiple samples need to be collected in a design appropriate for analysis. Taking photographs of the appropriate part of the animal may not be straightforward. For most species this must be done at sea, ideally from a small boat. For some species, photographs may come from aerial survey. If not all animals are uniquely identifiable, the proportion of unmarked animals must be able to be estimated.	Photographing a sufficient number of animals, in an area of appropriate size, over multiple sampling occasions may require a considerable investment in time. If the population to be sampled is distributed offshore, a suitable ship capable of staying at sea for extended periods may be necessary. Such ships are expensive to use. If the population is coastal, a small boat for day trips is required, which is much less expensive. A good quality camera and lens is needed. Processing and matching photographs requires experienced personnel and substantial time, depending on the number of images taken. Computer-assisted matching can reduce processing time and costs. Developing a system, if one is not already available, may require considerable resources.
	Line transect: ship or small boat Abundance can be estimated from ship or small boat line transect surveys of a defined area.	The defined survey area needs to match the area for which abundance is needed. If the defined survey area extends far from land, a ship of sufficient size must be available. Survey design needs to accommodate any directed migratory or seasonal movements of animals. Correction for perception bias and any responsive movement requires a ship that can accommodate two observation teams. If the defined survey area is coastal, a small boat may be a suitable survey platform, but collection of data to correct for perception bias and any responsive movement is likely unfeasible. If availability bias is to be corrected for using data collected on survey, the two-team tracker configuration may be needed. If data to correct for availability bias are not collected on survey, surfacing/diving behavior data will be needed to correct for this. Ideally, these data should be for the population being surveyed.	Ships capable of surveying offshore areas and staying at sea for extended periods of time are very expensive to use. Cost is based on days at sea, regardless of ability to survey. A team of observers, or two teams, the number of people depending on the data collection protocol, is required for the duration of the survey, which could be several weeks on offshore surveys. Observers require adequate training in data collection procedures, including species identification, group size estimation, and perpendicular distance data collection. Equipment needed depends on the data collection protocol but at minimum comprises binoculars and equipment for measuring distance and angle. Small boats for surveys in coastal waters are relatively inexpensive. Observation teams are typically smaller, but equipment needs are likely to be similar to ships used in offshore surveys. If existing surfacing/diving data to correct for availability bias are not available, the collection of new data may require a considerable investment in time and cost.

(Continued)

TABLE 3 | (Continued)

Taxon	Data collection method	Practical requirements and considerations	Resources (time, personnel, equipment, costs)
	Line transect: aerial Abundance can be estimated from line transect surveys of a defined area from an aircraft.	The defined survey area needs to match the area for which abundance is needed. The defined survey area must be able to be surveyed by aircraft safely. Survey design needs to accommodate any directed migratory or seasonal movements of animals. Unless the circle-back method can be implemented: - for perception bias to be estimated, the aircraft must be able to accommodate two independent observation teams; and - surfacing/diving behavior data will be needed to correct for availability bias.	The cost of aircraft use is high and depends on flying time and/or duration of the sampling period in remote areas. Total cost will therefore be driven primarily by total transect length, which will depend on the size and coverage of the area surveyed, and/or the location of survey. Suitable aircraft (ideally high-winged and with bubble windows) need to be available. Aircraft that can accommodate two independent observation teams are typically more expensive to use than those which cannot. Observers require adequate training in data collection procedures, including species identification, group size estimation, and perpendicular distance data collection. The observation team for aerial survey is typically smaller than for ship/boat surveys. Equipment for measuring perpendicular distance is needed. If existing surfacing/diving data to correct for availability bias are not available, the collection of new data may require a considerable investment in time and cost.
Sirenians	**Aerial strip transect** Abundance can be estimated from counts made on aerial strip surveys.	The defined survey area needs to match the area for which abundance is needed. The defined survey area must be able to be surveyed by aircraft safely. Counts need to be corrected for animals missed because they are unavailable (e.g., submerged in turbid water) or for other reasons.	See Line transect: aerial above.

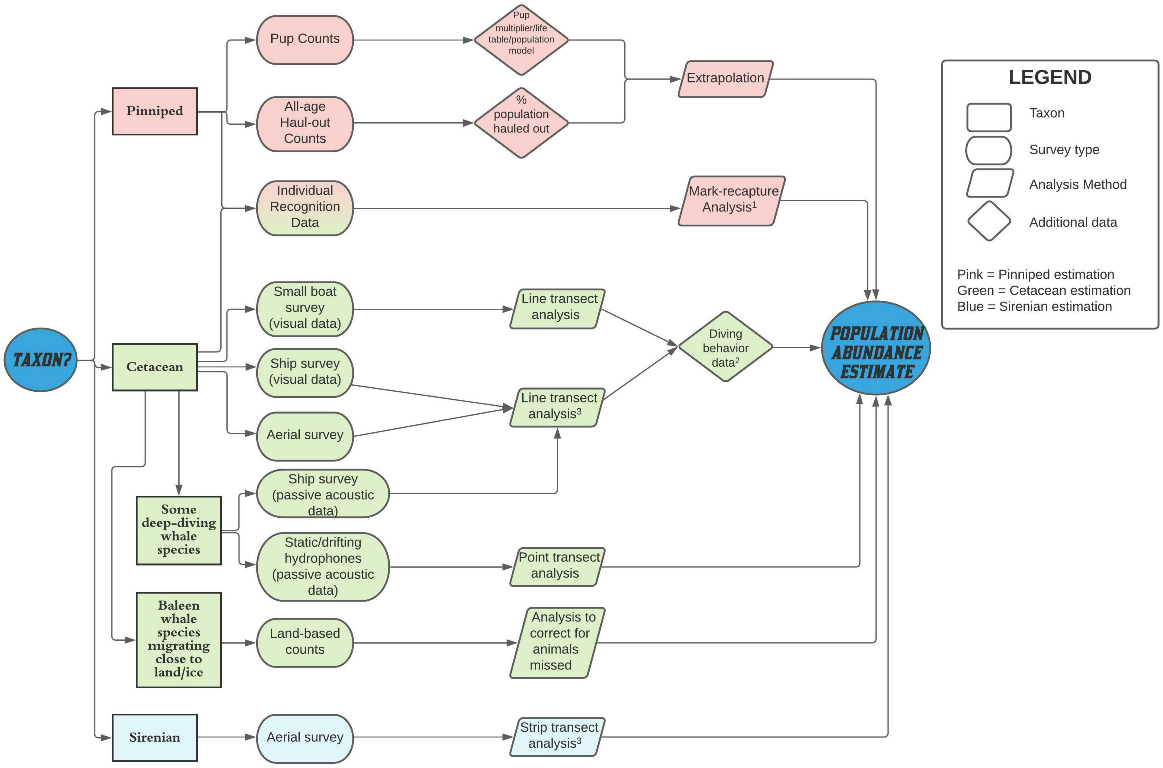
survey would be more effective. Multiple case-specific factors mean that it is not possible to generalize on how high capture probabilities should be, but the studies in **Table 2** provide examples for guidance.

As mentioned above in relation to definition of the population being estimated, surveys using line transect methods sample density within a defined area and thus estimate abundance within that area at the time of the survey. Mark-recapture methods sample individuals and therefore estimate the size of the population using the study area during a study period. Estimates of these two quantities may not be the same even for the same study/survey area. If the mark-recapture study period is longer than the line transect survey and if animals move in and out of the study area during that period, the population sampled will comprise more animals than are present in the area at one time. This may be important in terms of the objectives for the abundance estimate. If the objective of the study is to estimate abundance in an area at a particular time, a line transect survey may be most appropriate, but if it is to estimate the number of animals using the area over a longer period, mark-recapture could be more appropriate. Abundance has been estimated and compared using these two methods for humpback and blue whales off the west coast of the United States and Mexico (Calambokidis and Barlow, 2004), and for killer whales in British Columbia (Williams and Thomas, 2009).

Two studies that consider some of these issues for small cetaceans are Minton et al. (2013) and Kuit et al. (2021). The first study used small boat line transect surveys to estimate the abundance of Irrawaddy dolphin (*Orcaella brevirostris*) and Indo-Pacific finless porpoise (*Neophocaena phocaenoides*) in coastal waters of Sarawak, Borneo, and also used mark-recapture analysis of photo-id data to estimate the number of Irrawaddy dolphins using the study area. The estimates of Irrawaddy dolphin from the two methods were broadly comparable but those from mark-recapture were around 50% larger, a result interpreted by Minton et al. (2013) as reflecting the mark-recapture analysis representing a larger population beyond the study area.

The second study used small boat line transect surveys to estimate the abundance of Irrawaddy dolphin and Indo-Pacific finless porpoise in coastal waters of Matang, Malaysia, and mark-recapture analysis of photo-id data to estimate the number of Indo-Pacific humpback dolphins (*Sousa chinensis*) using the study area. Kuit et al. (2021) did not use mark-recapture for Irrawaddy dolphins because the animals' behavior led to poor-quality photographs and adversely affected individual identification. Minton et al. (2013) also noted that obtaining photo-id pictures of Irrawaddy dolphins was challenging.

There may be practical implications in terms of time to conduct the sampling and/or process the data. A line transect survey can be conducted relatively quickly and only one survey



1. Correct for proportion identifiable in the population, as appropriate
2. To correct for availability bias, as appropriate
3. Can incorporate corrections for perception bias (with double observer teams) and availability bias (method dependent)

FIGURE 2 | Simplified schematic of the flow of information and activity when estimating the abundance of marine mammal populations.

is needed to estimate abundance. Mark-recapture studies require at least two and ideally several capture occasions over a period of time that allows the population to mix between occasions. It is likely to take longer to process photo-id data than line transect survey data prior to analysis. Implementing multiple sampling occasions and processing photo-id data have cost implications that may exceed those of conducting a line transect survey (see **Table 3**).

There may also be practical implications in terms of observation platform. Photo-id can be done from ships but is best done from small boats. In riverine or enclosed marine habitats where navigation is impractical for larger vessels and where survey conditions are good, small boats may be the best platform for line transect surveys. However, small boats may have limitations, depending on the survey area and the method used. The observer is typically not very high above water level thus limiting the area that can be searched. It is unlikely that animals missed on the transect line as a result of perception bias will be able to be corrected for. Any movement of animals in response to the approaching boat prior to detection will also be impossible to account for; this may be important for some coastal species. Aerial surveys, on the other hand, are immune to responsive movement, and it is feasible to collect two-team data to estimate perception bias with an aircraft of sufficient size. However, aerial surveys are likely to be much more expensive and the logistics (surrounding terrain, location of suitable airports, etc.) and safety issues are more challenging (see below). Practical advice for designing and conducting line transect surveys for cetaceans in coastal and riverine habitats can be found in Dawson et al. (2008).

Aerial Survey or Ship Survey?

Some cetacean species are not typically confined to coastal waters but nevertheless may not occur a long way offshore. For example, harbor porpoises are largely found on the continental shelf in European Atlantic waters (Hammond et al., 2013). Even if the distribution of the species of interest extends beyond this, the objective may nevertheless be to estimate abundance within a limited range, for example a country's 200 nm exclusive economic zone. The choice of whether to survey using a ship or an aircraft may therefore arise.

A crucial overarching consideration is safety. Accidents can happen at sea and in the air but the consequences are inherently far more serious in aerial surveys; fatal accidents have occurred during aerial surveys of cetaceans and seals. The risks can be reduced by careful consideration of the type of aircraft, the environmental conditions, and the logistics. Twin-engine aircraft should always be used when at all possible, especially in waters away from the coast. Experienced pilots are essential. Important logistical issues include the location of suitable airports and the availability of fuel. Aerial surveys should only be undertaken by personnel who recognize the risks and where those risks have been mitigated to the extent possible.

Regarding choice of equipment to maximize the effectiveness of data collection, high-wing aircraft with bubble windows optimize visibility on and around the transect line. A clinometer is needed for measuring perpendicular distance; the recently developed "geometer" (Hansen et al., 2020) facilitates this and minimizes error.

Camera systems have been used on aerial surveys for some time to count pinnipeds on land or ice (e.g., Speckman et al., 2011; Russell et al., 2019; Thompson et al., 2019; Stenson et al., 2020) and also some cetaceans (e.g., beluga and narwhal – Gosselin et al., 2017; Doniol-Valcroze et al., 2020). They have also been used on aerial surveys for seabirds (e.g., Buckland et al., 2012) and are increasingly being used or considered for cetaceans (e.g., Williamson et al., 2016; Lennert-Cody et al., 2018). There remain challenges regarding data processing, which is time consuming without automatic image analysis, and in data analysis for unbiased abundance estimation, but there are recent developments. In particular, work on analytical methods to use data from two camera systems on one aircraft is promising (Stevenson et al., 2018; Borchers et al., 2020).

Advances in technology have also led to the development of various unoccupied aircraft systems (UAS) [also known as unmanned aerial vehicles – UAV] to collect photographic data on marine mammals with the aim of assessing their effectiveness for estimating abundance. Results show promise for pinnipeds (Goebel et al., 2015; Moreland et al., 2015), sirenians (Jones et al., 2006; Hodgson et al., 2013) and cetaceans (Aniceto et al., 2018); see also reviews in Marine Mammal Commission (2016) and Johnston (2019). UAS may also be useful to confirm species identity and estimate group size of cetaceans on ship surveys.

Unoccupied aircraft systems eliminate the human risk of flying but there are currently considerable challenges to use them to conduct surveys to estimate abundance. In a comparison of surveys of Arctic cetaceans conducted using UAS and piloted aircraft, Ferguson et al. (2018) found that estimates of density from UAS were smaller, less precise and an order of magnitude more expensive to obtain than estimates from conventional visual survey methods. UAS powered by jet fuel can survey for several hours (e.g., Angliss et al., 2018; Ferguson et al., 2018) but the endurance of more widely available battery powered UAS is usually too short (e.g., 20 min; Aniceto et al., 2018) for most surveys.

Ships typically survey at speeds that are an order of magnitude slower than aircraft, and thus cover transects much more slowly than aircraft per unit time. Even though the effective strip width on aerial surveys is typically narrower than on ship surveys, the effective search area is still much smaller per unit time on ship surveys than on aerial surveys. For example, on the SCANS-II survey of European Atlantic waters, the effective search areas per unit time for harbor porpoises, white-beaked dolphins (*Lagenorhynchus albirostris*) and bottlenose dolphins were 5–9 times larger for an aerial survey than a ship survey (Hammond et al., 2013). Aerial surveys are thus able to "cover a lot more ground."

However, because of the difference in survey speed, the probability of detecting animals is much smaller on aerial surveys than on ship surveys, although aircraft survey speed does mean that responsive movement does not need to be considered. On aerial surveys, animals are available to be seen for a shorter period of time and it is easier to miss them by chance. In other words, both availability bias and perception bias are larger on aerial surveys than on ship surveys. This also means that sample sizes of sightings may be smaller on aerial surveys, which may make it more challenging to estimate detection probability.

There are also differences between ship and aerial surveys regarding ease of species identification and determining group size. On an aerial survey, groups of animals are seen for only a very short period of time (a few seconds) but on a ship there is much more time available to identify the species composition and size of groups, especially if conducted in closing mode.

One advantage of ship surveys is that they allow different data types to be collected. These may include photo-id data (which can also be collected on aerial surveys for some species, e.g., right and bowhead whales), biopsy sampling for genetic studies, environmental data for studies of habitat use and potentially for model-based abundance estimation, and acoustic data for deep-diving cetacean species such as sperm or beaked whales. However, photo-id and biopsy sampling take time away from surveying and it is important to consider whether the objectives of the study would be compromised by collection of such additional data. Ships can also accommodate more people, which may be useful in the context of training and outreach activities.

In terms of cost, chartering and using ships and aircraft is expensive and the cost varies depending on the equipment. The cost of larger ships is typically charged for a block of time on a per-day basis. Time at sea not surveying due to poor weather is part of this cost, which means that the expected proportion of time thus lost should be factored into survey planning. For aerial surveys, equipment charter costs are typically primarily dependent on flying hours, so time not surveying may not incur much financial penalty. However, in remote areas, aircraft need to be chartered for blocks of time in the same way as a ship and unfavorable weather also needs to be considered in planning.

Data Analysis

Table 3 does not consider analysis of the data to estimate abundance. To estimate abundance from pup or all-age counts of pinnipeds, or land-based counts of migrating whales, analysis can follow methods described in studies referred to in the associated sections of this paper.

For line transect (distance) sampling, the software DISTANCE ([12]Thomas et al., 2010) is a commonly used analysis engine for Windows platforms. Several packages within the R analysis software (R Core Team, 2020) are also available for line transect analysis (Miller et al., 2019; Laake et al., 2020). Mark-recapture models to estimate population size are available in the widely used software MARK (see text footnote 8; Cooch and White, 2019) for Windows platforms. Efficient implementation of analyses in MARK can also be achieved through the R package RMark (Laake, 2013). DISTANCE and MARK can be used to implement standard methods and also more complex analyses that relax some of the assumptions of the more basic methods, as described above. The R package multimark (McClintock, 2015) fits mark-recapture models to data from more than one mark

---

[12] http://distancesampling.org/

type, e.g., left- and right- side photo-id images, or photo-id and genetic marks.

However, a quick scan of the studies referred to above indicates that some use statistically complex methods that cannot be implemented in standard software and will be beyond the expertise of most biologists. If such methods are pursued, experienced analysts will be required.

CONCLUDING REMARKS

Motivated by the need to estimate the abundance of marine mammal populations to inform assessments for conservation and management, this paper provides background to abundance estimation and reviews the various methods available for pinnipeds, cetaceans and sirenians.

For pinnipeds, the primary methods are extrapolating counts of pups to the all-age population using life history information, or extrapolating counts of non-pups hauled out on land or ice to the whole population by accounting for the proportion of animals in the water, ideally using telemetry data. For species that aggregate on land, counts can typically be made on site, but aerial surveys are needed to count widely distributed ice-associated seals.

The abundance of cetaceans and sirenians is primarily estimated from transect surveys. Ships are needed to survey wide areas offshore, but aerial surveys are commonly used in waters adjacent to coasts, and small boats may be desirable or required in coastal waters and rivers. The assumption of line transect sampling that all animals are detected on the transect line is rarely if ever met for marine mammals but, on ship or aerial surveys, double-observer data can be collected and used to correct, or partially correct, for animals missed on the transect line. Data on diving behavior can also be used to correct for animals that are unavailable because they are underwater. For some deep-diving cetaceans, and potentially other species, passive acoustic data collected from towed, static or drifting hydrophones can be used to estimate abundance. The abundance of baleen whale species that migrate close to land or ice can be estimated from land-based counts.

For species of cetaceans or pinnipeds in which individuals are naturally marked or can be physically tagged, mark-recapture analysis of photo-identification or tag data can be used to estimate the number of animals using the study area. These studies require multiple sampling occasions and typically take longer than line transect surveys. Clearly, they are not appropriate for species that do not possess useable natural markings or cannot be tagged.

Whilst these are the most commonly used methods, there is no simple formula to follow to decide on the most suitable way to estimate marine mammal abundance. When choosing an appropriate method and how to implement it, key issues include: defining the population to be estimated, consideration of candidate methods based on strengths and weaknesses in relation to a range of logistical and practical issues, being aware of the resources required to collect and analyze the data, and understanding the assumptions made, the violation of which may lead to bias.

Estimating the abundance of marine mammal populations can be logistically, financially, and technically challenging but the information is essential to inform assessments of conservation status. The challenge is ongoing because population size changes and assessments need to be updated periodically. Regarding bycatch assessment, although methods are available to assess bycatch without estimates of abundance, they depend on the availability of fairly accurate bycatch mortality estimates and are sensitive to uncertainties (Punt et al., 2021). Information on abundance is thus a key requirement of bycatch assessments (Wade et al., In review, Frontiers in Marine Science) (see text footnote 3), which also need to be updated regularly.

In some developed countries, there is legislation requiring regular assessment of conservation status, including assessment of the impact of human activities, such as fisheries bycatch. For example, the United States MMPA requires stock assessment reports to be reviewed at least every 3 years, and annually for stocks for which human-caused mortality exceeds PBR (Wade, 1998) or which are estimated to be declining. In Europe, the reporting cycle for assessments under the EU Habitats Directive and Marine Strategy Framework Directive is 6 years. The legislative imperative and the consequent availability of funding, means that information on marine mammal abundance in these countries tends to be quite good, with reasonable coverage spatially and temporally.

This is not the case in most developing countries, where estimates of marine mammal abundance are typically generated at a small scale by individuals who are often linked to Non-Governmental Organisations, as evidenced by some of the example studies referred to in this paper. A challenge for the future is thus to facilitate an expansion of resources in these countries to enable estimates of abundance of more marine mammal populations to become available to inform conservation assessments. This paper aims to help expand one resource – the expertise base of researchers with the skills to conduct studies to estimate marine mammal abundance.

AUTHOR CONTRIBUTIONS

PH, TF, and AP developed the initial manuscript idea. PH led the writing of the manuscript. All authors contributed to the conceptual framework, outline of the manuscript, writing and editing, and gave final approval for publication.

FUNDING

Support for this project was provided by the Lenfest Ocean Program.

ACKNOWLEDGMENTS

We thank Jay Barlow (NOAA SWFSC) and two reviewers for their comments on an earlier version of the manuscript. The views expressed are those of the authors and do not necessarily reflect the views of the Lenfest Ocean Program or the Pew Charitable Trusts.

REFERENCES

Adame, K., Pardo, M. A., Beier, E., and Elorriaga-Pardo, F. (2017). Detectability and categorization of California sea lions using an unmanned aerial vehicle. *Mar. Mamm. Sci*. 33, 913–925. doi: 10.1111/mms.12403

Amstrup, C. S., McDonald, T. L., and Manly, B. F. J. (2005). *Handbook of Capture-Recapture Analysis*. Princeton, NJ: Princeton University Press.

Angliss, R. P., Ferguson, M. C., Hall, P., Helker, V., Kennedy, A., and Sformo, T. (2018). Comparing manned to unmanned aerial surveys for cetacean monitoring in the Arctic: methods and operational results. *J. Unmanned Veh. Syst*. 6, 109–127. doi: 10.1139/juvs-2018-0001

Aniceto, A. S., Biuw, M., Lindstrøm, U., Solbø, S. A., Broms, F., and Carroll, J. (2018). Monitoring marine mammals using unmanned aerial vehicles: quantifying detection certainty. *Ecosphere* 9:e02122. doi: 10.1002/ecs2.2122

Arso Civil, M., Quick, N., Cheney, B., Islas-Villanueva, V., Graves, J. A., Janik, V., et al. (2019a). Variations in age- and sex-specific survival rates could explain population trend in a discrete marine mammal population. *Ecol. Evol.* 9, 533–544. doi: 10.1002/ece3.4772

Arso Civil, M., Quick, N. J., Cheney, B., Pirotta, E., Thompson, P. M., and Hammond, P. S. (2019b). Changing distribution of the east coast of Scotland bottlenose dolphin population and the challenges of area-based management. *Aquat. Conserv. Mar. Freshw. Ecosyst*. 29, 178–196. doi: 10.1002/aqc.3102

Avila, I. C., Kaschner, K., and Dormann, C. F. (2018). Current global risks to marine mammals: taking stock of the threats. *Biol. Conserv*. 221, 44–58. doi: 10.1016/j.biocon.2018.02.021

Baker, J. D., Harting, A. L., Johanos, T. C., and Littnan, C. L. (2016). Estimating Hawaiian monk seal range-wide abundance and associated uncertainty. *Endang. Species Res.* 31, 317–324. doi: 10.3354/esr00782

Balmer, B. C., Wells, R. S., Nowacek, S. M., Nowacek, D. P., Schwacke, L. H., McLellan, W. A., et al. (2008). Seasonal abundance and distribution patterns of common bottlenose dolphins (*Tursiops truncatus*) near St. Joseph Bay, Florida, USA. *J. Cetacean Res. Manage.* 10, 157–167.

Barlow, J. (2015). Inferring trackline detection probabilities, g(0), for cetaceans from apparent densities in different survey conditions. *Mar. Mamm. Sci.* 31, 923–943. doi: 10.1111/mms.12205

Barlow, J., Calambokidis, J., Falcone, E. A., Baker, C. S., Burdin, A. M., Clapham, P. J., et al. (2011). Humpback whale abundance in the North Pacific estimated by photographic capture−recapture with bias correction from simulation studies. *Mar. Mamm. Sci*. 27, 793–818. doi: 10.1111/j.1748-7692.2010.00444.x

Barlow, J., and Forney, K. A. (2007). Abundance and population density of cetaceans in the California current ecosystem. *Fish. Bull. U.S.* 105, 509–526.

Barlow, J., Fregosi, S., Thomas, L., Harris, D., and Griffiths, E. T. (2021a). Acoustic detection range and population density of Cuvier's beaked whales estimated from near-surface hydrophones. *J. Acoust. Soc. Am*. 149, 111–125. doi: 10.1121/10.0002881

Barlow, J., Trickey, J. S., Schorr, G. S., Rankin, S., and Moore, J. E. (2021b). Recommended snapshot length for acoustic point-transect surveys of intermittently available Cuvier's beaked whales. *J. Acoust. Soc. Am*. 149, 3830–3840. doi: 10.1121/10.0005108

Barlow, J., Moore, J. E., McCullough, J. L. K., and Griffiths, E. T. (2021c). Acoustic-based estimates of Cuvier's beaked whale (*Ziphius cavirostris*) density and abundance along the US West Coast from drifting hydrophone recorders. *Mar. Mamm. Sci.*

Barlow, J., and Taylor, B. L. (2005). Estimates of sperm whale abundance in the northeastern temperate Pacific from a combine acoustic and visual survey. *Mar. Mamm. Sci*. 21, 429–445. doi: 10.1111/j.1748-7692.2005.tb01242.x

Becker, E., Forney, K. A., Miller, D. L., Fiedler, P. C., Barlow, J., and Moore, J. E. (2020). *Habitat-based Density Estimates for Cetaceans in the California Current Ecosystem based on 1991-2018 Survey Data. NOAA Technical Memorandum NMFS-SWFSC-638*. Available online at: https://swfsc-publications.fisheries.noaa.gov/publications/CR/2020/2020Becker1.pdf (accessed September 10, 2021).

Begon, M., Harper, J. L., and Townsend, C. R. (1996). *Ecology: Individuals, Populations and Communities*, 3rd Edn. Oxford: Blackwell Science.

Bengtson, J. L., Hiruki-Raring, L. M., Simpkins, M. A., and Boveng, P. L. (2005). Ringed and bearded seal densities in the eastern Chukchi Sea, 1999–2000. *Pol. Biol*. 28, 833–845.

Bigg, M. (1982). An assessment of killer whale (*Orcinus orca*) stocks off Vancouver Island, British Columbia. *Rep. Int. Whal. Commn.* 32, 655–666.

Bogucki, R., Cygan, M., Khan, C. B., Klimek, M., Milczek, J. K., and Mucha, M. (2019). Applying deep learning to right whale photo identification. *Conserv. Biol.* 33, 676–684. doi: 10.1111/cobi.13226

Borchers, D. L., Nightingale, P., Stevenson, B. C., and Fewster, R. M. (2020). A latent capture history model for digital aerial surveys. *Biometrics* 2020, 1–12. doi: 10.1111/biom.13403

Borchers, D. L., Pike, D. G., Gunnlaugsson, T., and Víkingsson, G. A. (2009). Minke whale abundance estimation from the NASS 1987 and 2001 aerial cue–counting surveys taking appropriate account of distance estimation errors. *NAMMCO Sci. Publ.* 7, 95–110. doi: 10.7557/3.2708

Bortolotto, G. A., Danilewicz, D., Andriolo, A., Secchi, E. R., and Zerbini, A. N. (2016). Whale, whale, everywhere: increasing abundance of western South Atlantic humpback whales (*Megaptera novaeangliae*) in their wintering grounds. *PLoS One* 11:e0164596. doi: 10.1371/journal.pone.0164596

Bouchet, P. J., Miller, D. L., Roberts, J. J., Mannocci, L., Harris, C. M., and Thomas, L. (2020). dsmextra: extrapolation assessment tools for density surface models. *Methods Ecol. Evol.* 11, 1464–1469. doi: 10.1111/2041-210X.13469

Boyd, C., Hobbs, R. C., Punt, A. E., Shelden, K. E. W., Sims, C. L., and Wade, P. R. (2019). Bayesian estimation of group sizes for a coastal cetacean using aerial survey data. *Mar. Mamm. Sci.* 35, 1322–1346. doi: 10.1111/mms.12592

Boyd, C., and Punt, A. E. (2021). Shifting trends: detecting changes in cetacean population dynamics in shifting habitat. *PLoS One* 16:e0251522. doi: 10.1371/journal.pone.0251522

Boyd, I. L. (1993). Pup production and distribution of breeding Antarctic fur seals (*Arctocephalus gazella*) at South Georgia. *Antarct. Sci*. 5, 17–24. doi: 10.1017/S0954102093000045

Branch, T. A. (2007). Abundance of Antarctic blue whales south of 60°S from three complete circumpolar sets of surveys. *J. Cetacean Res. Manage.* 9, 253–262.

Branch, T. A. (2011). Humpback abundance south of 60°S from three complete circumpolar sets of surveys. *J. Cetacean Res. Manage.* 3, 53–69.

Branch, T. A., and Butterworth, D. S. (2001). Southern hemisphere minke whales: standardised abundance estimates from the 1978/79 to 1997/98 IDCR-SOWER surveys. *J. Cetacean Res. Manage.* 3, 143–174.

Brasseur, S. M. J. M., Reijnders, P. J. H., Cremer, J., Meesters, E., Kirkwood, R., Jensen, L. F., et al. (2018). Echoes from the past: regional variations in recovery within a harbour seal population. *PLoS One* 13:e0189674. doi: 10.1371/journal.pone.0189674

Braulik, G., Bhatti, Z. I., Ehsan, T., Hussain, B., Khan, A. R., Khan, A., et al. (2012). Robust abundance estimate for endangered river dolphin subspecies in South Asia. *Endang. Spec. Res.* 17, 201–215. doi: 10.3354/esr00425

Bravington, M. V., Skaug, H. J., and Anderson, E. C. (2016). Close-kin mark-recapture. *Statist. Sci.* 31, 259–274. doi: 10.1214/16-STS552

Brediñana-Romano, L., Viddi, F. A., Torres-Florez, J. P., Ruiz, J., Nery, M. F., Haro, D., et al. (2014). At-sea abundance and spatial distribution of South American sea lion (*Otaria byronia*) in Chilean Northern Patagonia: how many are there? *Mamm. Biol*. 79, 384–392. doi: 10.1016/j.mambio.2014.07.002

Brown, S. G. (1978). "Whale marking techniques," in *Animal Marking*, ed. B. Stonehouse (London: Palgrave), 71–80. doi: 10.1007/978-1-349-03711-7_8

Brusa, J. L., Rotella, J. J., Garrott, R. A., Paterson, J. T., and Link, W. A. (2020). Variation of annual apparent survival and detection rates with age, year and individual identity in male Weddell seals (*Leptonychotes weddellii*) from long−term mark−recapture data. *Popul. Ecol*. 62, 134–150. doi: 10.1002/1438-390X.12036

Buckland, S. T., Anderson, D. R., Burnham, K. P., Laake, J. L., Borchers, D. L., and Thomas, L. (2001). *Introduction to Distance Sampling: Estimating Abundance of Biological Populations*. Oxford: Oxford University Press.

Buckland, S. T., Burt, M. L., Rexstad, E. A., Mellor, M., Williams, A. E., and Woodward, R. (2012). Aerial surveys of seabirds: the advent of digital methods. *J. Appl. Ecol.* 49, 960–967. doi: 10.2307/23259202

Buckland, S. T., Rexstad, E. A., Marques, T. S., and Oedekoven, C. S. (2015). *Distance Sampling: Methods and Applications*. New York, NY: Springer International Publishing.

Buckland, S. T., and York, A. E. (2018). "Abundance estimation," in *Encyclopedia of Marine Mammals*, 3rd Edn, eds B. Würsig, J. G. M. Thewissen, and K. Kovacs (San Diego, CA: Academic Press), 1–6.

Burt, M. L., Borchers, D. L., Jenkins, K. J., and Marques, T. A. (2014). Using mark-recapture distance sampling methods on line transect surveys. *Methods Ecol. Evol.* 5, 1180–1191. doi: 10.1111/j.1365-2664.2012.02150.x

Butterworth, D. S., Punt, A. E., Oosthuizen, W. H., and Wickens, P. A. (1995). The effects of future consumption by the Cape fur seal on catches and catch rates of the Cape hakes. 3. Modelling the dynamics of the Cape fur seal population. *S. Afr. J. Mar. Sci.* 16, 161–183. doi: 10.2989/025776195784156511

Calambokidis, J., and Barlow, J. (2004). Abundance of blue and humpback whales in the eastern North Pacific estimated by capture-recapture and line-transect methods. *Mar. Mamm. Sci*. 20, 63–85. doi: 10.1111/j.1748-7692.2004.tb01141.x

Campbell, G. S., Thomas, L., Whitaker, K., Douglas, A. B., Calambokidis, J., and Hildebrand, J. A. (2015). Inter-annual and seasonal trends in cetacean distribution, density and abundance off southern California. *Deep Sea Res. II* 112, 143–157. doi: 10.1016/j.dsr2.2014.10.008

Cañadas, A., Desportes, G., and Borchers, D. (2004). The estimation of the detection function and g(0) for short-beaked common dolphins (*Delphinus delphis*), using double platform data collected during the NASS-95 Faroese survey. *J. Cetacean Res. Manage.* 6, 191–198.

Cañadas, A., and Hammond, P. S. (2006). Model-based abundance estimates for bottlenose dolphins off southern Spain: implications for conservation and management. *J. Cetacean Res. Manage.* 8, 13–27.

Cañadas, A., and Hammond, P. S. (2008). Abundance and habitat preferences of the short-beaked common dolphin (*Delphinus delphis*) in the South-western Mediterranean: implications for conservation. *Endang. Species Res.* 4, 309–331. doi: 10.3354/esr00073

Cardinale, B., Primack, R., and Murdoch, J. (2019). *Conservation Biology*. Oxford: Oxford University Press, 584.

Chan, S. C. Y., and Karczmarski, L. (2017). Indo-Pacific humpback dolphins (*Sousa chinensis*) in Hong Kong: modelling demographic parameters with mark-recapture techniques. *PLoS One* 12:e0174029. doi: 10.1371/journal.pone.0174029

Cheeseman, T., Southerland, K., Park, J., Olio, M., Flynn, K., Calambokidis, J., et al. (in press). "Advanced image recognition: a fully automated, high accuracy photo-identification matching system for humpback whales," in *Individual Identification and Photographic Techniques in Mammalian Ecological and Behavioural Research: Methods, Applications and Concepts. Mamm. Biol.,* eds L. Karczmarski, S. C. Y. Chan, D. I. ubenstein, S. Y. S. Chui, and E. Z. Cameron.

Cole, L. C. (1957). Sketches of general and comparative demography. *Cold Springs Harb. Symp. Quant. Biol.* 22, 1–15. doi: 10.1101/sqb.1957.022.01.004

Constantine, R., Jackson, J. A., Steel, D., Baker, C. S., Brooks, L., Burns, D., et al. (2012). Abundance of humpback whales in Oceania using photo—identification and microsatellite genotyping. *Mar. Ecol. Prog. Ser*. 453, 249–261. doi: 10.3354/meps09613

Cooch, E. G., and White, G. C. (2019). *Program MARK – A Gentle Introduction*, 19th Edn. Available online at: http://www.phidot.org/software/mark/docs/book/ (accessed September 10, 2021).

Cooke, J. G., Weller, D. W., Bradford, A. L., Burdin, A. M., and Brownell, R. L. Jr. (2005). "Estimates and projections of the western gray whale population using an individually-based population model," in *Paper Presented at the Scientific Committee of the International Whaling Commission SC/57/BRG22*, Ulsan.

Cordes, L. S., and Thompson, P. M. (2015). Mark-resight estimates of seasonal variation in harbor seal abundance and site fidelity. *Popul. Ecol.* 57, 467–472. doi: 10.1007/s10144-015-0496-z

Crespo, E. A., Pedraza, S. N., Grandi, M. F., Dans, S. L., and Garaffo, G. V. (2010). Abundance and distribution of endangered franciscana dolphins in Argentine waters and conservation implication. *Mar. Mamm. Sci.* 26, 17–35. doi: 10.1111/j.1748-7692.2009.00313.x

Curtis, K. A., Falcone, E. A., Schorr, G. S., Moore, J. E., Moretti, D. J., Barlow, J., et al. (2021). Abundance, survival, and annual rate of change of Cuvier's beaked whales (*Ziphius cavirostris*) on a Navy sonar range. *Mar. Mamm. Sci*. 37, 399-419. doi: 10.1111/mms.12747

Danilewicz, D., Moreno, I. B., Ott, P. H., Tavares, M., Azevedo, A. M., Secchi, E., et al. (2010). Abundance estimation for a threatened population of franciscana dolphins in southern Brazil: uncertainties and conservation considerations. *J. Mar. Biol. Assoc. U.K.* 90, 1649–1657. doi: 10.1017/S0025315409991482

Dawson, S., Wade, P. R., Slooten, E., and Barlow, J. (2008). Design and field methods for cetacean surveys in coastal and riverine habitats. *Mamm. Rev*. 38, 19–49. doi: 10.1111/j.1365-2907.2008.00119.x

Doi, T. (1974). "Further development of whale sighting theory. Chapter 16," in *The Whale Problem*, ed. W. E. Schevill (Cambridge, MA: Harvard University Press), 359–368.

Doniol-Valcroze, T., Gosselin, J.-F., Pike, D. G., Lawson, J. W., Asselin, N. C., Hedges, K., et al. (2020). Narwhal abundance in the eastern Canadian High Arctic in 2013. *NAMMCO Sci. Publ.* 11, 1–26. doi: 10.7557/3.5100

Durban, J., Ellifrit, D., Dahlheim, M., Waite, J., Matkin, C., Barrett-Lennard, L., et al. (2010). Photographic mark-recapture analysis of clustered mammal-eating killer whales around the Aleutian Islands and Gulf of Alaska. *Mar. Biol.* 150, 1033–1045. doi: 10.1007/s00227-010-1432-6

European Union (2017). *Regulation (EU) 2017/1004 of the European Parliament and of the Council of 17 May 2017 on the Establishment of a Union Framework for the Collection, Management and Use of Data in the Fisheries Sector and Support for Scientific Advice Regarding the Common Fisheries Policy and Repealing Council Regulation (EC) No 199/2008*. Available online at: https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A32017R1004 (accessed September 10, 2021).

Felix, F., Castro, C., Laake, J. L., Haase, B., and Schiedat, M. (2011). Abundance and survival estimates of the southeastern Pacific humpback whale stock from 1991–2006 photo-identification surveys in Ecuador. *J. Cetacean Res. Manage.* 3, 301–307.

Ferguson, M. C., Angliss, R. P., Kennedy, A., Lynch, B., Willoughby, A., Helker, V., et al. (2018). Performance of manned and unmanned aerial surveys to collect visual data and imagery for estimating Arctic cetacean density and associated uncertainty. *J. Unmanned Veh. Syst.* 6, 128–154. doi: 10.1139/juvs-2018-0002

Findlay, K. P., Cockcroft, V. G., and Guissamulo, A. T. (2011). Dugong abundance and distribution in the Bazaruto Archipelago, Mozambique. *Afr. J. Mar. Sci*. 33, 441–452. doi: 10.2989/1814232X.2011.637347

Forcada, J., and Aguilar, A. (2000). Use of photographic identification in capture-recapture studies of Mediterranean monk seals. *Mar. Mamm. Sci.* 16, 767–793. doi: 10.1111/j.1748-7692.2000.tb00971.x

Forcada, J., and Robinson, S. L. (2006). Population abundance, structure and turnover estimates for leopard seals during winter dispersal combining tagging and photo-identification data. *Pol. Biol.* 29, 1052–1062. doi: 10.1007/s00300-006-0149-y

Forney, K. A., Moore, J. E., Barlow, J., Carretta, J. V., and Benson, S. R. (2021). A multidecadal Bayesian trend analysis of harbor porpoise (*Phocoena phocoena*) populations off California relative to past fishery bycatch. *Mar. Mamm. Sci*. 37, 546–560. doi: 10.1111/mms.12764

Friday, N. A., Zerbini, A. N., Waite, J. M., Moore, S. E., and Clapham, P. J. (2013). Cetacean distribution and abundance in relation to oceanographic domains on the eastern Bering Sea shelf, June and July of 2002, 2008, and 2010. *Deep Sea Res. II* 94, 244–256. doi: 10.1016/j.dsr2.2013.03.011

Frie, A. K., Stenson, G. B., and Haug, T. (2012). Long-term trends in reproductive and demographic parameters of female Northwest Atlantic hooded seals (*Cystophora cristata*): population responses to ecosystem change? *Can. J. Zool*. 90, 376–392. doi: 10.1139/z11-140

Galatius, A., Teilmann, J., Dähne, M., Ahola, M., Westphal, L., Kyhn, L. A., et al. (2020). Extrapolating cetacean densities to quantitatively assess human impacts on populations in the high seas. *Wildl. Biol.* 2020:wlb.00711. doi: 10.2981/wlb.00711

Galletti Vernazzani, B., Jackson, J. A., Cabrera, E., Carlson, C. A., and Brownell, R. L. Jr. (2017). Estimates of abundance and trend of Chilean blue whales off Isla de Chiloé, Chile. *PLoS One* 12:e0168646. doi: 10.1371/journal.pone.0168646

García-Aguilar, M., and Morales-Bojórquez, E. (2005). Estimating the haul-out population size of a colony of northern elephant seals *Mirounga angustirostris* in Mexico, based on mark-recapture data. *Mar. Ecol. Prog. Ser*. 297, 297–302. doi: 10.3354/meps297297

George, J. C., Zeh, J., Suydam, R., and Clark, C. (2004). Abundance and population trend (1978-2001) of western Arctic bowhead whales surveyed near Barrow, Alaska. *Mar. Mamm. Sci*. 20, 755–773. doi: 10.1111/j.1748-7692.2004.tb01191.x

Gerondeau, M., Barbraud, C., Ridoux, V., and Vincent, C. (2007). Abundance estimate and seasonal patterns of grey seal (*Halichoerus grypus*) occurrence in Brittany, France, as assessed by photo-identification and capture-mark-recapture. *J. Mar. Biol. Assoc.* U.K. 87, 365–372. doi: 10.1017/S0025315407054586

Gerrodette, T., Perryman, W. L., and Oedekoven, C. S. (2019). Accuracy and precision of dolphin group size estimates. *Mar. Mamm. Sci.* 35, 22–39. doi: 10.1111/mms.12506

Gilles, A., Gunnlaugsson, T., Mikkelsen, B., Pike, D. G., and Víkingsson, G. (2020). Summer abundance of harbour porpoises (*Phocoena phocoena*) in the coastal waters of Iceland and the Faroe Islands. *NAMMCO Sci. Publ.* 11, 1–9. doi: 10.7557/3.4939

Gilles, A., Viquerat, S., Becker, E. A., Forney, K. A., Geelhoed, S. C. V., Haelters, J., et al. (2016). Seasonal habitat-based density models for a marine top predator, the harbor porpoise, in a dynamic environment. *Ecosphere* 7:e01367. doi: 10.1002/ecs2.1367

Givens, G. H., Edmondson, S. L., George, J. C., Suydam, R., Charif, R. A., Rahamand, A., et al. (2016). Horvitz–Thompson whale abundance estimation adjusting for uncertain recapture, temporal availability variation, and intermittent effort. *Environmetrics* 27, 134–146. doi: 10.1002/env.2379

Goebel, M. E., Perryman, W. L., Hinke, J. T., Krause, D. J., Hann, N. A., Gardner, S., et al. (2015). A small unmanned aerial system for estimating abundance and size of Antarctic predators. *Pol. Biol.* 38, 619–630. doi: 10.1007/s00300-014-1625-4

Gomez-Salazar, C., Trujillo, F., Portocarrero-Aya, M., and Whitehead, H. (2012). Population, density estimates, and conservation of river dolphins (*Inia* and *Sotalia*) in the Amazon and Orinoco river basins. *Mar. Mamm. Sci.* 28, 124–153. doi: 10.1111/j.1748-7692.2011.00468.x

Gordon, J. (2001). Measuring the range to animals at sea from boats using photographic and video images. *J. Appl. Ecol.* 38, 879–887. doi: 10.1046/j.1365-2664.2001.00615.x

Gosselin, J.-F., Hammill, M. O., Mosnier, A., and Lesage, V. (2017). *Abundance Index of St. Lawrence Estuary Beluga, Delphinapterus leucas, from Aerial Visual Surveys Flown in August 2014 and an Update on Reported Deaths. DFO Canadian Science Advisory Secretariat Research Document 2017/019.* Available online at: http://www.dfo-mpo.gc.ca/csas-sccs/

Greig, D. J., Gulland, F. M. D., Harvey, J. T., Lonergan, M., and Hall, A. J. (2019). Harbor seal pup dispersal and individual morphology, hematology, and contaminant factors affecting survival. *Mar. Mamm. Sci.* 35, 187–209. doi: 10.1111/mms.12541

Guinet, C., Jouventin, P., and Georges, J.-Y. (1994). Long term population changes of fur seals *Arctocephalus gazela* and *Arctocephalus tropicalis* on subantarctic (Crozet) and subtropical (St. Paul and Amsterdam) islands and their possible relationship to El Nifio Southern Oscillation. *Antarct. Sci.* 6, 473–478. doi: 10.1017/S0954102094000714

Guzman, H. M., and Condit, R. (2017). Abundance of manatees in Panama estimated from side-scan sonar. *Wildl. Soc. Bull.* 41, 556–565. doi: 10.1002/wsb.793

Hagihara, R., Cleguer, C., Preston, S., Sobtzick, S., Hamann, M., Shimada, T., et al. (2016). *Improving the Estimates of Abundance of Dugongs and Large Immature and Adult-Sized Green Turtles in Western and Central Torres Strait. Report to the National Environmental Science Programme*. Cairns, QLD: Reef and Rainforest Research Centre Limited, 53.

Hammond, P. S. (1986). Estimating the size of naturally marked whale populations using capture-recapture techniques. *Rep. Int. Whal. Commn. (Special Issue 8),* 253–282.

Hammond, P. S. (2010). "Estimating the abundance of marine mammals," in *Marine Mammal Ecology and Conservation: A Handbook of Techniques*, eds I. L. Boyd, W. D. Bowen, and S. Iverson (New York, NY: Oxford University Press), 42–67.

Hammond, P. S. (2018). "Mark-recapture," in *Encyclopedia of Marine Mammals*, 3rd Edn, eds B. Würsig, J. G. M. Thewissen, and K. Kovacs (San Diego, CA: Academic Press), 580–584.

Hammond, P. S., Berggren, P., Benke, H., Borchers, D. L., Collet, A., Heide-Jørgensen, M. P., et al. (2002). Abundance of harbour porpoises and other cetaceans in the North Sea and adjacent waters. *J. Appl. Ecol.* 39, 361–376. doi: 10.1046/j.1365-2664.2002.00713.x

Hammond, P. S., Macleod, K., Berggren, P., Borchers, D. L., Burt, M. L., Cañadas, A., et al. (2013). Cetacean abundance and distribution in European Atlantic shelf waters to inform conservation and management. *Biol. Conserv.* 164, 107–122. doi: 10.1016/j.biocon.2013.04.010

Hammond, P. S., Mizroch, S. A., and Donovan, G. P. (1990). *Individual Recognition of Cetaceans: Use of Photo-Identification and Other Techniques to Estimate Population Parameters. Rep. Int. Whal. Commn (Special Issue 12).* Cambridge: International Whaling Commission, 440.

Hamner, R. M., Wade, P. R., Oremus, M., Stanley, M., Brown, P., Constantine, R., et al. (2014). Critically low abundance and limits to human-related mortality for the Maui's dolphin. *Endang. Species Res.* 26, 87–92. doi: 10.3354/esr00621

Hansen, R. G., Boye, T. K., Larsen, R. S., Nielsen, N. H., Tervo, O., Nielsen, R. D., et al. (2018). Abundance of whales in West and East Greenland in summer 2015. *NAMMCO Sci. Publ.* 11, 1–17. doi: 10.7557/3.4689

Hansen, R. G., Pike, D., Thorgilsson, B., Gunnlaugsson, T., and Lawson, J. (2020). The geometer: a new device for recording angles in visual surveys. *NAMMCO Sci. Publ.* 11, 1–9. doi: 10.7557/3.5585

Hauksson, E. (2007). Abundance of grey seals in Icelandic waters, based on trends of pup-counts from aerial surveys. *NAMMCO Sci. Publ.* 6, 85–97. doi: 10.7557/3.2725

Heide-Jørgensen, M. P., and Laidre, K. L. (2015). Surfacing time, availability bias and abundance of humpback whales in West Greenland. *J. Cetacean Res. Manage.* 15, 1–8.

Hiby, A. R., and Hammond, P. S. (1989). Survey techniques for estimating abundance of cetaceans. *Rep. Int. Whal. Commn. (Special Issue 11),* 47–80.

Hiby, L. (1999). "The objective identification of duplicate sightings in aerial survey for porpoise," in *Marine Mammal Survey and Assessment Methods*, eds G. W. Garner, S. C. Armstrup, J. L. Laake, B. F. J. Manly, L. L. McDonald, and D. G. Robertson (Rotterdam: Balkema), 179–189.

Hiby, L., and Lovell, P. (1990). Computer-aided matching of natural markings: a prototype system for grey seals. *Rep. Int. Whal. Commn. (Special Issue 12),* 57–61.

Hodgson, A., Kelly, N., and Peel, D. (2013). Unmanned aerial vehicles (UAVs) for surveying marine fauna: a dugong case study. *PLoS One* 8:e79556. doi: 10.1371/journal.pone.0079556

Hostetler, J. A., Edwards, H. H., Martin, J., and Schueller, P. (2018). *Updated Statewide Abundance Estimates for the Florida Manatee. Florida Fish and Wildlife Conservation Commission, Fish and Wildlife Research Institute Technical Report No. 23.* Available online at: https://f50006a.eos-intl.net/ELIBSQL12_F50006A_Documents/TR23-18Hostetler-USAEF.pdf (accessed September 10, 2021).

ICES (2014). *General Advice on OSPAR Request on Implementation of MSFD for Marine Mammals. ICES Advice 2014, Book 1: 1.6.6.1.* Available online at: https://www.ices.dk/sites/pub/Publication%20Reports/Forms/DispForm.aspx?ID=28807 (accessed September 10, 2021).

IUCN (2021). *Glossary of Definitions*. Available online at: https://www.iucn.org/sites/dev/files/iucn-glossary-of-definitions_en_2021.05.pdf (accessed May 3, 2021).

IWC (2012). Requirements and guidelines for conducting surveys and analysing data within the revised management scheme. *J. Cetacean Res. Manage.* 13(Suppl.), 509–517.

IWC (2020). Scientific committee procedures for submission, review and validation of abundance estimates. *J. Cetacean Res. Manage.* 21(Suppl.), 273–726.

Jefferson, T. A., and Moore, J. E. (2020). Abundance and trends of Indo-Pacific finless porpoises (*Neophocaena phocaenoides*) in Hong Kong waters, 1996-2019. *Front. Mar. Sci.* 7:574381. doi: 10.3389/fmars.2020.574381

Jessopp, M. J., Forcada, J., Reid, K., Trathan, P. N., and Murphy, E. J. (2004). Winter dispersal of leopard seals (*Hydrurga leptonyx*): environmental factors influencing demographics and seasonal abundance. *J. Zool.* 263, 251–258. doi: 10.1017/S0952836904005102

Jewell, R., Thomas, L., Harris, C. M., Kaschner, K., Wiff, R., Hammond, P. S., et al. (2012). Global analysis of cetacean line-transect surveys: detecting trends in cetacean density. *Mar. Ecol. Prog. Ser.* 453, 227–240. doi: 10.3354/meps09636

Johnston, D. W. (2019). Unoccupied aircraft systems in marine science and conservation. *Ann. Rev. Mar. Sci.* 11, 439–463. doi: 10.1146/annurev-marine-010318-095323

Jones, G. P., Pearlstine, L. G., and Percival, H. F. (2006). An assessment of small unmanned aerial vehicles for wildlife research. *Wildl. Soc. Bull.* 34, 750–758.

Keating, J. L., Barlow, J., Griffiths, E. T., and Moore, J. E. (2018). *Passive Acoustics Survey of Cetacean abundance Levels (PASCAL-2016). Final Report*. Honolulu, HI: US Department of the Interior, Bureau of Ocean Energy Management. OCS Study BOEM 2018-025.

Koivuniemi, M., Auttila, M., Niemi, M., Levänen, R., and Kunnasranta, M. (2016). Photo-ID as a tool for studying and monitoring the endangered Saimaa ringed seal. *Endang. Species Res.* 30, 29–36. doi: 10.3354/esr00723

Koivuniemi, M., Kurkilahti, M., Niemi, M., Auttila, M., and Kunnasranta, M. (2019). A mark-recapture approach for estimating population size of the endangered ringed seal (*Phoca hispida saimensis*). *PLoS One* 14:e0214269. doi: 10.1371/journal.pone.0214269

Krebs, C. J. (1972). *Ecology: The Experimental Analysis of Distribution and Abundance. Harper International Edition*. New York, NY: Harper & Row.

Kuit, S. H., Ponnampalam, L. S., Hammond, P. S., Chong, V. C., and Then, A. Y.-H. (2021). Abundance estimates of three cetacean species in the coastal waters of Matang, Perak, Peninsular Malaysia. *Aquat. Conserv. Mar. Freshw. Ecosyst.* 2021, 1–13. doi: 10.1002/aqc.3699

Kuningas, S., Similä, T., and Hammond, P. S. (2014). Population size, survival and reproductive rates of northern Norwegian killer whales (*Orcinus orca*) in 1986-2003. *J. Mar. Biol. Assoc*. U.K. 94, 1277–1291. doi: 10.1017/S0025315413000933

Laake, J., Borchers, D., Thomas, L., Miller, D., and Bishop, J. (2020). *Mark-Recapture Distance Sampling*. Available online at: https://cran.r-project.org/web/packages/mrds/mrds.pdf (accessed September 10, 2021).

Laake, J. L. (2013). *RMark: An R Interface for Analysis of Capture-Recapture Data with MARK. AFSC Processed Rep. 2013-01*. Seattle WA 98115: Alaska Fish. Sci. Cent., NOAA, Natl. Mar. Fish. Serv., 25.

Laake, J. L., Calambokidis, J., Osmek, S. D., and Rugh, D. J. (1997). Probability of detecting harbour porpoise from aerial surveys: estimating $g(0)$. *J. Wildl. Manage.* 61, 63–75. doi: 10.2307/3802415

Laake, J. L., Punt, A. E., Hobbs, R., Ferguson, M., Rugh, D., and Breiwick, J. (2012). Gray whale southbound migration surveys 1967-2006: an integrated re-analysis. *J. Cetacean Res. Manage.* 12, 287–306.

Leaper, R., Burt, L., Gillespie, D., and Macleod, K. (2010). Comparisons of measured and estimated distances and angles from sightings surveys. *J. Cetacean Res. Manage.* 11, 229–237.

Leaper, R., and Gordon, J. (2001). Application of photogrammetric methods for locating and tracking cetacean movements at sea. *J. Cetacean Res. Manage.* 3, 131–141.

Lennert-Cody, C. E., Buckland, S. T., Gerrodette, T., Webb, A., Barlow, J., Fretwell, P. T., et al. (2018). Review of potential line-transect methodologies for estimating abundance of dolphin stocks in the eastern tropical Pacific. *J. Cetacean Res. Manage.* 19, 9–21.

Lewis, T., Gillespie, D., Lacey, C., Matthews, J., Danbolt, M., Leaper, R., et al. (2007). Sperm whale abundance estimates from acoustic surveys of the Ionian Sea and Straits of Sicily in 2003. *J. Mar. Biol. Assoc.* U.K. 87, 353–357. doi: 10.1017/S0025315407054896

Lonergan, M., Duck, C. D., Thompson, D., Moss, S., and McConnell, B. (2011). British grey seal (*Halichoerus grypus*) abundance in 2008: an assessment based on aerial counts and satellite telemetry. *ICES J. Mar. Sci.* 68, 2201–2209. doi: 10.1093/icesjms/fsr161

Lowry, M. S. (1999). Counts of California sea lions (*Zalophus californianus*) pups from aerial color photographs and from the ground: a comparison of two methods. *Mar. Mamm. Sci.* 15, 143–158. doi: 10.1111/j.1748-7692.1999.tb00786.x

Lowry, M. S., Condit, R., Hatfield, B., Allen, S. G., Berger, R., Morris, P. A., et al. (2014). Abundance, distribution, and population growth of the northern elephant seal (*Mirounga angustirostris*) in the United States from 1991 to 2010. *Aquat. Mamm.* 40, 20–31. doi: 10.1578/AM.40.1.2014.20

MacKenzie, D. L., and Clement, D. M. (2014). *Abundance and Distribution of ECSI Hector's Dolphin. New Zealand Aquatic Environment and Biodiversity Report No. 123,* 79. Available online at: https://healthyharbour.org.nz/wp-content/uploads/2019/05/MacKenzie-Clement-2014-Hectors-dolphin.pdf (accessed September 10, 2021).

Madon, B., Garrigue, C., Pradel, R., and Gimenez, O. (2013). Transience in the humpback whale population of New Caledonia and implications for abundance estimation. *Mar. Mamm. Sci.* 29, 669–678.

Madon, B., Gimenez, O., McArdle, B., Baker, C. S., and Garrigue, C. (2011). A new method for estimating animal abundance with two sources of data in capture-recapture studies. *Methods Ecol. Evol.* 2, 390–400. doi: 10.1111/j.2041-210X.2011.00091.x

Mannocci, L., Roberts, J. J., Miller, D. L., and Halpin, P. N. (2017). Extrapolating cetacean densities to quantitatively assess human impacts on populations in the high seas. *Conserv. Biol.* 31, 601–614. doi: 10.1111/cobi.12856

Marine Mammal Commission (2016). *Development and Use of UASs by the National Marine Fisheries Service for Surveying Marine Mammals*. Bethesda, MD: Marine Mammal Commission.

Marques, T. A., Thomas, L., Ward, J., DiMarzio, N., and Tyack, P. L. (2009). Estimating cetacean population density using fixed passive acoustic sensors: an example with Blainville's beaked whales. *J. Acoust. Soc. Am.* 125, 1982–1994. doi: 10.1121/1.3089590

Marsh, H., and Sinclair, D. F. (1989). Correcting for visibility bias in strip transect aerial surveys of aquatic fauna. *J. Wildl. Manage.* 53, 1017–1024. doi: 10.2307/3809604

Martin, J., Edwards, H. H., Fonnesbeck, C. J., Koslovsky, S. M., Harmak, C. W., and Dane, T. W. (2015). Combining information for monitoring at large spatial scales: first statewide abundance estimate of the Florida manatee. *Biol. Conserv.* 186, 44–51. doi: 10.1016/j.biocon.2015.02.029

Martin, M. J., Gridley, T., Roux, J.-P., and Elwen, S. H. (2020). First abundance estimates of Heaviside's (*Cephalorhynchus heavisidii*) and dusky (*Lagenorhynchus obscurus*) dolphins off Namibia using a novel visual and acoustic line transect survey. *Front. Mar. Sci.* 7:555659. doi: 10.3389/fmars.2020.555659

Martínez-Jauregui, M., Tavecchia, G., Cedenilla, M. A., Coulson, T., Fernández de Larrinoa, P., Muñoz, M., et al. (2012). Population resilience of the Mediterranean monk seal *Monachus monachus* at Cabo Blanco peninsula. *Mar. Ecol. Prog. Ser.* 461, 273–281. doi: 10.3354/meps09811

McClintock, B. T. (2015). multimark: an R package for analysis of capture-recapture data consisting of multiple "noninvasive" marks. *Ecol. Evol.* 5, 4920–4931. doi: 10.1002/ece3.1676

McClintock, B. T., Conn, P. B., Alonso, R. S., and Crooks, K. R. (2013). Integrated modelling of bilateral photo-identification data in mark-recapture analysis. *Ecology* 94, 1464–1471. doi: 10.1890/12-1613.1

McIntosh, R. R., Arthur, A. D., Dennis, T., Berris, M., Goldsworthy, S. D., Shaughnessy, P. D., et al. (2013). Survival estimates for the Australian sea lion: negative correlation of sea surface temperature with cohort survival to weaning. *Mar. Mamm. Sci.* 29, 84–108. doi: 10.1111/j.1748-7692.2011.00558.x

McIntosh, R. R., Holmberg, R., and Dann, P. (2018). Looking without landing – using remote piloted aircraft to monitor fur seal populations without disturbance. *Front. Mar. Sci.* 5:202. doi: 10.3389/fmars.2018.00202

McLaren, I. A. (1961). Methods of determining the numbers and availability of ring seals in the eastern Canadian Arctic. *Arctic* 14, 162–175. doi: 10.3389/fmars.2018.00202

Miller, D. L., Burt, M. L., Rexstad, E. A., and Thomas, L. (2013). Spatial models for distance sampling data: recent developments and future directions. *Methods. Ecol. Evol.* 4, 1001–1010. doi: 10.1111/2041-210X.12105

Miller, D. L., Rexstad, E., Thomas, L., Marshall, L., and Laake, J. L. (2019). Distance sampling in R. *J. Stat. Soft.* 89, 1–28. doi: 10.18637/jss.v089.i01

Minton, G., Peter, C., Poh, A. N. Z., Ngeian, J., Braulik, G., Hammond, P. S., et al. (2013). Population estimates and distribution patterns of Irrawaddy dolphins (*Orcaella brevirostris*) and Indo-Pacific finless porpoises (*Neophocaena phocaenoides*) in the Kuching Bay, Sarawak. *Raffles Bull. Zool.* 61, 877–888.

Mizroch, S. A., Beard, J. A., and Lynde, M. (1990). Computer-assisted photo-identification of humpback whales. *Rep. Int. Whal. Commn. (Special Issue 12),* 63–70.

Moore, J. E., and Barlow, J. (2011). Bayesian state-space model of fin whale abundance trends from a 1991-2008 time series of line-transect surveys in the

California Current. *J. Appl. Ecol.* 48, 195–205. doi: 10.1111/j.1365-2664.2011.02018.x

Moore, J. E., and Barlow, J. P. (2013). Declining abundance of beaked whales (Family Ziphiidae) in the California current large marine ecosystem. *PLoS One* 8:e52770. doi: 10.1371/journal.pone.0052770

Moore, J. E., and Barlow, J. P. (2014). Improved abundance and trend estimates for sperm whales in the eastern North Pacific from Bayesian hierarchical modelling. *Endang. Species Res.* 25, 141–150. doi: 10.3354/esr00633

Moreland, E. E., Cameron, M. F., Angliss, R. P., and Boveng, P. L. (2015). Evaluation of a ship-based unoccupied aircraft system (UAS) for surveys of spotted and ribbon seals in the Bering Sea pack ice. *J. Unmanned Aerial Syst.* 3, 114–122. doi: 10.1139/juvs-2015-0012

Moxley, J. H., Bogomolni, A., Hammill, M. O., Moore, K. M. T., Polito, M. J., Sette, L., et al. (2017). Google haul out: earth observation imagery and digital aerial surveys in coastal wildlife management and abundance estimation. *Bioscience* 67, 760–768. doi: 10.1093/biosci/bix059

Nachtsheim, D. A., Viquerat, S., Ramírez-Martínez, N. C., Unger, B., Siebert, U., and Gilles, A. (2021). Small cetacean in a human high-use area: trends in harbor porpoise abundance in the North Sea over two decades. *Front. Mar. Sci.* 7:606609. doi: 10.3389/fmars.2020.606609

Noad, M. J., Dunlop, R. A., Paton, D., and Cato, D. H. (2011). Absolute and relative abundance estimates of Australian east coast humpback whales (*Megaptera novaeangliae*). *J. Cetacean Res. Manage. (Special Issue 3),* 243–252.

Olesiuk, P. F., Bigg, M. A., and Ellis, G. M. (1990). Life history and population dynamics of resident killer whales (*Orcinus orca*) in the coastal waters of British Columbia and Washington State. *Rep. Int. Whal. Commn.* 12, 209–243.

Pace, R. M., Corkeron, P. J., and Kraus, S. D. (2017). State-space mark-recapture estimates reveal a recent decline in abundance of North Atlantic right whales. *Ecol. Evol.* 7, 8730–8741. doi: 10.1002/ece3.3406

Palka, D. L., and Hammond, P. S. (2001). Accounting for responsive movement in line transect estimates of abundance. *Can. J. Fish. Aquat. Sci.* 58, 777–787. doi: 10.1139/cjfas-58-4-777

Palsbøll, P., Allen, J., Bérubé, M., Clapham, P. J., Feddersen, T. P., Hammond, P. S., et al. (1997). Genetic tagging of humpback whales. *Nature* 388, 767–769. doi: 10.1038/42005

Panigada, S., Lauriano, G., Donovan, G., Pierantonio, N., Cañadas, A., Vazquez, J. A., et al. (2017). Estimating cetacean density and abundance in the Central and Western Mediterranean Sea through aerial surveys: implications for management. *Deep Sea Res. II* 141, 41–58.

Paschoalini, M., Almeida, R. M., Trujillo, F., Melo-Santos, G., Marmontel, M., Pavanato, H. J., et al. (2020). On the brink of isolation: population estimates of the Araguaian river dolphin in a human-impacted region in Brazil. *PLoS One* 15:e0231224. doi: 10.1371/journal.pone.0231224

Pike, D. G., Gunnlaugsson, T., Mikkelsen, B., Halldórsson, S. D., and Víkingsson, G. (2019). Estimates of the abundance of cetaceans in the central North Atlantic based on the NASS Icelandic and Faroese shipboard surveys conducted in 2015. *NAMMCO Sci. Publ.* 11, 1–22. doi: 10.7557/3.4941

Pike, D. G., Gunnlaugsson, T., Sigurjónsson, J., and Víkingsson, G. (2020). Distribution and abundance of cetaceans in Icelandic waters over 30 years of aerial surveys. *NAMMCO Sci. Publ* 11, 1–22. doi: 10.7557/3.4805

Pitcher, K. W., Olesiuk, P. E., Brown, R. F., Lowry, M. S., Jefries, S. J., Sease, J. L., et al. (2007). Abundance and distribution of the eastern North Pacific Steller sea lion (*Eumetopias jubatus*) population. *Fish. Bull. U.S.* 107, 102–115.

Pollock, K., Marsh, H. D., Lawler, I. R., and Alldredge, M. W. (2006). Estimating animal abundance in heterogeneous environments: an application to aerial surveys for dugongs. *J. Wildl. Manage.* 70, 255–262.

Preen, A. (2004). Distribution, abundance and conservation status of dugongs and dolphins in the southern and western Arabian Gulf. *Biol. Conserv.* 118, 205–218. doi: 10.1016/j.biocon.2003.08.014

Punt, A. E., Siple, M., Francis, T. B., Hammond, P. S., Heinemann, D., Long, K. J., et al. (2021). Can we manage marine mammal bycatch effectively in low data environments? *J. Appl. Ecol.* 58, 596–607. doi: 10.1111/1365-2664.13816

Punt, A. E., Siple, M., Sigurðsson, G. M., Víkingsson, G., Francis, T. B., Granquist, S. M., et al. (2020). Evaluating management strategies for marine mammal populations: an example for multiple species and multiple fishing sectors in Iceland. *Can. J. Fish. Aquat. Sci.* 77, 1316–1331. doi: 10.1139/cjfas-2019-0386

R Core Team (2020). *R: A Language and Environment for Statistical Computing*. Vienna: R Foundation for Statistical Computing.

Ramp, C., Delarue, J., Bérubé, M., Hammond, P. S., and Sears, R. (2014). Fin whale survival and abundance in the Gulf of St. Lawrence, Canada. *Endang. Species Res.* 23, 125–132. doi: 10.3354/esr00571

Rankin, S., Oedekoven, C., and Archer, F. (2020). Mark recapture distance sampling: using acoustics to estimate the fraction of dolphins missed by observers during shipboard line-transect surveys. *Environ. Ecol. Stat.* 27, 233–251.

Read, A. J., Drinker, P., and Northridge, S. (2006). Bycatch of marine mammals in U.S. and global fisheries. *Conserv. Biol.* 20, 163–169. doi: 10.1111/j.1523-1739.2006.00338.x

Read, A. J., Urian, K. W., Wilson, B., and Waples, D. M. (2003). Abundance of bottlenose dolphins in the bays, sounds and estuaries of North Carolina. *Mar. Mamm. Sci.* 19, 59–73. doi: 10.1111/j.1748-7692.2003.tb01092.x

Rogan, E., Cañadas, A., Macleod, K., Santos, M. B., Mikkelsen, B., Uriarte, A., et al. (2017). Distribution, abundance and habitat use of deep diving cetaceans in the North East Atlantic. *Deep Sea Res. II* 141, 8–19. doi: 10.1016/j.dsr2.2017.03.015

Rugh, D. J., Hobbs, R. C., Lerczak, J. A., and Breiwick, J. M. (2005). Estimates of abundance of the eastern North Pacific stock of gray whales (*Eschrichtius robustus*) 1997–2002. *J. Cetacean Res. Manage.* 7, 1–12.

Russell, D. J. F., Morris, C. D., Duck, C. D., Thompson, D., and Hiby, L. (2019). Monitoring long-term changes in UK grey seal pup production. *Aquat. Conserv. Mar. Freshw. Ecosyst.* 29, 24–39. doi: 10.1002/aqc.3100

Ruzzante, D. E., McCracken, G. R., Førland, B., MacMillan, J., Notte, D., Buhariwalla, C., et al. (2019). Validation of close-kin mark-recapture (CKMR) methods for estimating population abundance. *Methods Ecol. Evol.* 10, 1445–1453. doi: 10.1111/2041-210X.13243

Salgado Kent, C., Jenner, K., Jenner, M., Bouchet, P., and Rexstad, E. (2012). Southern hemisphere breeding stock D humpback whale population estimates from North West Cape, Western Australia. *J. Cetacean Res. Manage.* 12, 29–38.

Scheidat, M., Verdaat, H., and Aarts, G. (2012). Using aerial surveys to estimate density and distribution of harbour porpoises in Dutch waters. *J. Sea Res.* 69, 1–7. doi: 10.1016/j.seares.2011.12.004

Scott, M. D., Wells, R. S., and Irvine, A. B. (1990). "A long-term study of bottlenose dolphins on the west coast of Florida," in *The Bottlenose Dolphin*, eds S. Leatherwood, and R. R. Reeves (San Diego, CA: Academic Press), 235–244.

Seber, G. A. F. (1982). *The Estimation of Animal Abundance and Related Parameters*. London: Charles Griffin.

Sepúlveda, M., Oliva, D., Urra, A., and Perez-Alvarez, M. J. (2011). Distribution and abundance of the South American sea lion *Otaria flavescens* (Carnivora: Otariidae) along the central coast off Chile. *Rev. Chilena Hist. Nat.* 84, 97–106.

Sepúlveda, M., Quiñones, R. A., Carrasco, P., and Pérez-Alvarez, M. J. (2012). Daily and seasonal variation in the haul-out behavior of the South American sea lion. *Mamm. Biol.* 77, 288–292. doi: 10.1016/j.mambio.2012.03.008

Sharples, R. J., MacKenzie, M. L., and Hammond, P. S. (2009). Estimating seasonal abundance of a central place forager using counts and telemetry data. *Mar. Ecol. Prog. Ser.* 378, 289–298. doi: 10.3354/meps07827

Shaughnessy, P. D., Troy, S. K., Kirkwood, R., and Nicholls, A. O. (2000). Australian fur seals at Seal Rocks, Victoria: pup abundance by mark–recapture estimation shows continued increase. *Wildl. Res.* 27, 629–633. doi: 10.1071/WR99046

Sigourney, D. B., Chavez-Rosales, S., Conn, P. B., Garrison, L., Josephson, E., and Palka, D. (2020). Developing and assessing a density surface model in a Bayesian hierarchical framework with a focus on uncertainty: insights from simulations and an application to fin whales (*Balaenoptera physalus*). *PeerJ* 8:e8226. doi: 10.7717/peerj.8226

Siniff, D. B., DeMaster, D. P., Hofman, R. J., and Eberhardt, L. L. (1977). An analysis of the dynamics of a Weddell seal population. *Ecol. Monogr.* 47, 319–335. doi: 10.2307/1942520

Siple, M. C. (2021). *mcsiple/mmrefpoints: First Release (v0.0.1)*. Genève: Zenodo. doi: 10.5281/zenodo.4758402

Skaug, H. J., Øien, N., Schweder, T., and Bøthun, G. (2004). Abundance of minke whales (*Balaenoptera acutorostrata*) in the Northeast Atlantic: variability in time and space. *Can. J. Fish. Aquat. Sci.* 61, 870–886. doi: 10.1139/f04-020

Slooten, E., Dawson, S. M., and Rayment, W. J. (2004). Aerial surveys for coastal dolphins: abundance of Hector's dolphins off the South Island west coast, New Zealand. *Mar. Mamm. Sci.* 20, 477–490. doi: 10.1111/j.1748-7692.2004.tb01173.x

Smith, B. D., Braulik, G., Strindberg, S., Ahmed, B., and Mansur, R. (2006). Abundance of Irrawaddy dolphins (*Orcaella brevirostis*) and Ganges river dolphins (*Platanista gangetica gangetica*) estimated using concurrent counts made by independent teams in waterways of the Sundarbans mangrove forest in Bangladesh. *Mar. Mamm. Sci.* 22, 527–547. doi: 10.1111/j.1748-7692.2006.00041.x

Smith, H. C., Pollock, K., Waples, K., Bradley, S., and Bejder, L. (2013). Use of the robust design to estimate seasonal abundance and demographic parameters of a coastal bottlenose dolphin (*Tursiops aduncus*) population. *PLoS One* 8:e76574. doi: 10.1371/journal.pone.0076574

Smith, T. D. (1981). Line-transect techniques for estimating density of porpoise schools. *J. Wildl. Manage.* 45, 650–657. doi: 10.2307/3808698

Smith, T. D., Allen, J., Clapham, P. J., Hammond, P. S., Katona, S. K., Larsen, F., et al. (1999). An ocean-basin-wide mark-recapture study of the North Atlantic humpback whale (*Megaptera novaeangliae*). *Mar. Mamm. Sci*. 15, 1–32. doi: 10.1111/j.1748-7692.1999.tb00779.x

Speckman, S. G., Chernook, V. I., Burn, D. M., Udevitz, M. S., Kochnev, A. A., Vasilev, A., et al. (2011). Results and evaluation of a survey to estimate Pacific Walrus population size, 2006. *Mar. Mamm. Sci.* 27, 514–553. doi: 10.1111/j.1748-7692.2010.00419.x

Stenson, G. B., Gosselin, J.-F., Lawson, J. W., Buren, A., Goulet, P., Lang, S. L. C., et al. (2020). *Estimating Pup Production of Northwest Atlantic Harp Seals, Pagophilus groenlandicus, in 2017. DFO Canadian Science Advisory Secretariat Research Document 2020/056*. Available online at: https://www.dfo-mpo.gc.ca/csas-sccs/Publications/ResDocs-DocRech/2020/2020_056-eng.html (accessed September 10, 2021).

Stevenson, B. C., Borchers, D. L., and Fewster, R. M. (2018). Cluster capture-recapture to account for identification uncertainty on aerial surveys of animal populations. *Biometrics* 75, 326–336. doi: 10.1111/biom.12983

Stevick, P. T., Allen, J., Clapham, P. J., Friday, N., Katona, S. K., Larsen, F., et al. (2003). North Atlantic humpback whale abundance and rate of increase four decades after protection from whaling. *Mar. Ecol. Prog. Ser.* 258, 263–273. doi: 10.3354/meps258263

Sucunza, F., Danilewicz, D., Andriolo, A., Azevedo, A. F., Secchi, E. R., and Zerbini, A. N. (2020). Distribution, habitat use, and abundance of the endangered franciscana in southeastern and southern Brazil. *Mar. Mamm. Sci*. 36, 421–435. doi: 10.1111/mms.12650

Sucunza, F., Danilewicz, D., Cremer, M., Andriolo, A., and Zerbini, A. N. (2018). Refining estimates of availability bias to improve assessments of the conservation status of an endangered dolphin. *PLoS One* 13:e0194213. doi: 10.1371/journal.pone.0194213

Taylor, B. L., Martinez, M., Gerrodette, T., Barlow, J., and Hrovat, Y. N. (2007). Lessons from monitoring trends in abundance of marine mammals. *Mar. Mamm. Sci.* 23, 157–175. doi: 10.1111/j.1748-7692.2006.00092.x

Taylor, B. L., Rojas-Bracho, L., Moore, J., Jaramillo-Legorreta, A., Ver Hoef, J. M., Cardenas-Hinojosa, G., et al. (2017). Extinction is imminent for Mexico's endemic porpoise unless fishery bycatch is eliminated. *Conserv. Lett.* 10, 588–595. doi: 10.1111/conl.12331

Taylor, M. K., Laake, J., McLoughlin, P. D., Cluff, H. D., and Messier, F. (2009). Demography and population viability of polar bears in the Gulf of Boothia, Nunavut. *Mar. Mamm. Sci.* 25, 778–796. doi: 10.1111/j.1748-7692.2009.00302.x

Thomas, L., Russell, D. J. F., Duck, C., Morris, C. D., Lonergan, M., Empacher, F., et al. (2019). Modelling the population size and dynamics of the British grey seal. *Aquat. Conserv. Mar. Freshw. Ecosyst* 29, 6–23. doi: 10.1002/aqc.3134

Thomas, L. T., Buckland, S. T., Rexstad, E. A., Laake, J. L., Strindberg, S., Hedley, S. L., et al. (2010). Distance software: design and analysis of distance sampling surveys for estimating population size. *J. Appl. Ecol.* 47, 5–14. doi: 10.1111/j.1365-2664.2009.01737.x

Thompson, D., Duck, C. D., Morris, C. D., and Russell, D. J. F. (2019). The status of harbour seals (*Phoca vitulina*) in the UK. *Aquat. Conserv. Mar. Freshw. Ecosyst*. 29, 40–60. doi: 10.1002/aqc.3110

Thompson, P. M., Tollit, D. J., Wood, D., Corpe, H. M., Hammond, P. S., and MacKay, A. (1997). Estimating harbour seal abundance and status in an estuarine habitat in N.E. Scotland. *J. Appl. Ecol.* 34, 43–52. doi: 10.2307/2404846

Trites, A. W., and Larkin, P. A. (1996). Changes in the abundance of Steller sea lions (*Eumetopias jubatus*) in Alaska from 1956 to 1992: how many were there? *Aquat. Mamm.* 22, 153–166.

Turnock, B. J., and Quinn, T. J. (1991). The effect of responsive movement on abundance estimation using line transect sampling. *Biometrics* 47, 701–715. doi: 10.2307/2532156

Udevitz, M. S., Chadwick, V. J., Fischbach, A. S., and Garlich-Miller, J. L. (2009). Modeling haul-out behavior of walruses in Bering Sea ice. *Can. J. Zool.* 87, 1111–1128. doi: 10.1139/Z09-098

Urian, K., Read, A., Gorgone, A., Balmer, B., Wells, R. S., Berggren, P., et al. (2015). Recommendations for photo-identification methods used in capture-recapture models with cetaceans. *Mar. Mamm. Sci.* 31, 298–321. doi: 10.1111/mms.12141

Ver Hoef, J. M., Cameron, M. F., Boveng, P. L., London, J. M., and Moreland, E. E. (2014). A spatial hierarchical model for abundance of three ice-associated seal species in the eastern Bering Sea. *Stat. Methodol.* 17, 46–66. doi: 10.1016/j.stamet.2013.03.001

Vidal, O., Barlow, J., Hurtado, L. A., Torre, J., Cendrón, P., and Ojeda, Z. (1997). Distribution and abundance of the Amazon river dolphin (*Inia geoffrensis*) and the tucuxi (*Sotalia fluviatilis*) in the upper Amazon River. *Mar. Mamm. Sci.* 13, 427–445. doi: 10.1111/j.1748-7692.1997.tb00650.x

Víkingsson, G. A., Pike, D. G., Desportes, G., Øien, N., Gunnlaugsson, T., and Bloch, D. (2013). Distribution and abundance of fin whales (*Balaenoptera physalus*) in the Northeast and Central Atlantic as inferred from the North Atlantic Sightings Surveys 1987-2001. *NAMMCO Sci. Publ.* 7, 49–72. doi: 10.7557/3.2705

Wade, P. R. (1998). Calculating limits to the allowable human-caused mortality of cetaceans and pinnipeds. *Mar. Mamm. Sci.* 14, 1–37. doi: 10.1111/j.1748-7692.1998.tb00688.x

Wade, P. R., Kennedy, A., LeDuc, R., Barlow, J., Carretta, J., Shelden, K., et al. (2011). The world's smallest whale population. *Biol. Lett.* 7, 83–85. doi: 10.1098/rsbl.2010.0477

Wege, M., Etienne, M.-P., Oosthuizen, W. C., Reisinger, R. R., Bester, M. N., and de Bruyn, N. (2016). Trend changes in sympatric Subantarctic and Antarctic fur seal pup populations at Marion Island, Southern Ocean. *Mar. Mamm. Sci.* 32, 960–982. doi: 10.1111/mms.12306

Westlake, R. L., Perryman, W. L., and Ono, K. A. (1997). Comparison of vertical aerial photographic and ground censuses of Steller sea lions at Año Nuevo Island, July 1990-1993. *Mar. Mamm. Sci*. 13, 207–218. doi: 10.1111/j.1748-7692.1997.tb00628.x

Whitehead, H., and Gordon, J. (1986). Methods of obtaining data for assessing and modelling sperm whale populations which do not depend on catches. *Rep. Int. Whal. Commn. (Special Issue 8),* 149–165.

Whitehead, H. P. (1982). Populations of humpback whales in the northwest Atlantic. *Rep. Int. Whal. Commn.* 32, 345–353.

Whitehead, H. P., Payne, R., and Payne, M. (1986). Population estimate for right whales off Peninsula Valdes, Argentina, 1971-1976. *Rep. Int. Whal. Commn. (Special Issue 10),* 169–171.

Williams, R., Ashe, E., Gaut, K., Gryba, R., Moore, J. E., Rexstad, E., et al. (2017). Animal counting toolkit: a practical guide to small-boat surveys for estimating abundance of coastal marine mammals. *Endang. Species Res.* 34, 149–165. doi: 10.3354/esr00845

Williams, R., Burgess, M. G., Ashe, A., Gaines, S. D., and Reeves, R. R. (2016a). U.S. seafood import restriction presents opportunity and risk. *Science* 354, 1372–1374. doi: 10.1126/science.aai8222

Williams, R., Hedley, S. L., Branch, T. A., Bravington, M. V., Zerbini, A. N., and Findlay, K. P. (2011). Chilean blue whales as a case study to illustrate methods to estimate abundance and evaluate conservation status of rare species. *Conserv. Biol.* 25, 526–535. doi: 10.1111/j.1523-1739.2011.01656.x

Williams, R., Hedley, S. L., and Hammond, P. S. (2006). Modeling distribution and abundance of Antarctic baleen whales using ships of opportunity. *Ecol. Soc.* 11:1.

Williams, R., Leaper, R., Zerbini, A. N., and Hammond, P. S. (2007). Methods for investigating measurement error in cetacean line-transect surveys. *J. Mar. Biol. Assoc. U.K.* 87, 313–320. doi: 10.1017/S0025315407055154

Williams, R., Moore, J. E., Gomez-Salazar, C., Trujillo, F., and Burt, L. (2016b). Searching for trends in river dolphin abundance: designing surveys for looming threats, and evidence for opposing trends of two species in the Colombian Amazon. *Biol. Conserv.* 195, 136–145. doi: 10.1016/j.biocon.2015.12.037

Williams, R., and Thomas, L. (2007). Distribution and abundance of marine mammals in the coastal waters of British Columbia, Canada. *J. Cetacean Res. Manage.* 9, 15–28.

Williams, R., and Thomas, L. (2009). Cost-effective abundance estimation of rare animals: testing performance of small-boat surveys for killer whales in British Columbia. *Biol. Conserv.* 142, 1542–1547. doi: 10.1016/j.biocon.2008.12.028

Williamson, L. D., Brookes, K. L., Scott, B. E., Graham, I. M., Bradbury, G., Hammond, P. S., et al. (2016). Echolocation detections and digital video surveys provide reliable estimates of the relative density of harbour porpoises. *Methods Ecol. Evol.* 7, 762–769. doi: 10.1111/2041-210X.12538

Wilson, B., Hammond, P. S., and Thompson, P. M. (1999). Estimating size and assessing trends in a coastal bottlenose dolphin population. *Ecol. Appl.* 9, 288–300. doi: 10.2307/2641186

Zerbini, A. N., Waite, J. M., Durban, J. W., LeDuc, R., Dahlheim, M. E., and Wade, P. R. (2007). Estimating abundance of killer whales in nearshore waters of the Gulf of Alaska and Aleutian Islands using line-transect sampling. *Mar. Biol.* 150, 1033–1045. doi: 10.1007/s00227-006-0347-8

Zerbini, A. N., Waite, J. M., Laake, J. L., and Wade, P. R. (2006). Abundance, trends and distribution of baleen whales off Western Alaska and the central Aleutian Islands. *Deep Sea Res. I* 53, 1772–1790. doi: 10.1016/j.dsr.2006.08.009

Zurell, D., Franklin, J., König, C., Bouchet, P. J., Dormann, C. F., Elith, J., et al. (2020). A standard protocol for reporting species distribution models. *Ecography* 43, 1261–1277. doi: 10.1111/ecog.04960

Conflict of Interest: RR is employed by Okapi Wildlife Associates Reg'd.

The remaining authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

frontiers in Marine Science

ORIGINAL RESEARCH
published: 20 October 2021
doi: 10.3389/fmars.2021.719956

Estimating Cetacean Bycatch From Non-representative Samples (I): A Simulation Study With Regularized Multilevel Regression and Post-stratification

Matthieu Authier[1,2]*, *Etienne Rouby*[3] *and Kelly Macleod*[4]

[1] Observatoire Pelagis, UMS 3462, CNRS-La Rochelle Université, La Rochelle, France, [2] ADERA, Pessac, France, [3] Centre d'Études Biologiques de Chizé, UMR 7372 CNRS-La Rochelle Université, Villiers-en-Bois, France, [4] Joint Nature Conservation Committee, Aberdeen, United Kingdom

Bycatch, the non-intentional capture or killing of non-target species in commercial or recreational fisheries, is a world wide threat to protected, endangered or threatened species (PETS) of marine megafauna. Obtaining accurate bycatch estimates of PETS is challenging: the only data available may come from non-dedicated schemes, and may not be representative of the whole fisheries effort. We investigated, with simulated data, a model-based approach for estimating PETS bycatch from non-representative samples. We leveraged recent development in the statistical analysis of surveys, namely regularized multilevel regression with post-stratification, to infer total bycatch under realistic scenarios of data sampling such as under-sampling or over-sampling when PETS bycatch risk is high. Post-stratification is a survey technique to re-align the sample with the population and addresses the problem of non-representative samples. Post-stratification requires to sub-divide a population of interest into potentially hundreds of cells corresponding to the cross-classification of important attributes. Multilevel regression accommodate this data structure, and the statistical technique of regularization can be used to predict for each of these hundreds of cells. We illustrated these statistical ideas by modeling bycatch risk for each week within a year with as few as a handful of observed PETS bycatch events. The model-based approach led to improvements, under mild assumptions, both in terms of accuracy and precision of estimates and was more robust to non-representative samples compared to more design-based methods currently in use. In our simulations, there was no detrimental effects of using the model-based even when sampling was representative. Estimating PETS bycatch ideally requires dedicated observer schemes and adequate coverage of fisheries effort. We showed how a model-based approach combining sparse data typical of PETS bycatch and recent methodological developments can help when both dedicated observer schemes and adequate coverage are challenging to implement.

Keywords: marine megafauna, conservation, modeling, statistics, Bayesian, anthropogenic removals

Edited by:
Jeff Moore,
Southwest Fisheries Science Center (NOAA), United States

Reviewed by:
Peter Dillingham,
University of Otago, New Zealand
Devin S. Johnson,
Pacific Islands Fisheries Science Center, National Marine Fisheries Service, United States

Correspondence:
Matthieu Authier
mauthier@univ-lr.fr

Specialty section:
This article was submitted to Marine Megafauna, a section of the journal Frontiers in Marine Science

Received: *03 June 2021*
Accepted: *17 September 2021*
Published: *20 October 2021*

Citation:
Authier M, Rouby E and Macleod K (2021) Estimating Cetacean Bycatch From Non-representative Samples (I): A Simulation Study With Regularized Multilevel Regression and Post-stratification. Front. Mar. Sci. 8:719956. doi: 10.3389/fmars.2021.719956

1. INTRODUCTION

Bycatch, the non-intentional capture or killing of non-target species in commercial or recreational fisheries, is a world wide threat to protected, endangered or threatened species (PETS) of marine megafauna (Gray and Kennelly, 2018), including seabirds (Martin et al., 2019), elasmobranchs (Pacoureau et al., 2021) and cetaceans (Avila et al., 2018). Bycatch in fishing gears, such as gillnets, is currently driving some small cetacean species to extinction (Brownell et al., 2019; Jaramillo-Legorreta et al., 2019). The European Commission recently issued infringement procedures against several Members States for failing to correctly transpose some provisions of European environmental law (the Habitats Directive, Council Directive 92/43/EEC), in particular the obligations related to the establishment of a coherent monitoring scheme of cetacean bycatch[1]. The Data Collection Framework (DCF) provides a common framework in the European Union (EU) to collect, manage, and share data within the fisheries sector (Anonymous, 2019a). The Framework indicates that the Commission shall establish a Multi-Annual Union Programme (EU-MAP) for the collection and management of fisheries data which should be inclusive of data that allows the assessment of fisheries' impact on marine ecosystems. With respect to PETS (including cetaceans), the collection of high quality data usually requires a dedicated sampling scheme and methodology, and is generally different from those applied under the DCF (Stransky and Sala, 2019): "EU MAP remains not well-suited for the dedicated monitoring of rare and protected bycatch in high-risk fisheries since its main focus is the statistically-sound random sampling of all commercial fisheries (Ulrich and Doerner, 2021, p. 126)." In practice, the introduction of any programme on PETS bycatch under the DCF may be met with caution because of its perceived potential to disrupt data collection for fisheries management (Stransky and Sala, 2019). This perception implicitly relegates PETS bycatch as a side issue for fishery management rather than an integral part of it. It may explain the usually poor quality of bycatch data on PETS (ICES, 2020a).

Recent EU legislation (Regulation 2019/1241), referred to as the Technical Measures Regulation (TMR), requires Members States to collect *scientific data* on cetacean bycatch for the following métiers: pelagic trawls (single and pair), bottom-set gillnets and entangling nets; and high-opening trawls (Anonymous, 2019b). Unlike its predecessor (Council Regulation EC No. 812/2004), this Regulation does not require the establishment of dedicated observer schemes for cetacean bycatch data collection (Dolman et al., 2020). Furthermore, only vessels of an overall length of 15 m or more are to be monitored, but these represent a small fraction of the European fleet (less than 10% in 2019)[2]. This vessel length criterion introduces bias in the bycatch monitoring data as the sample of vessels larger than 15 m is almost certainly dissimilar to smaller vessels. Even within the sample of vessels that are monitored, pragmatic considerations can complicate sampling. For example, in the United States, observer sampling trips are allocated first by region, port, and month, then randomly to vessels of particular categories within those monthly and spatial strata (ICES, 2009). Random allocation of observers to vessels follows sound statistical methodology and increases the likelihood of collecting unbiased data (Babcock and Pikitch, 2003). In France, observer days are allocated by port and by month for each fishery, but the exact vessel allocation is then negotiated and left at the discretion of skippers (ICES, 2009). Allocation is no longer random as skippers may only accept observers when cetacean bycatch risk is low (Benoît and Allard, 2009). Non-random allocation means potential bias in the collected data for monitoring bycatch as the sub-sample of skippers accepting an observer may be very different from skippers refusing to do so (Babcock and Pikitch, 2003).

One pragmatic solution bypassing observers is to mandate skippers to self-declare the non-intentional capture or killing of any PETS, as already required under the DCF (Anonymous, 2019a). In France, a national law from 2011 mandate fisheries to declare (without fear of prosecution) the bycatch of any cetacean species, but this law remained largely unknown to French fishermen until late 2019 (Cloâtre, 2020). In general, self-reported PETS bycatch data are sub-optimal as they may be heavily biased, non-representative (ICES, 2009) and typically provide poor information on which to base management decisions (National Marine Fisheries Service, 2004). Once again, the set of skippers who choose to declare bycatch may differ markedly from those who do not: for example the former take the extra time required to fill logbooks and thus provide accurate data while the latter do not. If this behavior is correlated to other attributes, e.g., a more acute awareness of threats to PETS resulting in practices that tend to minimize impact on PETS, data collected from skippers reporting bycatch would not be representative. There may also be an element of skippers genuinely forgetting to log PETS bycatch in the bustle of the fishing operation but this is random and unlikely to introduce bias. In addition, ground-truthing, for example with remote-electronic monitoring (REM; Course et al., 2020), would be required in order to ensure the quality and accuracy of self-reported data before their statistical analyses.

Another hurdle, of the statistical kind, with cetacean bycatch is the low frequency of these events. Assuming that implementing a representative sampling program were feasible, if bycatch is a rare event (Komoroske and Lewison, 2015), then few events would be observed for realistic sampling effort (Babcock and Pikitch, 2003; ICES, 2009). This paucity of observed event means a large uncertainty in statistical estimates: with a bycatch rate of the order of 0.01 event per fishing operation, a sample size of 1,100 observed operations would be required to obtain, in the best case scenario (no bias, statistical independence, etc.), the US recommended coefficient of variation of 30% (National Marine Fisheries Service, 2005, 2016; ICES, 2009; Carretta and Moore, 2014). The amount of observer coverage needed to reach this precision depends on fishery size and trip duration (Babcock and Pikitch, 2003). In practice, the sampling error depends on the overall design of the survey, of which the sample size

---

[1] https://ec.europa.eu/info/news/july-infringements-package-commission-moves-against-member-states-not-respecting-eu-energy-rules-2019-jul-26_en

[2] https://appsso.eurostat.ec.europa.eu/nui/show.do?dataset=fish_fleet_alt&lang=en

is only one factor: for example a larger sample size could be needed if there are large "skipper-effects" as the same vessels would contribute fishing operations, and these would not be statistically independent. With a small sample size, uncertainty may be so large as to prevent using estimates altogether, even if one were to assume no bias in the data (Babcock and Pikitch, 2003). Given this challenge and the lack of uptake of dedicated monitoring programmes of cetacean bycatch in Europe over the last decade or more (Sala et al., 2019), it would appear prudent to seek methods of analysis that can handle the few and non-representative data available to robustly estimate bycatch rates.

The problem of having non-representative samples to carry out statistical analyses is ancient (Hansen and Hurwitz, 1946) and widespread: it pops up in many applied disciplines, including election forecasting (Wang et al., 2015; Kiewiet De Jonge et al., 2018), political sciences (Lax and Phillips, 2009; Zahorski, 2020), social sciences (Halsny, 2020), addiction studies (Rhem et al., 2020) or epidemiology (Zhang et al., 2014; Downes et al., 2018). In these disciplines, there are also intrinsic limits on improving the representativeness of sampling. For example, in polling, non-response rates can be above 90% (Forsberg, 2020). In other cases, some populations of interest may be hard to reach (Rhem et al., 2020), or answers may not be honest (St. John et al., 2014). Challenges lie in the accurate estimation of quantities of scientific interest (e.g., the true magnitude of bycatch in a fishery; Babcock and Pikitch, 2003) with the construction of statistical weights that can calibrate a non-representative survey sample to the population targets. Such weights are implicit with simple random sampling where each unit in a population has the same, non-nil, probability of being included in the sample. When inclusion probabilities differ between units, weights inversely proportional to the former can be used to adjust the sample. However, constructing survey weights is in general more elaborate than using inverse probabilities of selection in the sample (Gelman, 2007). Model-based approaches, and multilevel regression modeling with post-stratification in particular, has become an attractive alternative to weighting to adjust non-representative samples (Gelman, 2007).

Multilevel regression modeling allows researchers to summarize how predictions of an outcome of scientific interest vary across statistical units defined by a set of attributes or covariates (Gelman et al., 2021, p. 4): for example bycatch events are a binary outcome at the fishing operation level (a unit) associated with attributes, such as date-time, location, gears and vessels (e.g., Palka and Rossman, 2001). Post-stratification is a standard technique to generalize inferences from a sample to the population by adjusting for known discrepancies between the former and the latter. Post-stratification is a form of adjustment whereby statistical units are sorted out according to an auxiliary variable (hereafter a stratum) after completion of data collection; stratum-level effects (i.e., effects within each stratum or cell) are then estimated, and finally averaged with weights proportional to stratum size to obtain the population-level estimate. Post-stratification differs from blocking as the latter is done before data collection to ensure balance and representativeness at the design stage. Post-stratification is a *post hoc* statistical adjustment done at the analysis stage: it can remove bias, but at the price of an increased variance of estimates. Lennert et al. (1994) provided an early example of model-based estimates of bycatch with post-stratification.

In small samples post-stratification can degrade estimate precision, especially if the number of strata is large as each stratum will typically include very few data, or even not a single datum (the so-called "small-area" problem). In practice, adequate post-stratification may require handling hundreds of cells (the crossing of several attributes; e.g., week by statistical area by gears). Some predictions for each cell may be too noisy, especially if there are sparse or no data for that particular combination of attributes. Multilevel regression can offer a solution as it borrows strength from similar units to improve and stabilize (i.e., regularize) these predictions (Cam, 2012). In other words, multilevel regression allows an efficient use of a sparse sample to estimate the outcome of interest within each cell, even if these cells are very numerous (e.g., several hundreds). The key insight of combining multilevel regression modeling with post-stratification is thus: even if observations are not a representative sample of the population of interest, it may be possible to construct a regression model to first *predict* unobserved cases, and then *post-stratify* to average the fitted regression model's predictions over the population of interest (Gelman et al., 2021, p. 313). Good predictions may be obtained with regularization by means of multilevel models with structured priors (including so called "random-effects" models). The latter can increase precision by inducing shrinkage of parameter estimates across similar post-stratification cells, where similarity is encoded in the model specification (e.g., by using random effects that assume exchangeability). The amount of shrinkage, or partial-pooling across cells, is model-based and thus data-driven. However, in order to be able to leverage the information in the data, some model structure on the parameters of interest is necessary hence the need for structured priors. Relying on a model rather than just empirical means of the response variable addresses the bias-variance problem intrinsic to having a large number of cells in post-stratification, and leverages the large toolbox of regression-based models.

Technically, when data arise as signal plus noise, overfitting occurs when a regression model captures too much of the noise compared to the signal; that is in using an ill-conditioned (unstable) model that will provide an excellent in-sample fit but make poor out-of-sample predictions (Authier et al., 2017b; George and Ročková, 2021). Overfitting may result when using richly parametrized models without using adequate estimation methods such as regularization to stabilize parameter estimates and buffer them against noise (Gelman et al., 2021, p. 459–460). Weakly-informative priors in a Bayesian framework regularize the estimation of the large number of parameters that may be present in a multilevel model. Multilevel modeling takes into account complex data structures with structured prior models for batches of parameters; the simplest example are so-called "random effects" whereby a common (Gaussian) distribution centered on zero and with an unknown variance to be estimated for data is assumed for a group of parameters; for example years or sites (Cam, 2012). This common distribution for the parameters is a prior model, and this model for parameters means that the latter are not independently estimated but in tandem according to the postulated prior model. For example, Sims et al. (2008) used a model-based approach to obtain spatially smoothed estimates of bycatch in a gillnet fishery. Spatial-smoothing (also known as "small-area estimation"; Fay and Herriot, 1979) was used to stabilize estimated bycatch rates by using a Conditional Autoregressive prior model that leverages information from spatial neighbors to improve the prediction at a specific location. Prior models add some soft constraints to the overall model and these constraints are very useful in data sparse settings to mitigate variance and bias in predictions. In other words, these prior models represent additional assumptions about the data, assumptions, which if approximately correct, add information in the analyses and increase the precision and stability of predictions at the cost of a usually small estimation bias. Introducing bias to reduce variance is a common statistical technique known as shrinkage or regularization (George and Ročková, 2021).

Regularized multilevel regression with post-stratification is thus the combination of several important ideas to obtain accurate predictions (Gao et al., 2019). First, post-stratification is a survey technique to re-align the sample with the population and addresses the problem of non-representative samples. In practice, post-stratification requires to sub-divide the population of interest into many cells corresponding to the combination of important attributes. Multilevel regression can be used to accommodate all these cells in a single model, but the problem has now moved to how to obtain useful estimates for all these cells, which can number in the several hundreds. Regularization solves this estimation problem: it introduces model-driven bias in statistical estimates in order to stabilize them. These new developments in the statistical analysis of non-representative samples may help in obtaining a better quantification of bycatch rates and numbers. Our aim is to assess with simulations, the potential of regularized multilevel regression with post-stratification for analyzing already collected bycatch data, with the full knowledge that these data are non-representative and biased in several respects. These biases in sampling are manifold (see above): bias may be due to regulation exempting certain vessels (e.g., no monitoring for vessels smaller than 15 m); to non-dedicated observers or because sampling is driven for other purposes than bycatch monitoring of PETS (commercial discards, stock assessment); or in the case of dedicated schemes, to over-sampling a few "cooperative" skippers or focusing sampling in métiers with the highest or lowest bycatch risk. Our focus will be narrower, honing in on specific sampling scenarios whereby observer coverage is correlated to bycatch risk. In other words, we will assess the potential of regularized multilevel regression with post-stratification to estimate accurately bycatch numbers with samples preferentially collected either during low- or high-bycatch risk periods. Our investigation is largely framed from our knowledge on small cetacean bycatch in European waters, such as short-beaked common dolphin (*Delphinus delphis*, lower observer coverage when bycatch risk is higher) in the Bay of Biscay (Peltier et al., 2021) or harbor porpoises (*Phocoena phocoena*, higher observer coverage when bycatch risk is higher) in the Celtic Seas (Tregenza et al., 1997). In the remainder, we first introduce methods and notations to detail the proposed model to perform multilevel regression with post-stratification with bycatch data, using dolphins as an example. Next, we explain our data simulation scenarios and how we emulate non-representative sampling. We then compare the results (i.e., estimates of bycatch) from the proposed modeling approach with those from the method currently used by the working group on bycatch of protected species from the International Council for the Exploration of the Sea (ICES WGBYC) before concluding on some recommendations for future investigations.

2. MATERIALS AND METHODS

We carried out Monte Carlo simulations to assess the ability of regularized multilevel regression with post-stratification to estimate bycatch risk and bycatch numbers from representative and non-representative samples. ICES WGBYC collate data through an annual call from dedicated and DCF surveys collecting data on the bycatch of PETS through onboard observers or REM. These surveys may be qualified as "design-based" in the sense that, ideally, a representative coverage of fisheries would be sought in order to scale up the observed sample to the whole population using ratio-estimators. There are many caveats around the use of these ratio-estimators as EU MAP is not well-suited for monitoring PETS bycatch (Ulrich and Doerner, 2021). Given these shortcomings in the collection of bycatch data under EU MAP, the data available to ICES WGBYC are unlikely to be representative of fisheries of interest but nevertheless, ratio-estimators are used as part of a Bycatch Risk Approach (BRA) to identify relative risk of bycatch across species and metiers (ICES, 2018). Cetacean bycatch observer programmes may aim at achieving a pre-specified precision for bycatch rates (with a coefficient of variations less than 30%; National Marine Fisheries Service, 2005, 2016; ICES, 2009; Carretta and Moore, 2014). Achieving this is very difficult in practice, and a given coverage of effort deployed by the total fleet is, instead, aimed at: for example 10% (5%) for pair-trawlers (level-3 métier PTM) larger (smaller) than 15 m in France. Data from onboard observer programmes are then used to estimate total bycatch using ratio estimators (Lennert et al., 1994; Julian and Beeson, 1998; Amandè et al., 2012) and the bootstrap or a classical approach (Clopper-Pearson) for uncertainty quantification (ICES, 2018, p. 57). We used an approach similar to that of WGBYC (hereafter referred to as a "design-based" approach) as a benchmark to compare against results from regularized multilevel regression with post-stratification. We honed in on the accurate estimation of the number of bycatch events for a complete fleet. We assume that information on the total effort deployed by a fleet operating in a spatial domain are available and measured without error. This assumption is necessary to scale estimates from the sample to the population. We also assumed that there are no false-negatives in the sample, that is no bycatch event went unrecorded by onboard observers (assuming thereby a dedicated observer programme). These two assumptions are customary with ratio estimators, whether design- or model-based, and do not deviate from current norms. We assume however that these population data on total effort can be disaggregated at a finer temporal scale in order to post-stratify on calendar weeks. This assumption of accurate measurement of effort at the week-level is crucial for post-stratification.

2.1. Notations

The logit transform maps a quantity $p \in]0,1[$ to the real line: $\text{logit}(p) = \log\left(\frac{p}{1-p}\right)$. Its inverse is denoted by $\text{logit}^{-1}(x) = \frac{1}{1+e^{-x}}$ (sometimes called the "expit" transform). Let y_{ijkl} denote the ith fishing operation of vessel j in week k of year l, with $y_{ijkl} = 1$ if a bycatch event occurs and 0 otherwise:

$$y_{ijkl} \sim \text{Bernoulli}\left(\text{p}_{jkl} = \text{logit}^{-1}\left(\mu + \beta_{kl} + \alpha_j\right)\right) \qquad (1)$$

where p_{jkl} is the product of the probability of a bycatch event occurring and the probability of dolphin presence. This unconditional probability p_{jkl}, or "bycatch risk" hereafter, is not indexed by i: although there may be several fishing operations of vessel j in week k of year l, the risk is assumed constant over these. Bycatch risk is a function of several parameters (on a logit scale): μ is the intercept (overall risk), $\alpha_j \sim \mathcal{N}(0, \sigma_{\text{vessel}})$ are (unstructured, normal random effects) vessel-effects accounting for heterogeneity (e.g., "fishing style" of skippers); and β_{kl} are time effects, modeled with a Gaussian Process. A Gaussian process is written as $\mathcal{GP}(m, c)$ where m and c are the mean and covariance functions respectively (Gelman et al., 2014, p. 501). The Gaussian Process prior on the vector of week effects in year l, $\boldsymbol{\beta}_l$, defines this vector as a random function for which the values at any week $1, \ldots, k, \ldots, w$ are drawn form a w-multivariate normal distribution:

$$\beta_{1l}, \ldots, \beta_{wl} \sim \mathcal{N}\left(\left(m(1), \ldots, m(w)\right), \Omega\right) \qquad (2)$$

with mean m and covariance Ω. The function c specifies the covariance between any 2 weeks k and k', with Ω an $w \times w$ covariance matrix with element $\Omega\left(k, k'\right) = c(k, k')$. A Matérn covariance function of order $\frac{3}{2}$ and range parameter fixed to $\frac{3}{2}$ was assumed: $c\left(k, k'\right) = \sigma^2_{\text{year}} \times \left(1 + \frac{2\sqrt{3} \times d(k-k')}{3}\right) \times \exp - \frac{2\sqrt{3} \times d(k-k')}{2}$, where $d(k - k')$ is the temporal distance (in weeks) between weeks k and k'. The distance function was the absolute difference between calendar weeks within the same year: $d(k-k') = |k-k'|$. The choice of the Matérn covariance function translate an assumption of smoothness in the temporal profile of bycatch risk: bycatch risk is assumed to change gradually across weeks, with no abrupt increase or decrease. The range parameter is fixed and not estimated from data. This choice represents an additional assumption whereby the temporal correlation is 0.05 after 4 weeks corresponding to temporal independence after a month. This choice is to some extent arbitrary and represents an additional assumption. In theory, the range parameter could also be estimated from data but we assumed a data sparse setting with limited information (more so with Bernoulli data) to estimate this parameter.

The mean function m of the Gaussian process was modeled (on a logit scale) with a first order random walk, which was evaluated at specific values $k_{\in[1,\ldots,w]}$ corresponding to week number within a year:

$$\begin{cases} \left(m(1), \ldots, m(w)\right) = (\varepsilon_1, \ldots, \varepsilon_w) & \\ \varepsilon_k = 0 & k = 1 \\ \varepsilon_{k+1} \sim \mathcal{N}(\varepsilon_k, \sigma_{\text{week}}) & k > 1 \end{cases} \qquad (3)$$

The order of the random walk prior was assumed fixed at 1 and not estimated from data. This prior choice smooths the first order differences between adjacent elements of $\boldsymbol{\varepsilon}$ and represents an additional assumption, mainly to limit the number of parameters to estimate from the typically sparse data on bycatch. A random walk was chosen as an effective way to reveal the shape of the average risk profile without specifying a family of parametric curves.

The model in Equation (1) is a decomposition of bycatch risk into a time-varying component (at the week-scale, Equation 3; and with an interaction with year, Equation 2) and time-invariant component which can be interpreted as fishing-style effects whereby some skippers may have consistent practices that increase or decrease bycatch risk. Importantly, bycatch risk is modeled here with no attempt to model dolphin presence directly as relevant data to do so may be missing in the general case. Bycatch risk is thus to be estimated for each week of a year, and each of these weeks represent *de facto* a stratum. In any applied case, additional factors, such as statistical area, may need to be included in Equation (1) for improved realism. For simplicity, we did not consider space in simulations, and solely focused on time.

2.2. Data Simulation

To test the ability of model 1 to estimate bycatch risk, data were simulated (**Figure 1**).

1. Bycatch probability conditional on dolphin presence was constant and set to 0.3, that is roughly one fishing operation out of 3 generates a bycatch event when dolphins are present (corresponding to a high risk fishery, e.g., the trawl fishery in the Bay of Biscay).
2. Dolphin presence is seasonal (loosely inspired from the observed pattern of common dolphin in the Bay of Biscay where abundance is higher closer to the coasts in winter; Laran et al., 2017): it peaks at the beginning and end of the year, but quickly drops to 0 for roughly 2 thirds of a year.
3. A fishery of 20 vessels is operating all year round, with an overall activity rate of 80% each week (that is, for any week, $20 \times \frac{80}{100} = 16$ vessels are fishing). Each fishing day (5 days per week), on average 2.3 fishing operations are carried out. The expected total number of fishing operations for a year is $5 \times 52 \times 2.3 \times 16 \approx 10{,}000$. These values were loosely taken from an exploratory analysis of onboard observer data collected on PTM flying the French flag. During each of these operations, a bycatch event may occur depending on dolphin presence at the time and on a skipper-specific risk factor (drawn randomly from a normal distribution with scale parameter set to $\frac{\log(2)}{3}$ to induce moderate heterogeneity on a logit scale; Authier et al., 2017a).

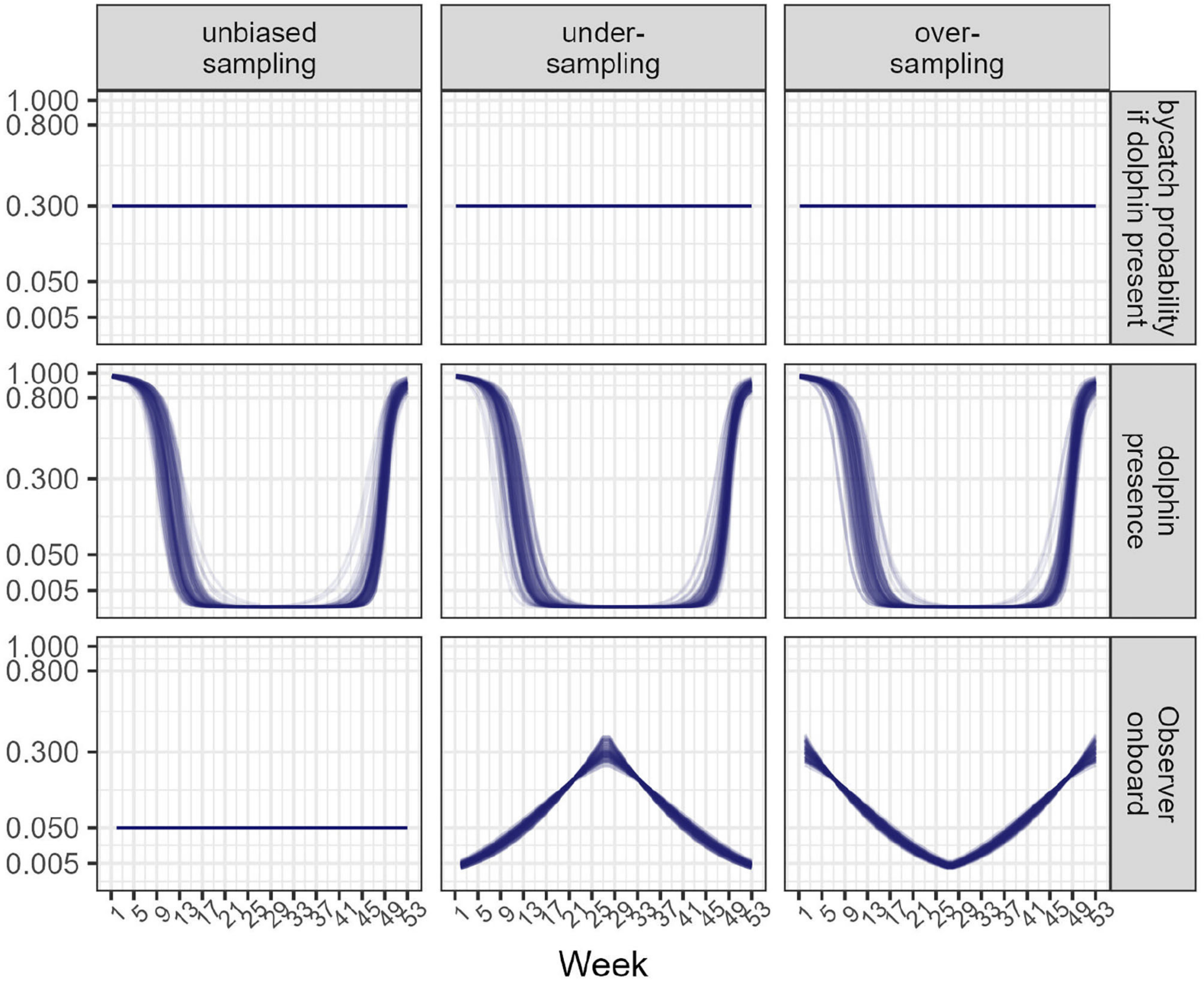


FIGURE 1 | Inputs for data simulation. **Top:** bycatch probability if dolphins are present during a fishing operation. **Middle:** dolphin presence during a year. **Bottom:** Probability for a skipper to accept an observer onboard. **Left:** sampling is unbiased; **Middle column:** sampling is biased downwards (under-sampling). **Right:** sampling is biased upwards (over-sampling). Each line corresponds to one of the 100 data simulations that were carried out. The *y*-axis is on a square-root scale to better visualize small values.

4. Observers are accepted onboard vessels either with a constant probability of 0.05 corresponding to a coverage of 5% of all fishing operations (unbiased sampling scenario) or with a probability that covaries with dolphin presence (biased sampling scenarios). In the latter case, realized coverage is a random variable. With under-sampling, the bulk of the observer data is collected when bycatch risk (the product of dolphin presence and bycatch probability) is nil (**Figure 1**). With over-sampling, the bulk of the observer data is collected when bycatch risk is high but no data are collected when the risk is nil (**Figure 1**).
5. In a year, the number of fishing operations is ≈ 10,000, and the number of bycatch events ≈ 300, which yields a rate of ≈ 3%. This rate is not large, but is not extremely rare either.

Bycatch events were simulated for each fishing operations during a day when an observer was present from a Bernoulli distribution according to the product of bycatch probability given dolphin presence and dolphin presence probability for that day. If no observer was present, no data were recorded. The data-generating mechanism used a parametric function for dolphin presence probability and was different from the statistical model used to analyzed the data (see https://gitlab.univ-lr.fr/mauthier/regularized_bycatch). For each sampling scenario, 100 datasets were generated for 1, 5, 10, or 15 years. All data simulations were carried out in R v.4.0.1 (R Core Team, 2020). When simulating only 1 year of data, Equation (2) is not necessary as there is no between-year variation to estimate: the model can be simplified with the omission of $\boldsymbol{\beta}_l$. Our Monte Carlo study had a comprehensive factorial design crossing (a) sampling regime (either unbiased or not) and (b) sample size as controlled with the number of years for which the observer programme was assumed to have been in operation.

2.3. Estimation

Estimation of the parameters of model 1 from simulated data was carried out in a Bayesian framework using programming language `Stan` (Carpenter et al., 2017) called from `R` v.4.0.1 (R Core Team, 2020) with library `Rstan` (Stan Development Team, 2020). `Stan` uses Hamiltonian dynamics in Markov chain Monte Carlo (MCMC) to sample values from the joint posterior distribution (Carpenter et al., 2017). Weakly-informative priors

$$\text{were used for regularization: } \begin{cases} \mu \sim \mathcal{N}(0, \frac{3}{2}) \\ \text{prop} \sim \mathcal{D}(1, 1, 1) \\ \sigma_{\text{total}} \sim \mathcal{GG}(\frac{1}{2}, \frac{1}{2}, \frac{\log 2}{10}) \end{cases}$$

where $\mathcal{D}()$ denotes the Dirichlet distribution for modeling proportions (such that $\sum_{i=1}^{3} \text{prop}_i = 1$) and $\mathcal{GG}()$ the Gamma-Gamma distribution for scale parameters (Griffin and Brown, 2017; Pérez et al., 2017). With this simplex parametrization, chosen to improve mixing and ease estimation with Monte Carlo methods (He et al., 2007), the several variance components of

$$\text{the model were: } \begin{cases} \sigma^2_{\text{vessel}} = \sigma^2_{\text{total}} \times \text{prop}_1 \\ \sigma^2_{\text{week}} = \sigma^2_{\text{total}} \times \text{prop}_2 \\ \sigma^2_{\text{year}} = \sigma^2_{\text{total}} \times \text{prop}_3 \end{cases}$$

These priors are weakly-informative (Gabry et al., 2019): the prior for the intercept covers the whole interval between 0 and 1 on the probability scale but is informative on the logit scale. The prior for the scale (square-root of the variance) is heavy tailed and has a median set to $\frac{\log 10}{2}$ (Griffin and Brown, 2017; Pérez et al., 2017), which translate an assumption about the plausible range of variations in bycatch risk spanning *a priori* two full order of magnitude from one tenth to a ten-fold increase compared to the mean bycatch rate. Thirty random realizations from our choice of priors are depicted on **Figure 2**: the whole interval between 0 and 1 is covered, and between-week variations can be large or small.

For each simulated dataset, four chains were initialized from diffuse random starting points (Carpenter et al., 2017, p. 20) and run for a total of 1,000 iterations, discarding the first 500 as warm-up. Default settings for the No-U-Turn Sampler (NUTS) were changed to 0.99 for `adapt delta` and 15 for `max treedepth` (Hoffman and Gelman, 2014). NUTS uses Hamiltonian Dynamics in MCMC and typically requires shorter runs than other MCMC algorithms both to reach convergence and to obtain an equivalent Effective Sample Size from the posterior (Hoffman and Gelman, 2014; Monnahan et al., 2017). Parameter convergence was assessed using the $\hat{R}$ statistics (Vehtari et al., 2019) and assumed if $\hat{R} < 1.025$. Upon diagnosing convergence of all parameters, a combined sample of $4 \times 500 = 2,000$ MCMC values were obtained to approximate the joint posterior distribution. Let $\hat{\mu}^{[m]}, \hat{\beta}^{[m]}_{kl}, \hat{\sigma}^{[m]}_{\text{vessel}}$ denote the mth MCMC sample for parameters μ, β_{kl} and σ_{vessel}. Bycatch risk $\hat{\text{p}}_{j^*kl}$ for a randomly chosen vessel j^* operating in week k of year l was computed from the mth MCMC draw from the joint posterior distribution as:

$$\hat{\text{p}}^{[m]}_{j^*kl} = \text{logit}^{-1}\left(\hat{\mu}^{[m]} + \hat{\beta}^{[m]}_{kl} + \hat{\alpha}^{[m]}_{j^*}\right) \tag{4}$$

where $\hat{\alpha}^{[m]}_{j^*} \sim \mathcal{N}(0, \hat{\sigma}^{[m]}_{\text{vessel}})$. This predicted bycatch risk incorporates between-vessel variability, that is it takes into account the fishing style of skippers. The predicted risk (on a logit scale) for a random chosen skipper is $\hat{\alpha}^{[m]}_{j^*}$ and was drawn from the posterior predictive distribution: not all skippers may be observed in the sample, and but the subset of skippers that accept an observer can be used to estimate a between-skipper variance in bycatch risk. In practice, the number of fishing operations carried out in the course of a week in a year by individual skippers is unknown, although the aggregated number of fishing operations may be known. If totals by skippers were available, and all skippers had been sampled, it would be more efficient to use skipper-specific estimated risk, but we did not assume that this would necessarily be the case.

The total number of bycatch events, T_{bycatch} was estimated as the average over the 2,000 MCMC draws from the posterior:

$$\hat{T}_{\text{model-based bycatch}} = \frac{1}{2000} \sum_{m=1}^{2000} \left(\sum_{l=1}^{n_{\text{year}}} \sum_{k=1}^{n_{\text{week}}} \hat{\text{p}}^{[m]}_{j^*kl} \times N_{kl} \right) \tag{5}$$

where N_{kl} is the total number of fishing operations that took place is week k of year l. The total number of strata for post-stratification was $n_{\text{year}} \times n_{\text{week}}$, with a maximum of $15 \times 52 = 780$ cells. Highest Posterior Density credible intervals at the 80% level were computed with function `HPDinterval` from package `coda` (Plummer et al., 2006) for uncertainty evaluation. Equation (5) is an instance of a ratio-estimator with post-stratification, except that it uses model-based estimates of bycatch risk. This model-based approach regularizes estimates with partial pooling (Gelman and Shalizi, 2013): the variance of estimates is greatly reduced by introducing some bias with structured priors (Gao et al., 2019). Our results were benchmarked against an approach similar to that of ICES WGBYC whereby total number of bycatch events was estimated1 as:

$$\hat{T}_{\text{design-based bycatch}} = \sum_{l=1}^{n_{\text{year}}} \left(\bar{p}_l \times \sum_{k=1}^{n_{\text{week}}} N_{kl} \right) \tag{6}$$

where $\bar{p}_l$ is the average bycatch risk estimated as the mean from the observed sample in year l. Confidence intervals at the 95% level were computed using either the bootstrap or the Clopper-Pearson approach as customary in ICES WGBYC. Both were considered as the Clopper-Pearson approach is known for being more conservative: it produces confidence intervals that above the nominal level (i.e., wider than necessary) but generates non-nil confidence intervals even if no bycatch has been observed (Northridge et al., 2019). In practice, ICES WGBYC often pooled several years to stabilize the estimate of $\bar{p}$ (e.g., ICES 2018, p. 57–58; Carretta and Moore, 2014): Equation (6) translate an ideal case that is rarely met in practice.

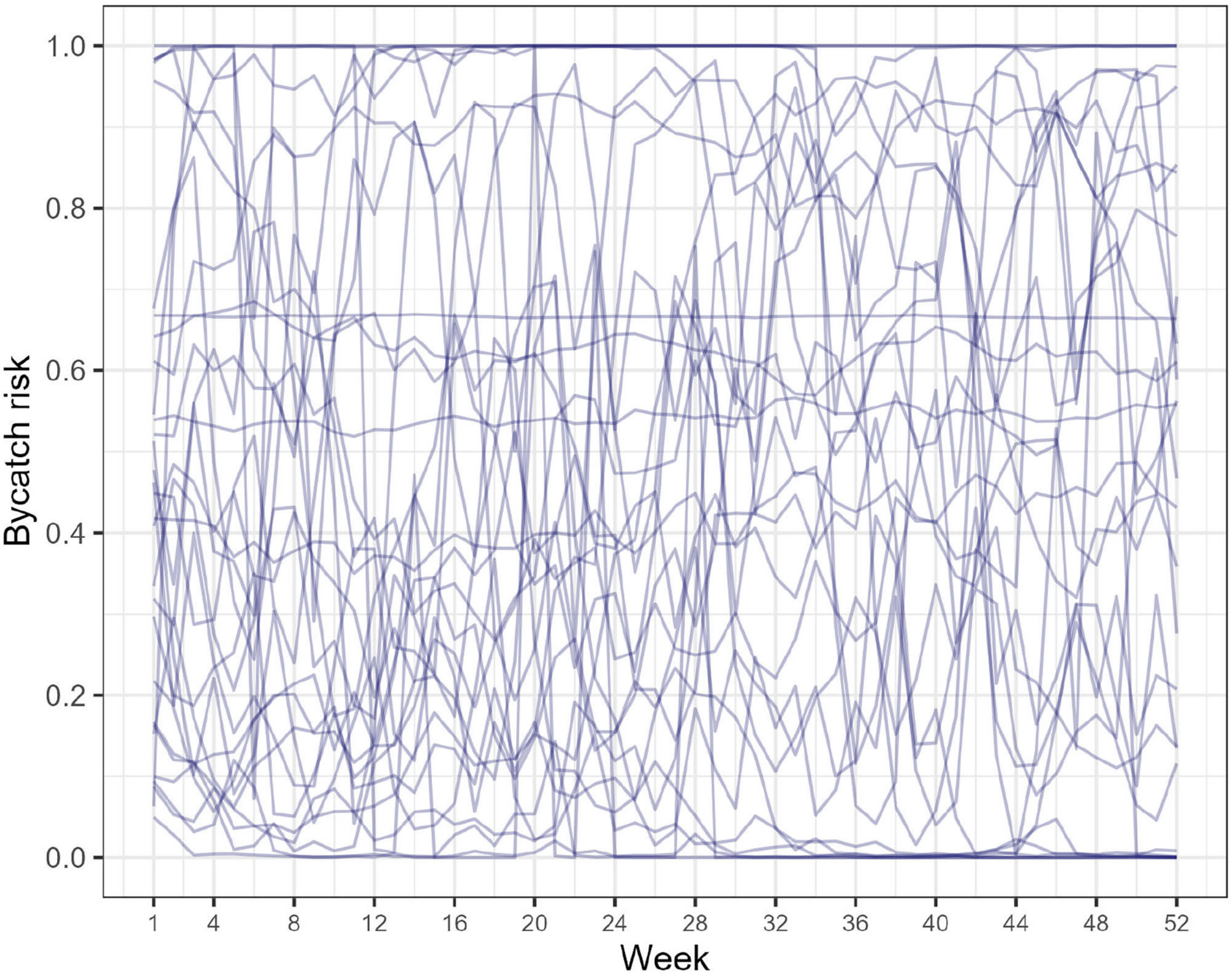


FIGURE 2 | Prior predictive checks *sensu* (Gabry et al., 2019). Bycatch risk (p_{ijkl} in Equation 1) is depicted: 30 random realizations from the priors are depicted.

ICES WGBYC usually works on bycatch rates (in number of PETS per unit effort), not bycatch risk. We focused on risk for simplicity, but scaling bycatch risk to a rate is straightforward by multiplying with the average number of PETS bycaught in a bycatch event. Dolphin presence was seasonal in the data-generating mechanism for simulations: pitching a method that can explicitly accommodate such seasonality against one that does not may be viewed as knocking down a strawman. However, current estimates of PETS bycatch in Europe are stratified by flag, ICES statistical areas, and métiers but not by season (e.g., Table 2 p. 17 in ICES 2019; Northridge et al., 2019, p. 27). The comparison remains relevant and topical as it matches current practices.

3. RESULTS

Convergence across all simulations and scenarios was assumed to be reached, with all $\hat{R} < 1.025$, for all parameters. For each simulation, chains were combined in a single sample of 2,000 values to approximate the joint posterior distribution of the model defined by Equations (1), (2), and (3).

3.1. Design- vs. Model-Based Approach

Comparing the design- and model-based approach was done with simulating 1 year of data. When data sampling was unbiased, both the design- and model-based approach were able to recover the true number of bycatch events (**Figure 3**; **Table 1**). Estimates of bycatch events were statistically unbiased but their precision low with a (frequentist 95%) confidence or (Bayesian 80%) credible interval (CI) as large as 100% of the point estimate (**Table 1**), as could be expected with only 15 bycatch events were recorded on average by onboard observers (**Table 1**). With under-sampling, design-based estimates were negatively biased (that is, they were under-estimates) whereas model-based estimates were still unbiased on average (**Figure 3**; **Table 1**). With over-sampling, design-based estimates were positively biased (that is, they were over-estimates) but so were model-based estimates, although bias was 5 times smaller (**Figure 3**; **Table 1**). In all cases, coverage was 100% but largely as a result of low precision:

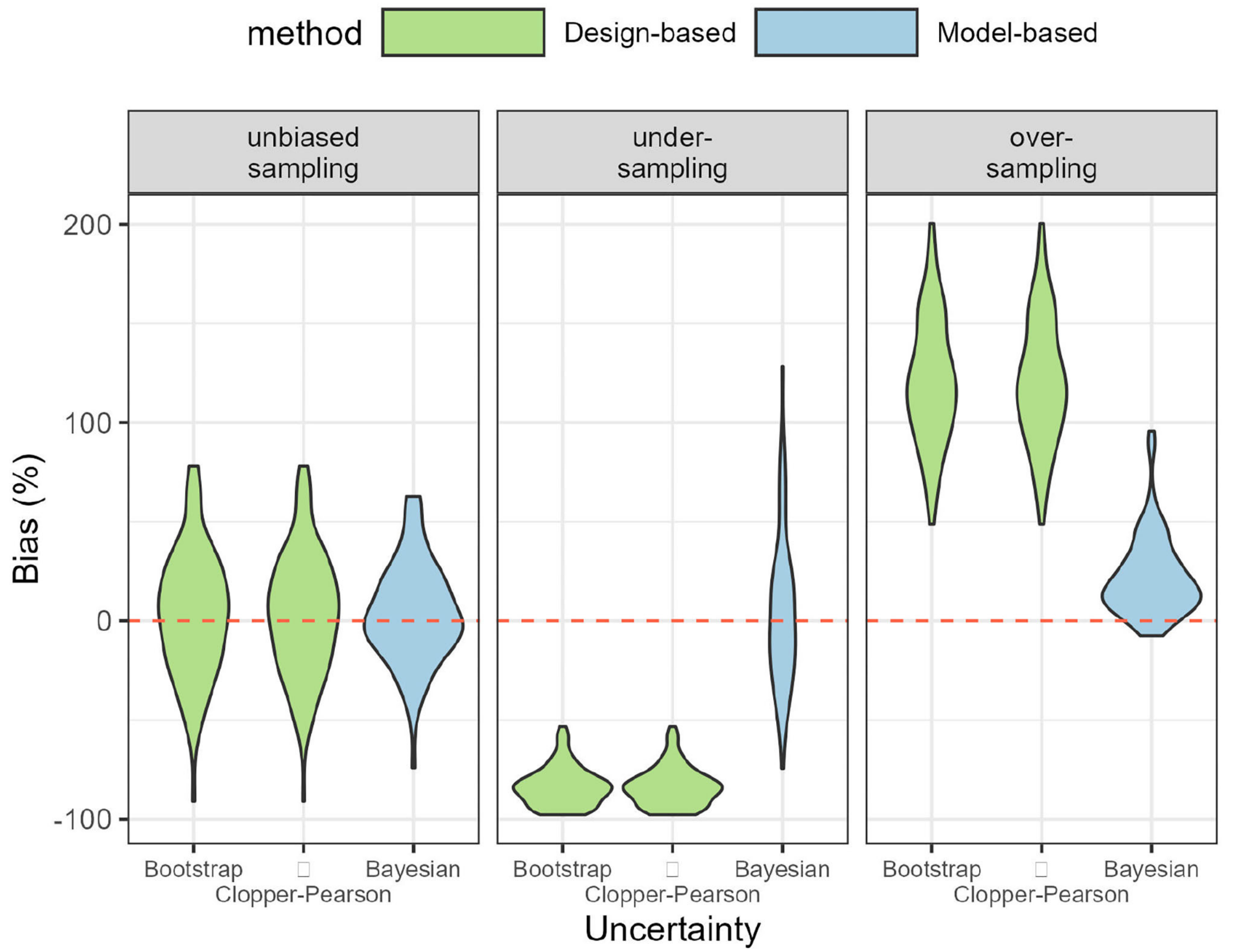


FIGURE 3 | Violin plot of bias in point estimates of total bycatch events. **Left:** data sampling was unbiased and all methods yielded statistically unbiased estimates. **Middle:** Under-sampling scenario: only the model-based approach was accurate. **Right:** Over-sampling scenario: both the design- and model-based approaches were biased upwards. Violin plots are based on 100 simulations.

TABLE 1 | Statistical properties of estimates from the design- and model-based approach.

Method	Uncertainty	Data sampling	n_{years}	Bias (%)	Coverage (%)	Width of CI (%)	n_{obs}
Design-based	Bootstrap	Unbiased	1	3.5	100.0	102.5	15
Design-based	Clopper-Pearson	Unbiased	1	3.5	100.0	115.0	15
Model-based	Bayesian	Unbiased	1	3.6	100.0	120.4	15
Design-based	Bootstrap	Under-	1	−83.5	100.0	195.0	5
Design-based	Clopper-Pearson	Under-	1	−83.5	100.0	259.6	5
Model-based	Bayesian	Under-	1	3.0	100.0	204.3	5
Design-based	Bootstrap	Over-	1	121.0	100.0	46.1	63
Design-based	Clopper-Pearson	Over-	1	121.0	100.0	50.1	63
Model-based	Bayesian	Over-	1	22.1	100.0	78.6	63

One year of data was simulated a 100 times. Bias of point estimate, coverage of (frequentist 95%) confidence or (Bayesian 80%) credible interval (CI) and precision (as CI width relative to the point estimate) are reported. The last column indicates the average number of bycatch events ($n_{\text{obs}} = \mathbb{E}\left[\sum_{ijk} y_{ijk}\right]$*) that were recorded by onboard observers during data sampling.*

precision was very low with CI spanning some 200% of the point estimate for the unbiased and under-sampling scenarios. This low precision was the result of having to work with as few as 5 observed bycatch events on average (**Table 1**). Precision improved with over-sampling, but was still as high as 50% of the point (over-)estimate. The model-based approach was well-calibrated in both the unbiased and under-sampling scenarios (**Figure 4**): model-based estimates were on average equal to the truth whereas this was only the case with design-based estimates when sampling was unbiased. In addition, the model-based approach was able to recover the temporal profile of bycatch risk (**Figure 5**) in these two scenarios, but with an increased accuracy and precision if sampling was unbiased. In the over-sampling scenario, both the design- and model-based approaches were not well -calibrated (**Figure 4**) and the model-based approach over-estimated bycatch risk when no data were collected (**Figures 1**, **5**).

3.2. Model-Based Approach With Several Years of Data

With several years of data, the model-based approach was able to yield nearly unbiased estimates: the bias was smaller than 3 bycatch events when sampling was unbiased, but as large as 10 (on average) with biased sampling and 3 years of data. The precision of estimates improved with several years of data, as expected with larger sample size. Precision of model-based estimates with over-sampling were already acceptable with 3 years of data: an 80% credible interval width of 50% corresponds to a coefficient of variation of $\frac{50}{2.5} \approx 20\%$ assuming a normal distribution for the posterior. The model-based approach allowed to obtain estimates at the weekly scale (**Figure 6**): these estimates were approximately unbiased in the unbiased and over-sampling scenarios, but were biased for the under-sampling scenario. In that latter case, the bias was correlated with the temporal pattern used to simulate dolphin presence (**Figure 1**): it was the largest when dolphin presence was at its highest but positive at the beginning of a year and negative at the end of the same year. Both biases were greatly attenuated with increased sample size.

4. DISCUSSION

Using Monte-Carlo simulations, we investigated the statistical properties of a model-based approach, regularized multilevel regression with post-stratification, to estimate the total number of bycatch events in a fishery operating year-round. Simulations were broadly informed from the case of common dolphins and pair-trawlers in the Bay of Biscay and from harbor porpoises and set-gillnets in Celtic Seas. A salient feature of simulations was biased sampling with observers being preferentially accepted onboard when bycatch risk was either high or low. Data simulations in that latter case, which is the most realistic one in the Bay of Biscay (Peltier et al., 2016), resulted in as few as 5 observed bycatch events per year on average (**Tables 1**, **2**). This aligns with the ubiquitous description of small cetacean bycatch being a rarely observed event. It was nevertheless possible to fit a regularized multilevel regression model on these data. Importantly, estimates from this model-based approach were statistically less biased than the design-based estimates when sampling was biased. Model-based estimates were, however, imprecise but this is largely to be expected (Amandè et al., 2012), especially with as few as 5 observed bycatch events per year. The design-based approach was also imprecise, even in the unbiased data sampling scenario of 5% coverage of the fleet, which is not reached in practice (Anonymous, 2016; ICES, 2020b). The design-based approach was very sensitive to how data were collected: this approach severely under- or over-estimated bycatch when sampling was biased, whereas the model-based approach was still well-calibrated with under-sampling, but not with over-sampling (**Figure 4**).

Biases in onboard observer data are pervasive and widely acknowledged (Babcock and Pikitch, 2003; Benoît and Allard, 2009; Peltier et al., 2016). Enforcing coverage as required to achieve a pre-specified precision in estimates can be challenging in practice. For example, in 2016, France only achieved a coverage rate less than 2% for most métiers and concluded on the impossibility of scaling-up observed bycatch rates to the whole fleet (Anonymous, 2016, p. 24). There were, however, 9 bycatch events of common dolphins in pair-trawlers targeting European hake (*Merluccius merluccius*). From these numbers, bycatch was described a "rare" event (Anonymous 2016, p. 23). Such a conclusion would be warranted if sampling were representative, in which case the design-based estimate could be used, even though its precision would still be very low. On the other hand, with under-sampling, this conclusion is misleading as our simulations further illustrated: although only 5 bycatch events were observed on average (**Figure 4**), the true number of bycatch events was on average 60 times larger (**Figure 4**). In our simulations, the true bycatch rate was on average ≈ 3% over a year, which is not rare, but not frequent either. Moreover, interviews with French skippers deploying trawls or gillnets in the Bay of Biscay revealed that more than 80% of respondents declared to having experienced at least one small cetacean bycatch event in a year (Cloâtre, 2020). Such a large proportion contradicts the idea of common dolphin bycatch being a rare event in the Bay of Biscay, but rather suggest severe biases in onboard observer data that result in the rare reporting of bycatch events, rather than a rarity of events *per se*. The common dolphin in the Bay of Biscay illustrates how under-sampling may distort the perception of bycatch as a very rare event when it can, in fact, be widespread. This is a catch-22 situation whereby cetacean bycatch is described as a rare event because it is rarely reported, and this perceived rarity may serve to argue against ambitious dedicated monitoring programmes out of cost-effective considerations, thereby preventing to dispel the initial misconception.

Finding an optimal sampling plan for fisheries with rare bycatch events is a long standing problem (ICES, 2009). Several strategies have been attempted: for example in the United States, one strategy is "pulsed sampling" whereby a particular fishery or métier is very heavily sampled for a short period of time in order to maximize the chance for observers to record any bycatch that might occur (ICES, 2009). This pulsed sampling strategy corresponds to our over-sampling scenario wherein monitoring effort is positively correlated with bycatch

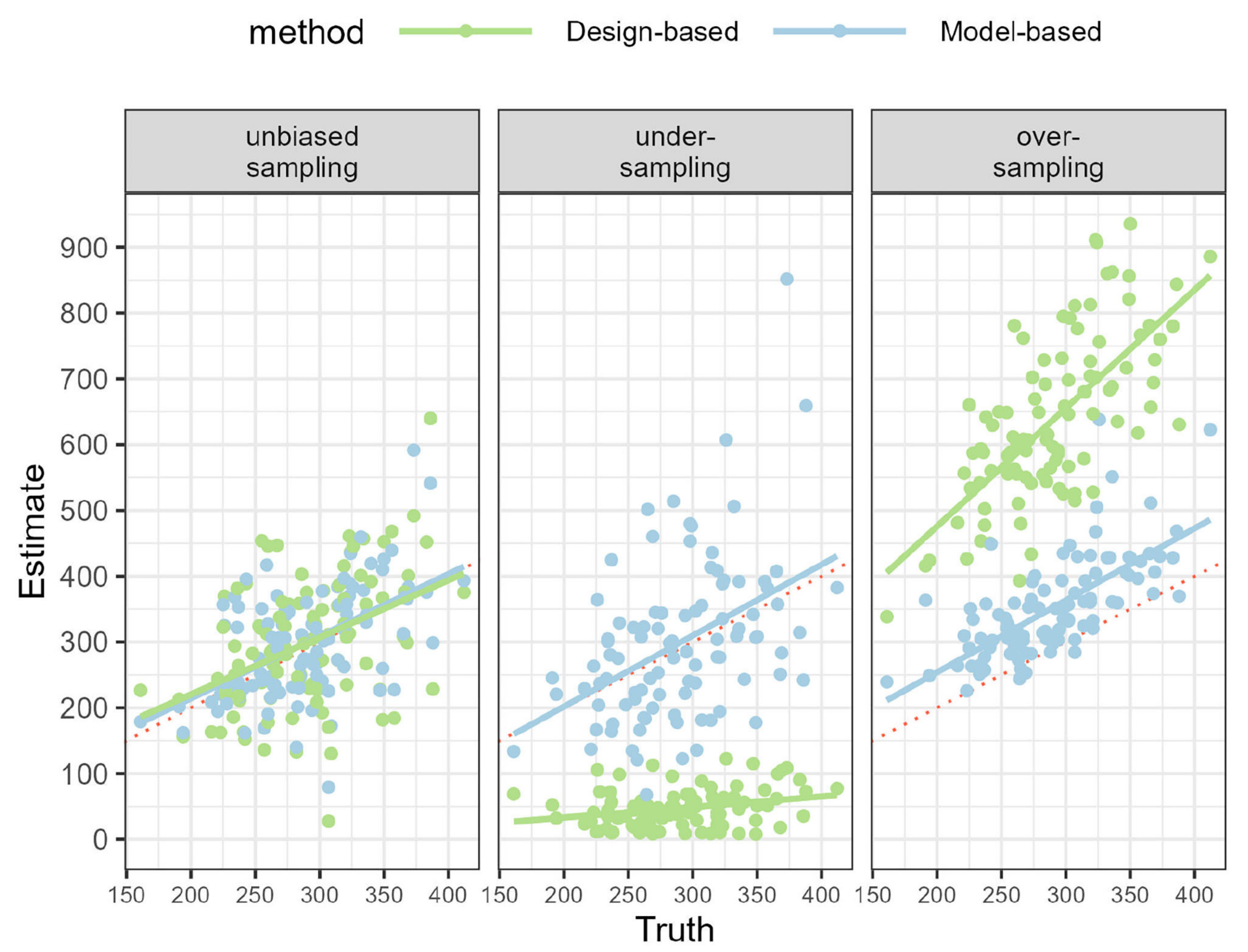


FIGURE 4 | Regression lines of point estimates against the true number of bycatch events, showing the calibration of the design- and model-based approach. The *x*-axis shows the true number of bycatch events across 100 simulations, spanning between 150 and 400 events. The red dotted line shows the identity line, i.e., no bias. **Left:** data sampling was unbiased and all methods yielded statistically unbiased estimates. **Middle:** Under-sampling scenario: only the model-based approach was well-calibrated. **Right:** Over-sampling scenario: both the design- and model-based approaches were not calibrated to the truth.

risk. Under this scenario, the absence of any sampling at all when bycatch risk was low was detrimental to the accurate estimation of bycatch events with our model. Model-based estimates were, however, less biased than design-based estimates. Arguably, this comparison is somewhat artificial as a correct comparison would use all the available information and uses estimators that are season-specific to account for under-sampling when bycatch risk is low if such a period is known to the investigator. Notwithstanding this shortcoming, model-based estimates represented an improvement and allowed to infer the bycatch risk profile accurately, especially with several years of data.

We showed with our Monte-Carlo simulations that regularized multilevel regression with post-stratification can nevertheless be used to analyze bycatch data despite concerns about non-representative sampling. Model-based approaches (Palka and Rossman, 2001), with post-stratification (Lennert et al., 1994), or machine learning (Carretta et al., 2017), or multilevel regression (Sims et al., 2008; Martin et al., 2015) have previously been used to estimate bycatch rates. Traditional, design-based, ratio estimates are biased if sampling is biased; imprecise if observer coverage is low (as is the usual case in the North East Atlantic; see for example Figure 14, p. 114 in ICES, 2020b); and volatile if bycatch events are only observed occasionally (Carretta et al., 2017). The traditional remedy to stabilize estimates and improve precision is to bypass year-specific estimation and pool several years together (Carretta and Moore, 2014; ICES, 2018). This pragmatic solution improves precision but does not address the problem of biased sampling. It also introduces estimation bias for any year-specific estimates by pooling completely several years in order to stabilize the variance of estimates (ICES, 2009, p. 36): any between-year differences are thus ignored in order to obtain a better precision of estimates. It is a reasonable approach in practice, but one

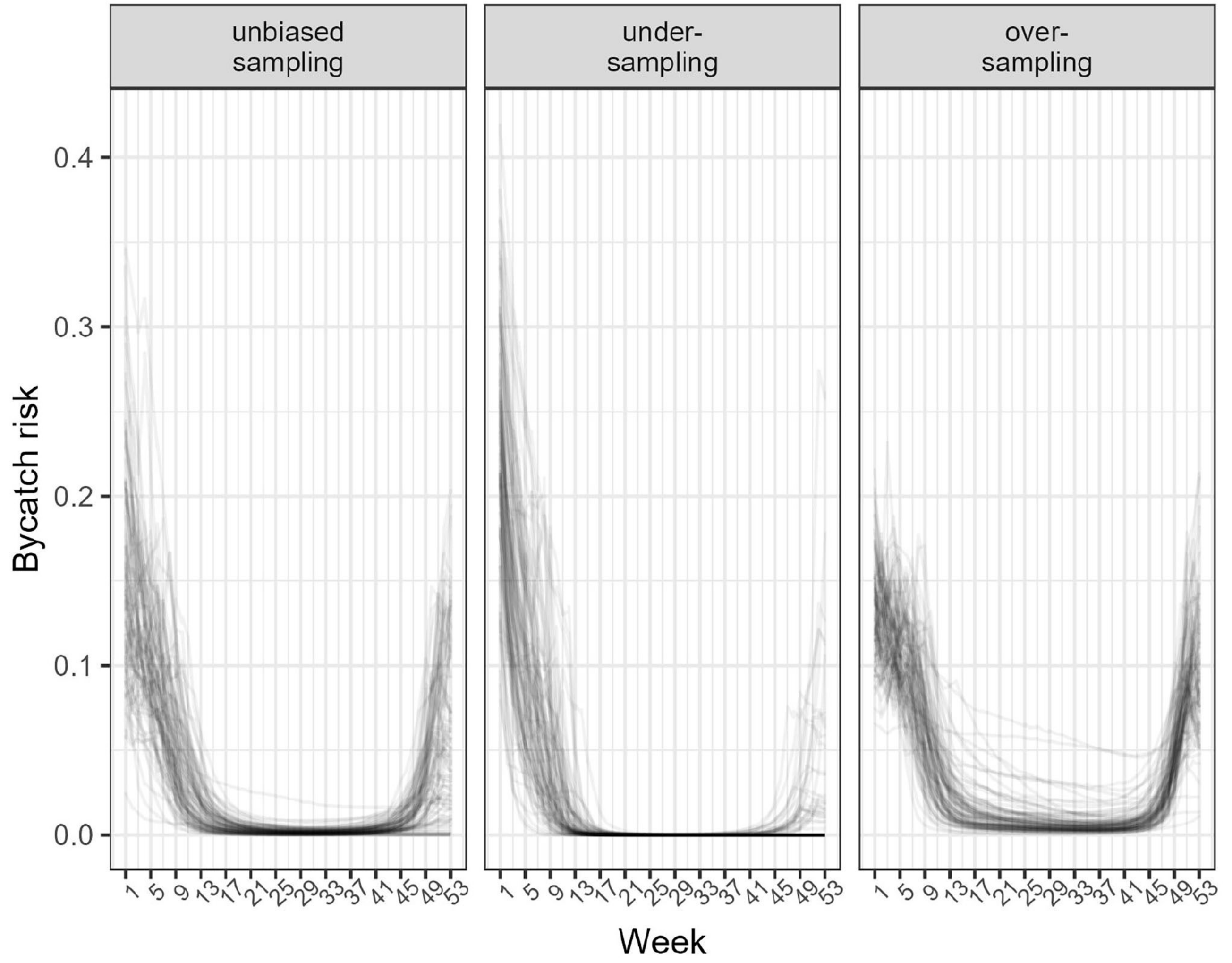


FIGURE 5 | Estimated temporal pattern in mean bycatch risk from the model-based approach. **Left:** data sampling was unbiased. **Middle:** Under-sampling. **Right:** Over-sampling. The model-based approach recovered the correct pattern overall, but overestimated risk in the over-sampling scenarios when risk was, in fact, nil but no data were collected.

that can be improved. Model-based approaches offer a trade-off between no-pooling (keeping all years separate) and complete-pooling with a third option: partial pooling or regularization (Gelman and Shalizi, 2013). Regularization is a general term for statistical procedures that give more stable estimates. Our model-based approach achieves regularization by leveraging, via a structured prior model (Equations 2 and 3, see section 2), the within-year information at the weekly scale. The result were more stable and accurate annual bycatch estimates at the cost of some modeling assumptions and weakly-informative priors. Importantly, weekly estimates could also be obtained with our model-based approach.

Our model-based approach is semi-parametric as it uses a random walk prior to learn from the data the weekly pattern in bycatch risk. This prior is also ensuring some smoothness in the temporal risk profile as it translates an assumption on the correlation between 2 consecutive weeks. This random walk model remains simple as the order is fixed to 1. We further expanded this model to allow for between-years variation in the weekly risk profile with a Gaussian Process prior (Neal, 1998; Goldin and Purse, 2016). Importantly, these two prior choices (a random walk and a Gaussian Process prior) add structure to the model and help in leveraging the information present in the sparse data typical of onboard observer programmes. Even when with over-sampling, these choices were not detrimental as model-based estimates were statistically unbiased and precise with 3 years of data (**Table 2**). The explicit consideration of time effects is key to mitigate bias in sampling. In our simulations, dolphin presence was caricaturally seasonal, and observers could be preferentially allowed on fishing vessels when dolphins were less or more likely to be present (**Figure 1**). Our model was still able to provide statistically unbiased estimates of bycatch in those scenarios, although these estimates were very imprecise with under-sampling. However, they were not

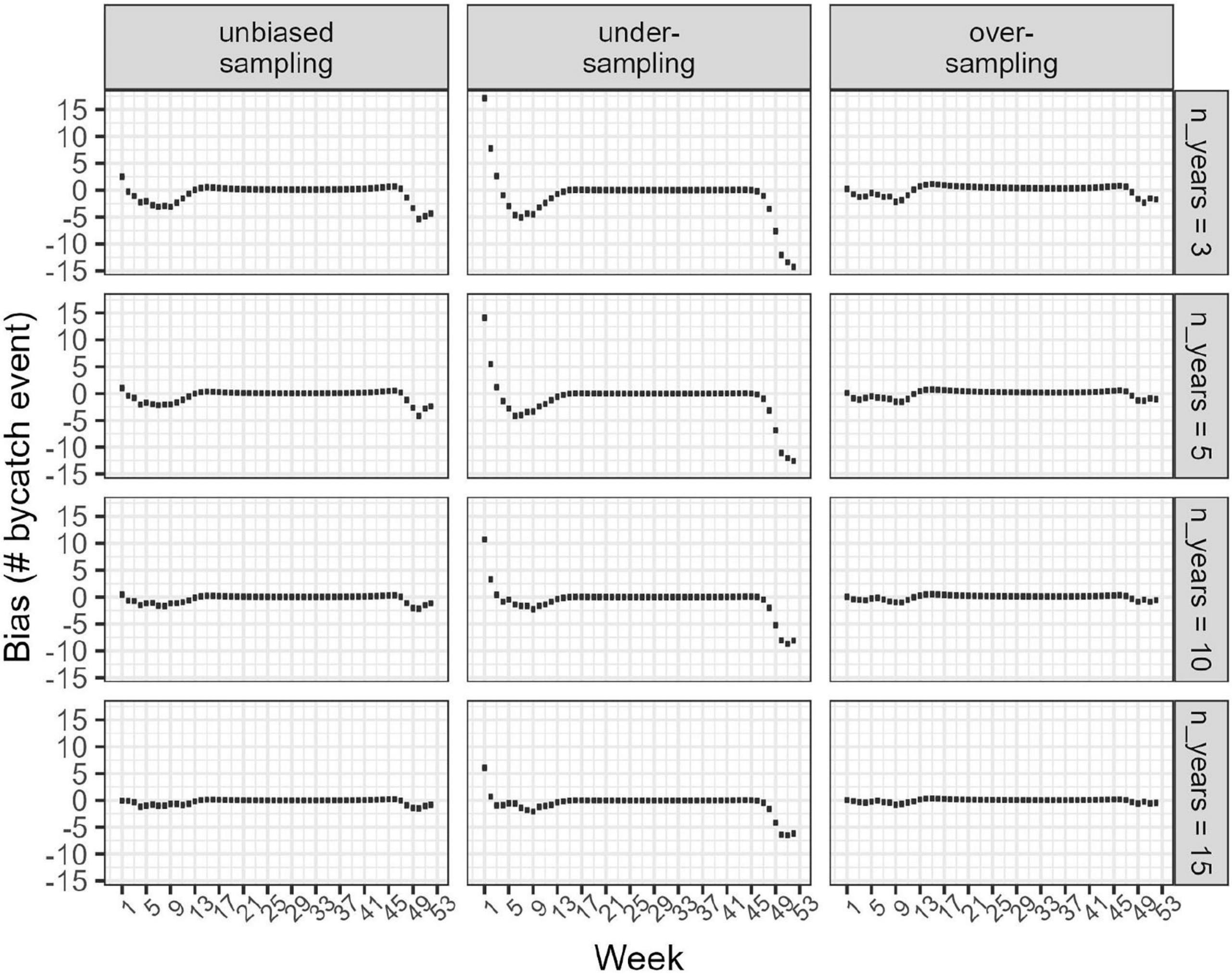


FIGURE 6 | Box plots of bias (in number of estimated bycatch events compared to the truth) in the weekly model-based estimates of bycatch events. **Left:** data sampling was unbiased. **Middle:** Under-sampling. **Right:** Over-sampling. Each row corresponds to data simulated for a different number of years.

more imprecise than the traditional (but biased) design-based estimates (**Table 1**) if 80% credible interval were used. In addition to being unbiased, these estimates could also reveal with accuracy the temporal risk profile (**Figure 5**). It is important to keep in mind here that our model is different from the data-generating model used in simulating data: our results were not simply an instance of using a true model, which is impossible in practice as a model is by definition a simplification used to capture the salient features of a phenomenon. Our model had some shortcomings: for example, bias increased with 3 years of data compared to 1 year for the under-sampling scenario (contrast **Tables 1**, **2**). This increased bias (toward the prior model) was the result of partial pooling but came with a gain in precision as evidenced in the width of credible intervals. The bias progressively wore off with more years of data, illustrating thereby the attractiveness of partial pooling and structured priors to regularize estimates (Gelman and Shalizi, 2013; Gao et al., 2019). The gain in reducing bias in estimates and increasing their precision was most evident with over-sampling (**Tables 1**, **2**).

Our model could also provide weekly bycatch estimates which were largely unbiased except in the under-sampling scenario where a positive and negative bias remained at the beginning and end of a year respectively, even with 15 years of data (**Figure 6**). With under-sampling, few observed bycatch events can be collected by design because observers are very unlikely to be accepted on board by skippers. Weekly estimates were too high at the beginning of a year but too low at the end, but this somewhat canceled out at the year-level. There was still a slight overestimation bias resulting from our choice of a non-symmetric pattern for dolphin presence and a symmetric pattern for biased coverage: observing bycatch events at the end of a year was comparatively more difficult than at the beginning of a year because overlap between a non-nil coverage and dolphin presence was smaller at the end of year (**Figure 1**). These shortcomings illustrate that a model-based approach should

TABLE 2 | Statistical properties of estimates from the model-based approach.

Method	Uncertainty	Data sampling	n_{years}	Bias (bycatch events)	Coverage (%)	Width of CI (%)	n_{obs}
Model-based	Bayesian	Unbiased	3	3.0	100.0	91.1	45
Model-based	Bayesian	Unbiased	5	2.1	100.0	76.3	75
Model-based	Bayesian	Unbiased	10	1.1	100.0	59.1	150
Model-based	Bayesian	Unbiased	15	1.9	100.0	50.9	225
Model-based	Bayesian	Under-	3	10.0	100.0	164.6	15
Model-based	Bayesian	Under-	5	6.4	100.0	142.0	25
Model-based	Bayesian	Under-	10	8.3	100.0	112.9	50
Model-based	Bayesian	Under-	15	5.3	100.0	97.8	75
Model-based	Bayesian	Over-	3	7.4	100.0	53.2	63
Model-based	Bayesian	Over-	5	4.8	100.0	42.6	126
Model-based	Bayesian	Over-	10	3.5	100.0	32.6	630
Model-based	Bayesian	Over-	15	3.3	100.0	27.7	756

Several years of data were simulated a 100 *times. Bias of point estimate (in number of bycatch events), coverage of (Bayesian 80%) credible interval (CI) and precision (as CI width relative to the point estimate) are reported. The last column indicates the average number of bycatch events (*$n_{\text{obs}} = \mathbb{E}\left[\sum_{ijkl} y_{ijkl}\right]$*) that were recorded by onboard observers during data sampling.*

be tailored to the context of the study, and we designed our simulations largely from our knowledge on the common dolphin in the Bay of Biscay. However, the framework of regularized multilevel regression with post-stratification is very flexible and we believe our proposed model has large potential for generality as it simply translates a decomposition of bycatch risk into a smooth time-varying and (unstructured) time-invariant effects. The model can easily be made more complex, data permitting, to accommodate spatial effects with, for example, a Besag-type prior (Sims et al., 2008; Morris et al., 2019).

Several important assumptions are structurally built into our model: in particular, a first order random was assumed for the mean function of the Gaussian Process prior, with no attempt to estimate from data the correlation parameter (e.g., using an AR(1) prior instead). The choice of a first order random walk was not aiming at uncovering the true data-mechanism: our aim were to reveal a temporal pattern in bycatch risk from sparse data using a flexible, yet parsimonious approach. This was particularly true in the under-sampling scenario where few bycatch events could be observed in any given year of simulated data. In the other scenarios, other choices than the first order random walk could be considered as more data are collected. We also assumed that the range parameter of the covariance function in the Gaussian Process prior for week effects was known and such that bycatch risk was temporally uncorrelated after 4 weeks. Fixing the range parameter is usually not recommended but was motivated by consideration of the data-to-parameter ratio, and computation convenience. Bycatch data are binary and can be sparse: these two features underscore how little information may be available. In this context, limiting the number of parameters to estimate can be justified on pragmatic consideration. The model we are proposing is parameter-rich, but some structure are assumed on these parameters in the form of the prior used. These priors represent choices from the analyst and may be reconsidered and tested, data permitting. There was some evidence that bycatch risk was under-smoothed in the over-sampling scenario which resulted in an over-estimation of bycatch risk (**Figure 5**, rightmost panel). Model expansion is seamless with `Stan` (Gabry et al., 2019), and the above mentioned parameters could be estimated, rather than fixed, with adequate data. Despite somewhat arbitrary prior and modeling choices, our model provided more accurate estimates of bycatch numbers and bycatch risk in under- and over-sampling scenarios. This satisfactory predictive ability points to another important limitation.

Our model is phenomenological, i.e., it is agnostic of the causes behind the temporal variations in bycatch risk. Bycatch risk is the product of dolphin presence and bycatch probability given presence (the latter was constant in our simulations). The model only estimates this product of two probabilities and thus cannot disentangle them without other sources of data. This limitation seems inconsequential in our simulations for the aim of accurate estimation of the total number of bycatch events as interest lies in the effects of causes (how much bycatch?) rather than in the causes of effects (why bycatch occurred?). A straightforward model expansion (as pointed out by a reviewer) would be the consideration of p vessel-level covariates $(z_{1j}, \ldots, z_{pj})$ in Equation (1):

$$\alpha_j \sim \mathcal{N}\left(\sum_{b=1}^{p} \left(\xi_b \times z_{jb}\right), \sigma_{\text{vessel}}\right) \tag{7}$$

Candidate covariates such as vessel length or gear-attributes (e.g., mesh size) could be incorporated in the analysis to improve the exchangeability assumption on vessel-effects. An obvious covariate to consider for detecting self-selection of skippers into observer programme participation is to include whether a skipper has ever accepted an observer, or the number of times it did so in the past: a negative regression coefficient could be interpreted as voluntary skippers having an intrinsically lower risk of bycatch. Including skipper-level covariates could reduce the between-skipper variance σ^2_{vessel}, and improve ultimately precision of bycatch estimates. Consideration of other distributions than the normal (e.g., a skew-normal, or a Student-t distribution with a fixed degree of freedom) would be straightforward with `Stan` but is probably worthwhile only with large enough amount of data for all practical purposes (McCulloch and Neuhaus, 2011).

An important assumption underlying accurate estimation is that the information on the total effort must also be accurate and available at the scale of weeks for post-stratification. This assumption is crucial to scale-up estimates from the (potentially biased) sample to the population, but it does not necessarily hold with fisheries effort as the latter is more often estimated rather than measured directly (Julian and Beeson, 1998; ICES, 2018, 2020b). Here we assumed that the total number of fishing operations (e.g., number of tows for trawls; Tremblay-Boyer and Berkenbusch, 2020) are available as auxiliary information for post-stratification. This assumption about the availability of disaggregated data stems from the explicit consideration of time as an important predictor of variations in bycatch risk. This assumption is necessary for using post-stratification to align the sample with the population targets but may be difficult to meet in practice. Currently, ICES WGBYC uses in its BRA a coarse, but admittedly comparable proxy across fisheries and countries to quantify fishing effort, namely days at sea (ICES, 2019). A day at sea is any continuous period of 24 h (or part thereof) during which a vessel is present within an area and absent from port (Anonymous, 2019a). Importantly, this definition is not at the level of a fishing operation, and effort thus quantified is already aggregated at a level above that at which bycatch data are collected. This coarsening of fisheries effort data is fundamentally a measurement problem, and one that modeling should not be expected to remedy easily. BRA uses an estimate of total fishing effort for the fisheries of concern in a specific region, together with some estimate of likely or possible bycatch rates that might apply for the species of concern, in order to evaluate whether or not the total bycatch in that area might be a conservation issue. A regularized multilevel regression model could be used to obtain estimates of bycatch rates to be used in BRA. Post-stratification could also be attempted using the coarse days at sea proxy for effort, and thus our framework could be adapted to match the requirements of ICES WGBYC.

Assuming that our framework were to be adopted to produce bycatch estimates, how would both fisheries and Non-Governmental Organizations (NGOs) react given the salience of bycatch as a policy issue in Europe? Such a prospective question inevitably entails some speculations (as with all "what-if" questions), but may nevertheless bring some insights as highlighted by a reviewer. Within Europe, the conservation reference currently available for assessing bycatch is that established under the Agreement on the Conservation of Small Cetaceans of the Baltic, North East Atlantic, Irish and North Seas. The agreement has the conservation objective to minimize anthropogenic removals of harbor porpoises (and other small-sized cetaceans), and to restore and/or maintain population depletion to/at 80% or more of the carrying capacity in each assessment unit (ASCOBANS, 2000; ICES, 2020c). Methods for setting conservation reference points were agreed in March 2021 at the meeting of the Biodiversity Committee of the Olso-Paris Regional Sea Convention. This committee adopted the use of the Removals Limit Algorithm for harbor porpoises in the North Sea assessment unit and a modified Potential Biological Removal (Wade, 1998) for common dolphins in the North-East Atlantic (Genu et al.)[3]. Accurate bycatch estimates will be needed for assessment against these reference points. However, fisheries may challenge the accuracy of estimates precisely because they will result from a new statistical model. While a healthy skepticism is warranted, and model improvements are certainly possible, it must be kept in mind that our model only addresses the issue of having a correlation between observer coverage and bycatch risk, and does so with some assumptions. There would remain many biases to be addressed in bycatch data (Babcock and Pikitch, 2003), and many of them would be best addressed with a proper random allocation of professional observers to vessels (that is better design and better measurement). A purely model-based solution can be brittle (Sarewitz, 1999), and may lead to displacement of the problem of bycatch assessment to a never-ending problem of model improvement that would delay any corrective measures or decision (Rayner, 2012). Model-based estimates offer a pragmatic approach to the analysis of already collected data, but should not deflect from improving survey design where possible. Assuming that model-based estimates would be endorsed by a fishery industry, NGOs could challenge in court any reference point that is not zero for PETS, since by definition, it *ought* to be zero. The Habitats Directive requires strict protection and prohibits "all forms of *deliberate* capture or killing" (emphasis added) of all species listed on its Annex IV which includes all cetacean species. The Court of Justice of the European Union has consistently ruled that the adjective "deliberate" is to be understood in the sense of "conscious acceptance of consequences" (Trouwborst and Somsen, 2019): in other words, using knowingly a gear that may potentially catch a protected species contravenes the Habitat Directives. What will eventually play out remains to be seen, but strongly hinges on how polarized the bycatch issue is. As scientists, our duty remains to provide the best available evidence on bycatch and to outline all management actions and their consequences in light of this evidence (Pielke, 2007). Our model is unlikely to change bycatch management in France in the near term: both fisheries and NGOs are at loggerheads, vying for public and official support. They are building constituencies and advertising unyielding positions in diverse medias: we content that a legal confrontation at a national or supra-national level is extremely likely and probably being prepared. We nevertheless think our model, by making use of data already collected within the DCF framework and by encouraging further, ideally dedicated, monitoring; can be part of

---

[3] Genu, M., Gilles, A., Hammond, P., Macleod, K., Paillé, J., Paradinas, I. A., et al. (in preparation). Evaluating strategies for managing anthropogenic mortality on marine mammals: an R Implementation with the Package RLA.

a messy solution to the wicked problem (Frame, 2008) of dolphin bycatch in the medium to long term, once the gavel hits and the dust settles.

5. CONCLUSION

We investigated with simulations the ability of multilevel regularized regression with post-stratification to estimate cetacean bycatch for observer programmes when coverage is correlated to bycatch risk. Our aims were to provide a first investigation on model-based estimates obtained from samples preferentially collected either during low- or high-bycatch risk periods. The unbiased sampling case is unrealistic (Babcock and Pikitch, 2003): biased sampling, either under-sampling or over-sampling (ICES, 2009), may be the general case. We considered both of these cases, under quite extreme scenarios whereby data collection was highly correlated with bycatch risk, resulting in either very few observed events with under-sampling, and a large number of observed events with over-sampling. In both cases, multilevel regularized regression with post-stratification was able to produce nearly unbiased bycatch estimates with as few as 5 observed events data. With only 1 year of data, precision was low, especially with under-sampling, and there was some estimation bias with over-sampling one. These results stemmed from the extreme scenarios we considered but illustrate nevertheless that a model cannot be expected to solve all the deficiencies of data collection and measurement. Good measurement is key for accurate estimation and our results actually re-emphasize the importance of design. However, they also show that a good data collection design *and* an adequate modeling framework are synergistic and allow to extract a lot of information for sparse data. Assuming a normal distribution for the bycatch estimates (which is not necessary as the posterior is available, but the following are back-of-the-envelope calculations to be used for deriving heuristics), a 80% Bayesian CI width divided by 2.5 gives an idea of the associated coefficient of variation: the model-based approach can yield a coefficient of variation of 50% with as few as 15 observed events if sampling is unbiased. With under-sampling, one would need 10 years of data (under our data simulation schemes) to obtain the same precision. This re-iterates the need to (i) have dedicated observer schemes, (ii) ensure adequate observer coverage and (iii) use a model-based approach tailored to extract as much information as possible from sparse data, as the first two points are very difficult to live up to in practice.

The key assumptions behind regularized multilevel regression with post-stratification in our simulations are that bycatch risk changes smoothly through time and that accurate data on the number of fishing operations at the same temporal scale are available (e.g., number of tows for trawls; Tremblay-Boyer and Berkenbusch, 2020). When both assumptions can be reasonably entertained, we showed how a model-based approach using recent methodological developments is attractive, irrespective of how data were collected. A further asset of the explicit consideration of a temporal scale is that it may help in pinpointing more precisely windows of heightened risk in order to target adequate mitigation measures (e.g., spatio-temporal closures). The framework of multilevel modeling is very flexible and can accommodate spatial effects, etc., data permitting. Regularization will, in general, be needed to mitigate data sparsity and leverage partial pooling in order to obtain stable estimates of bycatch. Given the satisfactory performance of regularized multilevel regression with post-stratification in our simulations, we recommend further investigations using this technique to estimate bycatch rate and numbers from both representative or non-representative samples. The modeling choices we made (e.g., a first order random walk for the mean function, or fixing the range parameter in the covariance function of the Gaussian Process prior) are not prescriptive, and other choices of prior models for parameters should be investigated. Investigations should be tailored to the context, and modeling choices motivated by the latter: given the complexity of PETS bycatch, a one-size-fits-all solution is unlikely. A re-analysis of >15 years of observer data on common dolphin bycatch in pair trawlers flying the French flag is currently underway (Rouby et al.)[4] in order to obtain better bycatch estimates that could be further used to estimate conservation reference points in order to better manage this fishery in the long run (Cooke, 1999; Punt et al., 2021).

DATA AVAILABILITY STATEMENT

The datasets presented in this study can be found in online repositories. The names of the repository/repositories and accession number(s) can be found below: https://gitlab.univ-lr.fr/mauthier/regularized_bycatch.

AUTHOR CONTRIBUTIONS

MA: led the analyses, the conception, and writing of the paper. ER and KM: support in analyses, paper conception, and writing. All authors contributed to the article and approved the submitted version.

FUNDING

ADERA and JNCC provided support in the form of salaries for MA and KM, respectively, but did not have any additional role in the study design, data collection and analysis, decision to publish, or preparation of the manuscript.

ACKNOWLEDGMENTS

We thank two reviewers from critical and constructive comments that help in improving the present manuscript. MA thanks Amaterasu for support when carrying the simulations.

---

[4] Rouby, E., Dubroca, L., Cloâtre, T., Demanèche, S., Genu, M., Macleod, K., et al. (in preparation). Estimating cetacean bycatch from non-representative samples (II): a Case Study on Common Dolphins in the Bay of Biscay.

REFERENCES

Amandè, M. J., Chassot, E., Chavance, P., Murua, H., de Molina, A. D., and Bez, N. (2012). Precision in bycatch estimates: the case of Tuna Purse-Seine Fisheries in the Indian Ocean. *ICES J. Mar. Sci.* 69, 1501–1510. doi: 10.1093/icesjms/fss106

Anonymous (2016). *Rapport Annuel sur la mise en øeuvre du Réglement Européen (CE) 812/2004 Établissant les Mesures Relatives Aux Captures Accidentelles de Cétacés dans les Pêcheries*. Rapport annuel réglementaire. Technical report, Direction des pêches maritimes et de l'aquaculture (DPMA).

Anonymous (2019a). *Commission Implementing Decision (EU) 2019/910. Establishing the Multiannual Union Programme for the Collection and Management of Biological, Environmental, Technical and Socioeconomic Data in the Fisheries and Aquaculture Sectors*. European Commission.

Anonymous (2019b). *Regulation (EU) 2019/1241 of the European Parliament*. The Council of 20 June 2019 on the Conservation of Fisheries Resources and the Protection of Marine Ecosystems Through Technical Measures.

ASCOBANS (2000). Annex O - report of the IWC-ASCOBANS Working Group on Harbour Porpoises. *J. Cetacean Res. Manage.* 2, 297–305.

Authier, M., Aubry, L., and Cam, E. (2017a). Wolf in sheep's clothing: model misspecification undermines tests of the neutral theory for life histories. *Ecol. Evol.* 7, 3348–3361. doi: 10.1002/ece3.2874

Authier, M., Saraux, C., and Péron, C. (2017b). Variable selection and accurate predictions in habitat modelling: a shrinkage approach. *Ecography* 40, 549–560. doi: 10.1111/ecog.01633

Avila, I. C., Kaschner, K., and Dormann, C. F. (2018). Current global risks to marine mammals: taking stock of the threats. *Biol. Conserv.* 221, 44–58. doi: 10.1016/j.biocon.2018.02.021

Babcock, E., and Pikitch, E. K. (2003). *How Much Observer Coverage is Enough to Adequately Estimate Bycatch*? Technical report, Pew Institute for Ocean Science.

Benoît, H., and Allard, J. (2009). Can the data from at-sea observer surveys be used to make general inference about catch composition and discards? *Can. J. Fish. Aquat. Sci.* 66, 2025–2039. doi: 10.1139/F09-116

Brownell, R. J., Reeves, R., Read, A., Smith, B., Thomas, P., Ralls, K., et al. (2019). Bycatch in gillnet fisheries threatens critically endangered small cetaceans and other aquatic megafauna. *Endangered Species Res.* 40, 285–296. doi: 10.3354/esr00994

Cam, E. (2012). "Each site has its own survival probability, but information is borrowed across sites to tell us about survival in each site": random effects models as means of borrowing strength in survival studies of wild vertebrates. *Anim. Conserv.* 15, 129–132. doi: 10.1111/j.1469-1795.2012.00533.x

Carpenter, B., Gelman, A., Hoffman, M. D., Lee, D., Goodrich, B., Betancourt, M., et al. (2017). `Stan`: a probabilistic programming language. *J. Stat. Softw.* 76, 1–32. doi: 10.18637/jss.v076.i01

Carretta, J. V., and Moore, J. E. (2014). *Recommendations for Pooling Annual Bycatch Estimates when Events are Rare*. Technical report. NOAA-TMNMFS-SWFSC-528, NOAA. NOAA Technical Memorandum.

Carretta, J. V., Moore, J. E., and Forney, K. A. (2017). *Regression Tree and Ratio Estimates of Marine Mammal, Sea Turtle, and Seabird Bycatch in the California Drift Fishery: 1990-2015*. Technical Report NOAA-TM-NMFS-SWFSC-568, NOAA. NOAA Technical Memorandum.

Cloâtre, T. (2020). *Projet LICADO - Limitation des Captures Accidentelles de Dauphins Communs dans le Golfe de Gascogne. Bilan des Enquêtes auprés des Professionnels du Golfe de Gascogne* (Paris).

Cooke, J. G. (1999). Improvement of fishery-management advice through simulation testing of harvest algorithms. *ICES J. Mar. Sci.* 56, 797–810. doi: 10.1006/jmsc.1999.0552

Course, G. P., Pierre, J., and Howell, B. K. (2020). *What's in the Net? Using Camera Technology to Monitor, and Support Mitigation of, Wildlife Bycatch in Fisheries*. Technical report, Worldwide Wildlife Fund.

Dolman, S., Evans, P., Ritter, F., Simmonds, M., and Swabe, J. (2020). Implications of new technical measures regulation for cetacean bycatch in European waters. *Mar. Policy* 124:104320. doi: 10.1016/j.marpol.2020.104320

Downes, M., Gurrin, L. C., English, D. R., Pirkis, J., Currier, D., Spittal, M. J., et al. (2018). Multilevel regression and poststratification: a modeling approach to estimatingpopulation quantities from highly selected survey samples. *Am. J. Epidemiol.* 187, 1780–1790. doi: 10.1093/aje/kwy070

Fay, R., and Herriot, R. (1979). Estimates of income for small place: an application of james-stein procedures to census data. *J. Am. Stat. Assoc.* 74, 269–277. doi: 10.1080/01621459.1979.10482505

Forsberg, O. J. (2020). Polls and the US presidential election: real or fake? *Significance* 17, 6–7. doi: 10.1111/1740-9713.01437

Frame, B. (2008). 'Wicked', 'Messy', and 'Clumsy': long-term frameworks for sustainability. *Environ. Plan C Govern. Policy* 26, 1113–1128. doi: 10.1068/c0790s

Gabry, J., Simpson, D., Vehtari, A., Betancourt, M., and Gelman, A. (2019). Visualization in Bayesian workflow. *J. R. Stat. Soc. Ser. A* 182, 389–402. doi: 10.1111/rssa.12378

Gao, Y., Kennedy, L., Simpson, D., and Gelman, A. (2019). *Improving Multilevel Regression and Poststratification with Structured Priors*. Technical report, University of Toronto & Columbia University.

Gelman, A. (2007). Struggles with survey weighting and regression modelling. *Stat. Sci.* 22, 153–164. doi: 10.1214/088342307000000203

Gelman, A., Carlin, J. B., Stern, H. S., Dunson, D. B., Vehtari, A., and Rubin, D. B. (2014). *Bayesian Data Analysis, 3rd Edn*. Boca Raton, FL: CRC Press. doi: 10.1201/b16018

Gelman, A., Hill, J., and Vehtari, A. (2021). *Regression and Other Stories, 1st Edn*. Cambridge, MA: Cambridge University Press. doi: 10.1017/9781139161879

Gelman, A., and Shalizi, C. (2013). Philosophy and the practice of Bayesian statistics. *Brit. J. Math. Stat. Psychol.* 66, 8–38. doi: 10.1111/j.2044-8317.2011.02037.x

George, E. I., and Ročková, V. (2021). Comment: regularization via Bayesian penalty mixing. *Technometrics* 62, 438–442. doi: 10.1080/00401706.2020.1801258

Goldin, N., and Purse, B. V. (2016). Fast and flexible Bayesian species distribution modelling using gaussian processes. *Methods Ecol. Evol.* 7, 598–608. doi: 10.1111/2041-210X.12523

Gray, C. A., and Kennelly, S. J. (2018). Bycatches of endangered, threatened and protected species in marine fisheries. *Rev. Fish Biol. Fish.* 28, 521–541. doi: 10.1007/s11160-018-9520-7

Griffin, J., and Brown, P. (2017). Hierarchical shrinkage priors for regression models. *Bayesian Anal.* 12, 135–159. doi: 10.1214/15-BA990

Halsny, V. (2020). Nonresponse bias in inequality measurement: cross-country analysis using luxembourg income study surveys. *Soc. Sci. Q.* 101, 712–731. doi: 10.1111/ssqu.12762

Hansen, M. H., and Hurwitz, W. N. (1946). The problem of non-response in sample surveys. *J. Am. Stat. Assoc.* 41, 517–529. doi: 10.1080/01621459.1946.10501894

He, Y., Hodges, J., and Carlin, B. (2007). Re-considering the variance parametrization in multiple precision models. *Bayesian Anal.* 2, 529–556. doi: 10.1214/07-BA221

Hoffman, M. D., and Gelman, A. (2014). The no-U-turn sampler: adaptively setting path lengths in Hamiltonian Monte Carlo. *J. Mach. Learn. Res.* 15, 1593–1623.

ICES (2009). *Report of the Study Group for Bycatch of Protected Species (SGBYC)*. Technical report, Copenhagen. doi: 10.1093/ppar/19.1.22

ICES (2018). *Report from the Working Group on Bycatch of Protected Species, (WGBYC)*. Technical report, International Council for the Exploration of the Sea, Reykjavik.

ICES (2019). *Working Group on Bycatch of Protected Species (WGBYC)*. Technical report, International Council for the Exploration of the Sea, Faro.

ICES (2020a). *Bycatch of Protected and Potentially Vulnerable Marine Vertebrates - Review of National Reports under Council Regulation (EC) No. 812/2004 and Other Information*. Technical report.

ICES (2020b). *Report from the Working Group on Bycatch of Protected Species, (WGBYC)*. Technical report. International Council for the Exploration of the Sea.

ICES (2020c). *Workshop on Fisheries Emergency Measures to minimize BYCatch of Short-Beaked Common Dolphins in the Bay of Biscay and Harbour Porpoise in the Baltic Sea (WKEMBYC)*. International Council for the Exploration of the Sea.

Jaramillo-Legorreta, A. M., Cardenas-Hinojosa, G., Nieto-Garcia, E., Rojas-Bracho, L., Thomas, L., Ver Hoef, J. M., et al. (2019). Decline towards extinction of Mexico's Vaquita Porpoise (*Phocoena sinus*). *R. Soc. Open Sci*. 6:190598. doi: 10.1098/rsos.190598

Julian, F., and Beeson, M. (1998). Estimates of marine mammal, turtle, and seabird mortality for two California Gillnet Fisheries: 1990-1995. *Fish. Bull*. 96, 271–284.

Kiewiet De Jonge, C. D., Langer, G., and Sinozich, S. (2018). Predicting state presidential election results using national tracking polls and multilevel regression with poststratification (MRP). *Public Opin. Q*. 82, 419–446. doi: 10.1093/poq/nfy023

Komoroske, L. M., and Lewison, R. L. (2015). Addressing fisheries bycatch in a changing world. *Front. Mar. Sci*. 2:83. doi: 10.3389/fmars.2015.00083

Laran, S., Authier, M., Blanck, A., Dorémus, G., Falchetto, H., Monestiez, P., et al. (2017). Seasonal distribution and abundance of cetaceans within French waters-part II: the Bay of Biscay and the English channel. *Deep Sea Res. II* 141, 31–40. doi: 10.1016/j.dsr2.2016.12.012

Lax, J., and Phillips, J. (2009). How should we estimate public opinion in the states? *Am. J. Polit. Sci*. 53, 107–121. doi: 10.1111/j.1540-5907.2008.00360.x

Lennert, C., Kruse, S., Beeson, M., and Barlow, J. (1994). *Estimates of incidental marine Mammal Bycatch in California Gillnet Fisheries for July Through December, 1990*. Report of the International Whaling Commission.

Martin, R., Pearmain, E. J., Burfield, I. J., Small, C., Phillips, R. A., Yates, O., et al. (2019). Threats to seabirds: a global assessment. *Biol. Conserv*. 237, 525–537. doi: 10.1016/j.biocon.2019.06.033

Martin, S. L., Stohs, S. M., and Moore, J. E. (2015). Bayesian inference and assessment for rare-event bycatch in marine fisheries: a drift gillnett fishery case study. *Ecol. Appl*. 25, 416–429. doi: 10.1890/14-0059.1

McCulloch, C. E., and Neuhaus, J. M. (2011). Misspecifying the shape of a random effects distribution: why getting it wrong may not matter. *Stat. Sci*. 26, 388–402. doi: 10.1214/11-STS361

Monnahan, C. C., Thorson, J. T., and Branch, T. A. (2017). Faster estimation of bayesian models in ecology using Hamiltonian Monte Carlo. *Methods Ecol. Evol*. 8, 339–348. doi: 10.1111/2041-210X.12681

Morris, M., Wheeler-Martin, K., Simpson, D., Mooney, S. J., Gelman, A., and DiMaggio, C. (2019). Bayesian hierarchical spatial models: implementing the Besag York Mollié model in Stan. *Spat Spatio Temp Epidemiol*. 31:100301. doi: 10.1016/j.sste.2019.100301

National Marine Fisheries Service (2004). *Evaluating Bycatch: a National Approach to Standardized Bycatch Monitoring Programs*. Technical Report. U.S. Department of Commerce, NOAA.

National Marine Fisheries Service (2005). *Revisions to Guidelines for Assessing Marine Mammal Stocks*. Technical report, NOAA.

National Marine Fisheries Service (2016). *Guidelines for Preparing Stock Assessment Reports Pursuant to the 1994 Amendments to the MMPA*. Technical report, NOAA.

Neal, R. (1998). "Regression and classification using Gaussian process priors," in *Bayesian Statistics*, eds J. M. Bernardo, J. O. Berger, A. P. Dawid, and A. F. M. Smith (Oxford: Oxford University Press), 475–501.

Northridge, S., Kingston, A., and Thomas, L. (2019). *Annual Report on the Implementation of Council Regulation (EC) No 812/2004 During 2018*. Technical report, University of St Andrews.

Pacoureau, N., Rigby, C. L., Kyne, P. M., Sherley, R. B., Winker, H., Carlson, J. K., et al. (2021). Half a century of global decline in oceanic sharks and rays. *Nature* 589, 567–571. doi: 10.1038/s41586-020-03173-9

Palka, D. L., and Rossman, M. C. (2001). *Bycatch Estimates of Coastal Bottlenose Dolphin* (*Tursiops truncatus*) in US Mid-Atlantic Gillnet Fisheries for 1996 to 2000. Technical Report. Northeast Fisheries Science Center.

Peltier, H., Authier, M., Caurant, F., Dabin, W., Daniel, P., Dars, C., et al. (2021). In the wrong place at the wrong time: identifying spatiotemporal co-occurrence of bycaught common dolphins and fisheries in the Bay of Biscay (NE Atlantic) from 2010 to 2019. *Front. Mar. Sci*. 8:617342. doi: 10.3389/fmars.2021.617342

Peltier, H., Authier, M., Deaville, R., Dabin, W., Jepson, P., van Canneyt, O., et al. (2016). Small cetacean bycatch as estimated from stranding schemes: the common dolphin case in the Northeast Atlantic. *Environ. Sci. Policy* 63, 7–18. doi: 10.1016/j.envsci.2016.05.004

Pérez, M., Perrichi, L. R., and Ramírez, I. (2017). The scaled beta2 distribution as a robust prior for scales. *Bayesian Anal*. 12, 615–637. doi: 10.1214/16-BA1015

Pielke, R. A. Jr. (2007). *The Honest Broker - Making Sense of Science in Policy and Politics, 1st Edn*. Cambridge, MA: Cambridge University Press.

Plummer, M., Best, N., Cowles, K., and Vines, K. (2006). CODA: convergence diagnosis and output analysis for MCMC. *R News* 6, 7–11.

Punt, A. E., Siple, M., Francis, T. B., Hammond, P. S., Heinemann, D., Long, K. J., et al. (2021). Can we manage marine mammal bycatch effectively in low-data environments? *J. Appl. Ecol*. 58, 596–607. doi: 10.1111/1365-2664.13816

R Core Team (2020). *R: A Language and Environment for Statistical Computing*. Vienna: R Foundation for Statistical Computing.

Rayner, S. (2012). Uncomfortable knowledge: the social construction of ignorance in science and environmental policy discourses. *Econ. Soc*. 41, 107–125. doi: 10.1080/03085147.2011.637335

Rhem, J., Kilian, C., Rovira, P., Shield, K. D., and Manthey, J. (2020). The elusiveness of representativeness in general population surveys for alcohol. *Drug Alcohol* 40, 161–165. doi: 10.1111/dar.13148

Sala, A., Konrad, C., and Doerner, H. (eds.). (2019). *Review of the Implementation of the EU Regulation on the Incidental Catches of Cetaceans (STECF-19-07)*. Publications Office of the European Union, Luxembourg.

Sarewitz, D. (1999). How science makes environmental controversies worse. *Environ. Sci. Policy* 7, 385–403. doi: 10.1016/j.envsci.2004.06.001

Sims, M., Cox, T., and Lewison, R. (2008). Modeling spatial patterns in fisheries bycatch: improving bycatch maps to aid fisheries management. *Ecol. Appl*. 18, 649–661. doi: 10.1890/07-0685.1

Stan Development Team (2020). *RStan: the R interface to Stan*. R package version 2.21.22. New York, NY.

St. John, F., Keane, A., Jones, J., and Milner-Guiland, E. (2014). Robust study design is as important on the social as it is on the ecological side of applied ecological research. *J. Appl. Ecol*. 51, 1479–1485. doi: 10.1111/1365-2664.12352

Stransky, C., and Sala, A. (2019). *Scientific, Technical and Economic Committee for Fisheries (STECF) - Revision of the EU-MAP and Work Plan Template (STECF-19-12)*. Technical report, Publications Office of the European Union, Luxembourg.

Tregenza, N. J. C., Berrow, S. D., Hammond, P. S., and Leaper, R. (1997). Harbour porpoise (*Phocoena phocoena* L.) by-catch in set gillnets in the Celtic Sea. *ICES J. Mar. Sci*. 54, 896–904. doi: 10.1006/jmsc.1996.0212

Tremblay-Boyer, L., and Berkenbusch, K. (2020). *Characterisation of Marine Mammal Interactions with Fisheries* & *Bycatch Mitigation*. Report for project INT2019-03 prepared for Department of Conservation. Technical report, Dragonfly Data Science.

Trouwborst, A., and Somsen, H. (2019). Domestic cats (*Felis catus*) and European nature conservation law–applying the EU birds and habitats directives to a significant but neglected threat to wildlife. *J. Environ. Law* 32, 391–415. doi: 10.1093/jel/eqz035

Ulrich, C., and Doerner, H. (eds.). (2021). *Scientific, Technical and Economic Committee for Fisheries (STECF) - 66th Plenary Report (PLEN-21-01)*. Publications Office of the European Union, Luxembourg.

Vehtari, A., Gelman, A., Simpson, D., Carpenter, B., and Bürkner, P. (2019). *Rank-Normalization, Folding, and Localization: an Improved $\hat{R}$ for Assessing Convergence of McMC*. Technical report, Helsinki Institute for Information Technology, Department of Computer Science, Aalto University.

Wade, P. R. (1998). Calculating limits to the total allowable human-caused mortality of cetaceans and pinnipeds. *Mar. Mammal Sci*. 14, 1–37. doi: 10.1111/j.1748-7692.1998.tb00688.x

Wang, W., Rothschild, D., Goel, S., and Gelman, A. (2015). Forecasting elections with non-representative polls. *Int. J. Forecast*. 31, 980–991. doi: 10.1016/j.ijforecast.2014.06.001

Zahorski, A. (2020). *Multilevel Regression With Post-Stratification for the National Level VIBER/Street Poll on the 2020 Presidential Election in Belarus*. Technical report. Uladzimir Karatkevich National University of Belarus.

Zhang, X., Holt, J. B., Lu, H., Wheaton, A. G., Ford, E. S., Greenlund, K. J., et al. (2014). Multilevel regression and poststratification for small-area estimation of population health outcomes: a case study of chronic obstructive pulmonary disease prevalence using the behavioural risk factor surveillance system. *Am. J. Epidemiol*. 179, 1025–1033. doi: 10.1093/aje/kwu018

Conflict of Interest: MA is employed by the commercial company ADERA which did not play any role in this study beyond that of employer.

The remaining authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

frontiers in Marine Science

ORIGINAL RESEARCH
published: 27 October 2021
doi: 10.3389/fmars.2021.775187

Using Bayesian Models to Estimate Humpback Whale Entanglements in the United States West Coast Sablefish Pot Fishery

Jason E. Jannot[1]*†, Eric J. Ward[2]†, Kayleigh A. Somers[1], Blake E. Feist[2], Thomas P. Good[2], Dan Lawson[3] and James V. Carretta[4]

[1] Fishery Resource Analysis Monitoring Division, Northwest Fisheries Science Center, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, Seattle, WA, United States, [2] Conservation Biology Division, Northwest Fisheries Science Center, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, Seattle, WA, United States, [3] Protected Resources Division, West Coast Regional Office, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, Long Beach, CA, United States, [4] Marine Mammal and Turtle Division, Southwest Fisheries Science Center, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, La Jolla, CA, United States

Edited by:
Andre Eric Punt,
University of Washington,
United States

Reviewed by:
Charlotte Boyd,
International Union for Conservation of Nature, United States
Milani Chaloupka,
Ecological Modelling Services Pty Ltd., Australia
Paul Kinas,
Federal University of Rio Grande, Brazil

***Correspondence:**
Jason E. Jannot
jason.jannot@noaa.gov

† These authors have contributed equally to this work and share first authorship

Specialty section:
This article was submitted to Marine Megafauna, a section of the journal Frontiers in Marine Science

Received: *13 September 2021*
Accepted: *07 October 2021*
Published: *27 October 2021*

Citation:
Jannot JE, Ward EJ, Somers KA, Feist BE, Good TP, Lawson D and Carretta JV (2021) Using Bayesian Models to Estimate Humpback Whale Entanglements in the United States West Coast Sablefish Pot Fishery. Front. Mar. Sci. 8:775187. doi: 10.3389/fmars.2021.775187

Protected species bycatch can be rare, making it difficult for fishery managers to develop unbiased estimates of fishing-induced mortality. To address this problem, we use Bayesian time-series models to estimate the bycatch of humpback whales (*Megaptera novaeangliae*), which have been documented only twice since 2002 by fishery observers in the United States West Coast sablefish pot fishery, once in 2014 and once in 2016. This model-based approach minimizes under- and over-estimation associated with using ratio estimators based only on intra-annual data. Other opportunistic observations of humpback whale entanglements have been reported in United States waters, but, because of spatio-temporal biases in these observations, they cannot be directly incorporated into the models. Notably, the Bayesian framework generates posterior predictive distributions for unobserved entanglements in addition to estimates and associated uncertainty for observed entanglements. The United States National Marine Fisheries Service began using Bayesian time-series to estimate humpback whale bycatch in the United States West Coast sablefish pot fishery in 2019. That analysis resulted in estimates of humpback whale bycatch in the fishery that exceeded the previously anticipated bycatch limits. Those results, in part, contributed to a review of humpback whale entanglements in this fishery under the United States Endangered Species Act. Building on the humpback whale example, we illustrate how the Bayesian framework allows for a wide range of commonly used distributions for generalized linear models, making it applicable to a variety of data and problems. We present sensitivity analyses to test model assumptions, and we report on covariate approaches that could be used when sample sizes are larger. Fishery managers anywhere can use these models to analyze potential outcomes for management actions, develop bycatch estimates in data-limited contexts, and guide mitigation strategies.

Keywords: Endangered Species, Biological Opinion, fisheries management, rare event analysis, bycatch, statistical analysis, whale entanglement, fisheries observer

INTRODUCTION

Estimating the mortality of marine mammals and other protected species incidentally caught during commercial fishing operations (bycatch) is an important, but often, challenging task. Economic, logistical, and other constraints make a complete census of fishing effort and bycatch impractical in most cases (NMFS, 2004). Therefore, managers must rely on estimates of bycatch to accurately assess marine mammal stocks (Wade, 1998), and to set fishing impact reference points (Moore et al., 2013), population recovery goals, and species-specific protective status. Estimates of fishing mortality can define the conservation priorities for marine mammals and help determine if mitigating fishing impacts is necessary or if limited conservation resources should be applied elsewhere. The rarity of bycatch events, which vary by fishery and marine mammal species involved, makes it challenging to develop robust bycatch estimates that are critical for setting recovery goals and conservation priorities. Bycatch can be rare for a number of reasons, including: fishing vessels and animals only occasionally overlap in time or space; vessel-mammal interactions are unobserved (i.e., cryptic; Gilman et al., 2013) or observation rates are low (Moore et al., 2011; Wakefield et al., 2018; Curtis and Carretta, 2020); fishers deliberately avoid marine mammals; or simply because the species itself is rare, sometimes as a consequence of fishery or other human-induced mortality.

Robust estimation of rare bycatch events has been identified as "…a central challenge to bycatch research" (Komoroske and Lewison, 2015). The sample size necessary to accurately estimate rare events is usually prohibitively large (Babcock et al., 2003; Dixon et al., 2005; Amande et al., 2012; Wakefield et al., 2018). The rarity of bycatch leads to a large number of zeros (non-events) which, when modeled with standard methods, tends to over- or under-estimate both the total number of mortalities as well as the associated uncertainty of bycatch estimates (Lewin et al., 2010; Carretta and Moore, 2014; Martin et al., 2015; Wakefield et al., 2018; Parsa et al., 2020). Ratio estimators have been widely used in bycatch estimation (Stratoudakis et al., 1999; Borges et al., 2005; Walmsley et al., 2007). Ratio estimators rely on the assumption that bycatch is proportional to some metric or proxy of fishing effort, such as fishery landings (Rochet and Trenkel, 2005) and the ratio is used to expand bycatch estimates from the observed vessels to the unobserved vessels in the fleet. Ratio estimators are ill-suited for highly variable (i.e., over-dispersed) bycatch data, because events are too few to accurately assess bycatch probability (McCracken, 2004; Amande et al., 2012; Carretta and Moore, 2014; Martin et al., 2015). In extreme cases where bycatch has never been observed, ratio estimators predict zero probability of bycatch without properly estimating the probability of an unobserved event (Carretta and Moore, 2014), though tools exist to assess this probability (Curtis and Carretta, 2020). In conservation scenarios where minimizing the risk of harm to protected species is a priority, an estimate of zero probability does not adequately capture the risk or the consequences of unobserved bycatch.

Several modeling solutions have been proposed to better capture the risk of rare bycatch events. Pooling across years of estimates has been used in marine mammal stock assessments and to set limits on bycatch (Carretta and Moore, 2014) and pooling across similar vessels has been used in seabird bycatch estimation (Parsa et al., 2020). However, the number of years or vessels to pool, even when standardized among analysts, is often based on expert opinion and unique to the situation or dataset. Various forms of probability models have been employed to estimate rare events; however, these methods often require large sample sizes to overcome the lack of bycatch events (Lewin et al., 2010; Stock et al., 2020). More recently, machine learning techniques (i.e., random forests; Breiman, 2001) have been used to estimate rare species distributions (Siders et al., 2020) and bycatch (Carretta et al., 2017). Machine learning techniques can reduce bias in rare event data, but are typically data-intensive and can be challenging to interpret (Breiman, 2001). More recently, Bayesian methods have gained traction as a model-based alternative to using machine learning techniques for rare bycatch events (Cosandey-Godin et al., 2015; Martin et al., 2015; Parsa et al., 2020).

Humpback whales (family Balaenopteridae) are found in all oceans of the world. They were listed as endangered under the United States Endangered Species Act (ESA) in 1973 and classified as depleted under the United States Marine Mammal Protection Act (MMPA) in that same year (Carretta et al., 2020b). Fourteen populations of humpback whales have been identified (Bettridge et al., 2015). Of the 14 populations, four are listed as endangered, and one is listed as threatened. Three populations occur off the coasts of Washington, Oregon, and California – the Hawaii population (not ESA-listed), the Mexico population (ESA-threatened), and the Central America population (ESA-endangered) (NMFS, 2020a).

One direct threat to humpback whales is entrapment and entanglement in fishing gear (NMFS, 1991). Along with ship collisions, fishing gear represents most of the serious injuries and mortalities reported around the globe for humpback whales (review in Carretta et al., 2020b). Pot and trap fishery entanglements are the most frequently documented source of serious injury and mortality of this species in United States West Coast waters, and, starting in 2014, entanglement reports began to increase (Carretta et al., 2020b). The specific population of each individual humpback whale entangled in United States West Coast fisheries are usually not known; however, NMFS assumes that animals from ESA-listed populations (i.e., Mexico and Central America) interact with these fisheries based on their relative abundances along the United States West Coast (NMFS, 2020a).

We illustrate the use of Bayesian models to estimate humpback whale entanglements in the United States West Coast pot fishery targeting sablefish (*Anoplopoma fimbria*), which overlaps in time and space with the three humpback whale populations found on the United States West Coast (NMFS, 2020a). This model-based approach minimizes under- and over-estimation associated with using ratio estimators based only on intra-annual data. Our framework also generates probability distributions of unobserved entanglements in addition to estimates and associated uncertainty for observed entanglements. Estimating unobserved entanglements is particularly important in the case of United States West Coast humpback whales. Opportunistic observations of humpback whale entanglements by the sablefish pot fishery have been reported in United States waters, and these numbers likely represent the minimum number of entanglements that have occurred. However, due to the spatio-temporal biases in these observations, they cannot be directly incorporated into the models.

The NMFS began using Bayesian models to estimate humpback whale bycatch in the United States West Coast sablefish pot fishery in 2019. Here we demonstrate the use of these methods to estimate the annual and 5-year average bycatch of humpback whales and compare these estimates against two management thresholds (NMFS, 2020a). We compare our estimates of bycatch to management thresholds originally developed in 2012 (NMFS, 2012) and subsequently revised in 2020 (NMFS, 2020a). We use the humpback whale example to illustrate how the Bayesian framework allows for a wide range of commonly used distributions for count data or other non-normal data types, making it applicable to a variety of data and problems. We also present sensitivity analyses to test model assumptions and report on covariate approaches that could be used when sample sizes are larger.

MATERIALS AND METHODS

To illustrate the use of Bayesian models, we employed fisheries-dependent data from the United States West Coast sablefish pot fishery. In this article, we focus on two sectors in the sablefish pot fishery: the limited entry (LE) sector (∼ 90 vessels/year), where fishers have individual quota to catch sablefish during the seven month season (April–October), and the open access (OA) sector (∼ 472 vessels/year), which is managed by per-trip landing limits on sablefish and is open year-round. In both the LE and OA sectors, a subset of vessels are monitored for bycatch, and the observed portion of each of these fleets is used to estimate bycatch for the entire fleet (observed + unobserved). Estimates were obtained for each sector separately and then the separate estimates were summed for comparison against management bycatch thresholds. A third sector, the Catch Share (CS) pot sector, also fishes along the United States West Coast. In the CS sector, individual permit holders obtain and fish individual quota for a number of groundfish species including sablefish. Since its inception (2011), the CS program requires 100% monitoring on all trips. During 2011–2014, all CS pot trips carried a fisheries observer for monitoring compliance purposes. Since 2015, roughly 50% of the CS pot trips have been monitored by fishery observers and the remaining trips are monitored by cameras and other automated sensing devices (collectively known as electronic monitoring, or EM). There has never been an observed humpback whale entanglement in the CS pot sector; therefore, we concentrate our analyses on the LE and OA sectors that have had observed entanglements of humpback whales and where monitoring is <100% of trips. Although no estimates of historical entanglements in the CS pot sector have been made, the potential risk of entanglements in the CS pot sector in the future was considered in the 2020 Biological Opinion (NMFS, 2020a).

Data from the year when fishery observers were first deployed (LE = 2002 and OA = 2003) until 2019 were provided by the National Oceanographic and Atmospheric Administration (NOAA) Northwest Fisheries Science Center (NWFSC) Fisheries Observation Science (FOS) Program. The FOS collects independent, at-sea fisheries data by deploying trained scientists (a.k.a., observers) on commercial fishing vessels along the United States West Coast, including in the LE and OA sablefish pot fisheries (NWFSC, 2020c). During fishing trips, observers record information about catch by species, including at-sea discards, as well as the location and depth of fishing effort. Monitoring the catch for marine mammal and other protected species interactions and bycatch is the observer's highest priority (NWFSC, 2020c). FOS strives to deploy observers on 30% of LE sablefish pot fishery trips, which has priority over the OA pot fishery where target observation rates are 5–10%. Pot vessels in both sectors are randomly selected for observation prior to the start of the fishing season. Realized annual observer coverage varies between 14 and 72% for the LE fleet and between 2 and 12% for the OA fleet, based on the percentage of total fleet-wide landings (Somers et al., 2020a). Fleet-wide landings are estimated from landing receipts, called fish tickets, generated when the fish is purchased at the dock (**Supplementary Text**). Across all years, the observed portion of the LE fishery deploys gear at an average depth of 489 m and between roughly 36° and 48° north latitude, whereas the observed portion of the OA pot fishery deploys gear in an average depth of 485 m deep, typically between 32° and 47° north latitude. There have been slight inter-annual shifts in average fishing depth in both sectors, with a greater proportion of retained catches being from greater depths in recent years (**Supplementary Figure 1**; see also Somers et al., 2020b). The two observed humpback entanglements occurred when the pot gear was being fished between a depth of 140 and 220 m.

Serious Injury and Mortality Determinations

Serious injury and mortality designations were determined by marine mammal experts (Carretta et al., 2020a) using established guidelines. Under the MMPA and ESA, a "take" is defined as any act that harasses, hunts, captures, or kills, or attempts to harass, hunt, capture, or kill a marine mammal, including all humpback whale entanglements, regardless of lethality. Fisheries observer notes and data, and, when available, photographs and video, recorded at the time of interactions, informed take designations. Observers typically detail the nature of the injury and changes in the animal's behavior following its release. Noted factors indicating a potential mortality could include evidence of bleeding, broken bones, wounds, trailing gear, vomiting, and abnormal behavior (NWFSC, 2020c).

Bycatch Estimation

Statistical Model

We used Bayesian models to estimate annual means and variability of humpback whale bycatch within the LE and OA sectors, for both the observed and unobserved portions of the fleets. For any application of these methods to bycatch data, there are three parameterization choices to be made. First, the effort metric on observed vessels must be chosen; effort is used to expand estimated observed bycatch to unobserved bycatch. For our application there are three possible choices as a proxy for fishing effort: number of gear deployments, number of gear units, or mass of landed catch [as weight in metric tons (mt)]. Second, these models allow for constant or time-varying bycatch rates, either as autoregressive processes or as a function of covariates. Third, the bycatch-generating process or data model must be specified; examples include Poisson, negative binomial, or zero-inflated models. Even though our simulations and code include options for fitting zero-inflated models, we did not apply those to the humpback whale data because of the limited number of observed takes. We formally compare all combinations of the three effort metrics, two potential bycatch rates, and two possible bycatch-generating models, Poisson and negative binomial. We use methods from the R package implementing Stan (Stan Development Team, 2021), *loo* (Vehtari et al., 2020) as implemented in the R package, *bycatch*[1] (Ward and Jannot, 2021) to compare among models. Final estimates are presented from the single model that best fits the data.

For each sector (LE and OA), the base model assumed bycatch rate was constant and inferred annual expected mortality conditioned on fishing effort, using a simple Poisson process model (Martin et al., 2015), where the total number of observed bycatch events were assumed to follow a Poisson distribution,

$$n_{take,y} \sim \left(\lambda_y = \theta \cdot E_y\right)$$

where:

$n_{take,y}$ = number of observed bycatch events (or take events) in year y

λ_y = expected observed bycatch

θ = estimated observed bycatch rate

E_y = observed effort in year y

The estimated bycatch rate, θ, in the simplest scenario, is assumed to be constant through time, but the quantity $\theta \cdot E_y$ includes parameter uncertainty because θ is estimated. Thus, a time series of the expected observed bycatch can be generated for a given species, with a given metric of effort. Fluctuations in fishing effort through time then result in year-to-year variability (percent observer coverage only affects the expansion). We used a Bayesian implementation of this model (Martin et al., 2015) to generate mean and 95% credible intervals (CIs) of the bycatch rate parameter, θ, as well as for the expected bycatch in the observed portion of the fleet, $\theta \cdot E_y$. For more information regarding distributions and implementation in R and Stan (Stan Development Team, 2021), please see the articles in the *bycatch* package (Ward and Jannot, 2021).

We built upon the simplified model above with the goal of finding the model that most accurately estimates bycatch and variance. To do that, we compared models to: (a) find the most suitable effort metric; (b) test the assumption that θ is constant through time; and (c) compare distributions (Poisson to negative binomial). Though our code allows for the inclusion of covariates, which may vary through time, we only considered time-varying models that treat bycatch rate as a random walk (in log space), $\theta_y \sim Normal\left(\theta_{y-1}, \sigma_\theta\right)$, where σ_θ is an estimated parameter controlling the year to year variability.

Model Diagnostics and Selection

Before comparing among models, each model must be tested for efficacy using the Pareto-K values. Theoretically, the Pareto smooth importance sampling (PSIS) should converge to a mean and variance for the distribution. However, due to the use of random variables, convergence does not always emerge. General rules of thumb for evaluating the Pareto-K statistics are that "low" Pareto-K values (<0.5) indicate convergence of the mean and variance "slightly high" Pareto-K values ($0.5 \leq K < 1$) indicate a model whose variance either does not converge at all, or converges slowly, and "high" Pareto-K statistics ($K > 1$) indicates neither the mean nor the variance converges (Vehtari et al., 2019).

In addition to Pareto-K values, Leave One Out (LOO) can be used to test for over-parameterization by generating a p-LOO value which is compared to the number of parameters used in the model. The parameters for the model include all the incorporated covariates, as well as time, effort, and distribution. A p-LOO less than the number of parameters denotes an appropriately parameterized model.

Once a model is considered suitable, the optimal model can be chosen by comparing among LOOIC estimates. For each sector (LE and OA) there are a total of 12 possible models (three effort metrics, two bycatch rates and two bycatch processes). Leave One Out Cross Validation (LOOCV) is a widely used tool to identify models with good predictive ability; this can be done in a Bayesian framework, but could be slow depending on the number of folds used. As an alternative, the *loo* package approximates LOOCV by implementing LOO sampling, which tests the efficacy of the model based on its predictive ability for new data (Vehtari et al., 2020). LOO is based on PSIS. Importance sampling is typically used when multiple distributions may be present, or when the density of the distribution is only partially known (Vehtari et al., 2019). Like more familiar model selection criteria, such as AIC, the preferred model is the model with the lowest LOOIC estimate.

The 12 models within each fishing sector were tested, in the order given below, and excluded if any of the following cases were met:

1. Pareto-K $>$ 0.7, as suggested by Vehtari et al. (2019)
2. p-LOO $>$ 3 (the number of parameters)
3. LOOIC is not the minimum.

Sensitivity Analysis: Model Assumptions

To evaluate the ability of our approach to identify the correct data-generating model, we performed a series of analyses on simulated datasets. We simulated a time series (20 time steps) using low or high mean bycatch rates (0.1 and 1.0, respectively), and generated observations using either a Poisson or negative binomial data model. For each simulated time series, we fit the model using the Stan code in our bycatch package with three different estimation models (Poisson, negative binomial, and zero-inflated Poisson distribution). We repeated the diagnostics

---

[1] https://ericward-noaa.github.io/bycatch/

described above for each simulated time series. For each set of estimation models fit to the same simulated dataset, we calculated the lowest LOOIC value and difference between the LOOIC estimate from each model and the lowest value. Smaller differences correspond to greater data support, or a greater similarity in predictive ability between a given model and the model with the lowest LOOIC. We used 100 replicates for each of the above combinations (1200 estimation models applied to 400 simulated datasets).

Sensitivity Analysis: Data Assumptions

Though our model selection procedure indicated that the sparsity of the data prevented us from fitting complex models, we performed a sensitivity analysis to explore how assumptions about data, and specifically changes in how effort is distributed across depth, may influence results. For each sector, we used partitioning around medoids (PAM) clustering (Hennig, 2020), with an unknown number of clusters, to identify groups. Each sector supported two depth strata, and breakpoints were similar across sectors (**Supplementary Figure 1**; 395.5 m for LE sector and 360 m for OA sector). We repeated the primary analysis described above, using the best model, and compared results using data from all depths to the results obtained when only including data from the shallower depth strata where takes were observed.

Expanding Bycatch to Unobserved Portion of Fleet

Because observer coverage is less than 100% in both fishery sectors, and variable through time, we need to expand the estimated bycatch in the observed portion of the fleet, $\theta \cdot E_y$, to the entire fleet, which includes unobserved vessels. One approach for expansion would be to divide $\theta \cdot E_y$ by the percent observer coverage; however, this ignores uncertainty in the expansion. We accounted for uncertainty in the expansion by estimating the posterior predictive distribution of unobserved takes, given unobserved effort and estimated parameters, $P(Y^*|Y) = \int_\theta P(Y^*|\theta, Y)P(\theta|Y)d\theta$. We subtracted the observed effort from the total effort to obtain the unobserved effort. We used these simulated posterior predictive values to generate 95% CIs for the predicted total bycatch in each year (adding observed bycatch to the posterior predictive distribution of unobserved bycatch). Details on the implementation of this in R can be found in the *bycatch* package (Ward and Jannot, 2021). Fleet-wide bycatch of humpback whales was estimated for each sector using observer coverage data (Somers et al., 2020a).

Comparison to Management Thresholds

Both the 2012 and the 2020 Biological Opinions (NMFS, 2012, 2020a) specify annual and 5-year running average bycatch limits. To compare our annual estimates to these management thresholds, we estimated total bycatch (observed + unobserved) for each sector separately, summed the LE + OA estimates and compared the combined annual 2019 estimate to the annual thresholds defined in the in the Biological Opinions (NMFS, 2012, 2020a). We then used the LE + OA summed annual estimates to calculate the 5-year average total bycatch for 2015–2019, and compared that estimate to the 2012 and 2020 5-year average thresholds. Because our Bayesian estimates are inherently probabilistic, we also generated probabilities of exceeding the 2012 and 2020 thresholds.

Statistical Software

The statistical software R (R Core Team, 2020) was used to produce the analyses, tables, and figures in this report. Specifically, we relied on the R packages *bycatch* (Ward and Jannot, 2021) for modeling and simulation, *ggplot2* (Wickham, 2016) for plotting figures, *loo* (Vehtari et al., 2020) for model comparisons, and *tidyverse* (Wickham et al., 2019) for data wrangling.

RESULTS

Estimated Bycatch of Humpback Whales

In both fishery sectors, the models that converged and had the lowest LOOIC used a constant bycatch rate and a Poisson process for bycatch (**Table 1**). Most models that treated observed bycatch as originating *via* a negative binomial distribution, or models that included time-varying bycatch rates did not meet the convergence criteria; specifically, the variance of the random walk was not identifiable. We did not estimate a single model for both sectors combined because the fishing areas, targets, and tactics are sufficiently different between the two sectors to warrant separate models for each sector. When comparing the three measures of fishing effort, in the LE pot fishery, the number of pots deployed was the best proxy of fishing effort, whereas in the OA pot fishery, the observed landings was the best proxy of effort (**Table 1**).

Humpback whales were observed entangled in United States West Coast sablefish pot gear twice by fishery observers since 2002. The single 2014 entanglement in the LE pot fishery led to an annual estimate in the most recent year of available data (2019) of 0.13 entanglements (95% CI: 0.0–1.0; **Table 2**). The single 2016 entanglement in the OA pot fishery led to a 2019 estimate of 1.02 entanglements (CI: 0.0–4.0; **Table 3**).

The 2019 annual estimate of entanglements from the LE + OA sectors combined was 1.15 (CI: 0.0–5.0). This estimate was below both the 2012 and 2020 annual entanglement threshold (**Figure 1** bottom panel; 2012: 3 whales/year; NMFS, 2012, 2020a: 5 whales/year; NMFS, 2020a). The most recent estimated 5-year average (2015–2019) of entanglements from the LE + OA sectors combined was 1.60 (CI: 0.2–4.8; **Tables 2**, **3** and **Figure 1**). This estimate was above the 2012 5-year average threshold of 1 animal/year (**Figure 1** top panel; NMFS, 2012), but below the 2020 5-year average threshold of 2.34 animals/year (NMFS, 2020a; **Figure 1** top panel). Exceeding the 2012 5-year average threshold contributed, in part, to the re-evaluation of the original 2012 Biological Opinion and resulted in the revised threshold value.

Both the annual and the 5-year average estimated takes showed a peak in 2009, then trended downward until 2013 and then upward until 2017, after which the estimates level off (**Figure 1**). The probability of exceeding the thresholds follows a similar trend over time as the estimated takes and uncertainty, with the entire probability trend shifting location along the y-axis

TABLE 1 | Model diagnostics [convergence, LOOIC, and LOOIC standard error (SE)] by fishery sector for each fishing effort metric, time-varying, and bycatch process model choice. Asterisk (*) indicates the model that both converged and had the lowest LOOIC.

Fishery sector	Fishing effort metric	Is bycatch rate time varying?	Bycatch process	Did the model converge?	LOOIC	LOOIC SE
OA	Number of pots deployed	No	Poisson	Yes	10	6.9
OA	Soak time	No	Poisson	Yes	10.3	7.4
OA*	Observed landings by weight	No	Poisson	Yes	9.6	6.5
OA	Number of pots deployed	Yes	Poisson	No	11.6	8.2
OA	Soak time	Yes	Poisson	No	12.3	8.1
OA	Observed landings by weight	Yes	Poisson	No	10.8	7.4
OA	Number of pots deployed	No	Negative binomial	Yes	10.5	7.3
OA	Soak time	No	Negative binomial	No	11.5	8.3
OA	Observed landings by weight	No	Negative binomial	No	10.0	6.8
OA	Number of pots deployed	Yes	Negative binomial	No	12.0	8.2
OA	Soak time	Yes	Negative binomial	No	15.0	8.5
OA	Observed landings by weight	Yes	Negative binomial	No	11.2	7.6
LE*	Number of pots deployed	No	Poisson	Yes	11.3	8.3
LE	Soak time	No	Poisson	No	11.5	8.7
LE	Observed landings by weight	No	Poisson	No	11.7	8.6
LE	Number of pots deployed	Yes	Poisson	No	11.3	8.5
LE	Soak time	Yes	Poisson	No	12.3	9.4
LE	Observed landings by weight	Yes	Poisson	No	12.6	9.1
LE	Number of pots deployed	No	Negative binomial	No	11.7	8.4
LE	Soak time	No	Negative binomial	No	11.9	8.9
LE	Observed landings by weight	No	Negative binomial	No	11.8	8.5
LE	Number of pots deployed	Yes	Negative binomial	No	13.0	8.8
LE	Soak time	Yes	Negative binomial	No	13.5	9.4
LE	Observed landings by weight	Yes	Negative binomial	No	13.0	9.0

(probability) depending on the threshold (**Figure 2**). This results in the probability of exceeding the bycatch thresholds higher overall in 2012, when the thresholds were lower (three whales in a single year or a 5-year average of 1/year) as compared to the 2020 thresholds (five whales in a single year or a 5-year average of 2.34/year; **Figure 2**). Irrespective of the specific values of the threshold (e.g., 2012 vs. 2020), the probability of exceeding the 5-year average threshold appears to always be greater than the probability of exceeding the annual threshold (**Figure 2**).

Sensitivity Analysis: Model Assumptions

Our simulation results highlight that with short time series (20 time steps) and sparse observations, data models that have more parameters than the Poisson are generally not supported (**Figure 3**). As expected, when data are generated from a Poisson model, the Poisson estimation model generally has the lowest LOOIC estimate, corresponding to more support for a Poisson bycatch process (**Figure 3**). This result was true regardless of the simulated bycatch rate, which controlled the sparsity in the data. Sparse and over-dispersed data from a negative binomial distribution is more challenging; with low mean bycatch rates (0.1), we found more support for zero-inflated Poisson models, whereas, at higher bycatch rates (1.0) the negative binomial model was favored (smaller changes in LOOIC; **Figure 3**). This indicates that the negative binomial model might not be the best choice for sparse datasets. A benefit of many Bayesian model selection tools, such as LOOIC, is that, in addition to individual point estimates, standard errors can also be estimated. As expected with short time series and sparse data, we found that for many of our simulated replicates, standard errors overlapped between models; the average difference in LOOIC between models was 1.98, and average standard error across all estimates was 8.16.

Sensitivity Analysis: Data Assumptions

Using the depth breakpoints identified for each sector (**Supplementary Figure 1**; 395.5 m for LE sector and 360.0 m for OA sector), we repeated our analysis using a model with a

TABLE 2 | The data used to calculate humpback whale bycatch in the LE sablefish pot fishery collected by fishery observers (observed) and the estimated mean and 95% credible interval (CI: lower–upper) from the best model. The best model included a constant bycatch rate, a Poisson count process, and used the number of observed pots deployed as fishing effort.

Year	Sector	Observed			Estimated bycatch	
		Pots (#)	Landings (%)	Bycatch (#)	Mean (#)	CI (95%)
2002	LE	5438	23	0	0.1	0–2
2003	LE	9017	25	0	0.2	0–2
2004	LE	5378	13	0	0.2	0–1
2005	LE	13,822	46	0	0.1	0–1
2006	LE	10,708	34	0	0.1	0–1
2007	LE	5816	21	0	0.1	0–1
2008	LE	13,638	57	0	0.1	0–1
2009	LE	3883	14	0	0.2	0–1
2010	LE	11,294	28	0	0.2	0–1
2011	LE	9029	37	0	0.1	0–1
2012	LE	14,218	35	0	0.2	0–1
2013	LE	1934	14	0	0.1	0–1
2014	LE	7561	31	1	1.1	1–2
2015	LE	11,329	61	0	0.1	0–1
2016	LE	21,219	71	0	0.1	0–1
2017	LE	7852	31	0	0.1	0–1
2018	LE	18,424	72	0	0.1	0–1
2019	LE	17,518	50	0	0.1	0–1

TABLE 3 | The data used to calculate humpback whale bycatch in the OA sablefish pot fishery collected by fishery observers (observed) and the estimated mean and 95% credible interval (CI: lower–upper) from the best model. The best model included a constant bycatch rate, a Poisson count process, and used the total weight (mt) of landings from all trips that carried an observer (observed) as fishing effort.

Year	Sector	Observed			Estimated bycatch	
		Landings (mt)	Landings (%)	Bycatch (#)	Mean (#)	CI (95%)
2003	OA	2.9	2	0	1.0	0–4
2004	OA	17.0	9	0	1.2	0–5
2005	OA	10.7	3	0	2.5	0–9
2006	OA	7.9	2	0	2.8	0–10
2007	OA	8.8	3	0	2.1	0–7
2008	OA	10.4	4	0	1.8	0–7
2009	OA	8.8	2	0	3.2	0–12
2010	OA	10.7	3	0	2.5	0–9
2011	OA	18.9	7	0	1.9	0–7
2012	OA	9.1	7	0	1.0	0–4
2013	OA	6.3	9	0	0.5	0–3
2014	OA	11.7	8	0	1.0	0–4
2015	OA	14.6	7	0	1.4	0–6
2016	OA	15.3	7	1	2.5	1–7
2017	OA	24.9	12	0	1.3	0–5
2018	OA	17.2	10	0	1.1	0–5
2019	OA	16.7	11	0	1.0	0–4

Poisson distribution. Because the deep stratum for both sectors had no observed takes, we focused on results for the shallow stratum. Estimated observed bycatch rates for both sectors highlighted that the expected observed takes using all depths were very similar to expected observed takes using only the shallow depths where takes had been observed (**Supplementary Figure 2**). For the LE sector during the 2012–2013 period, estimates using all depths were slightly greater than using only the shallow depths, but these differences were because of a sharp reduction in retained catch in the shallow sector during this period (**Supplementary Figure 1**). The posterior distribution of total takes (observed + posterior predictive distribution of unobserved takes) was also similar between depth scenarios (**Supplementary Figure 3**). The LE sector estimates

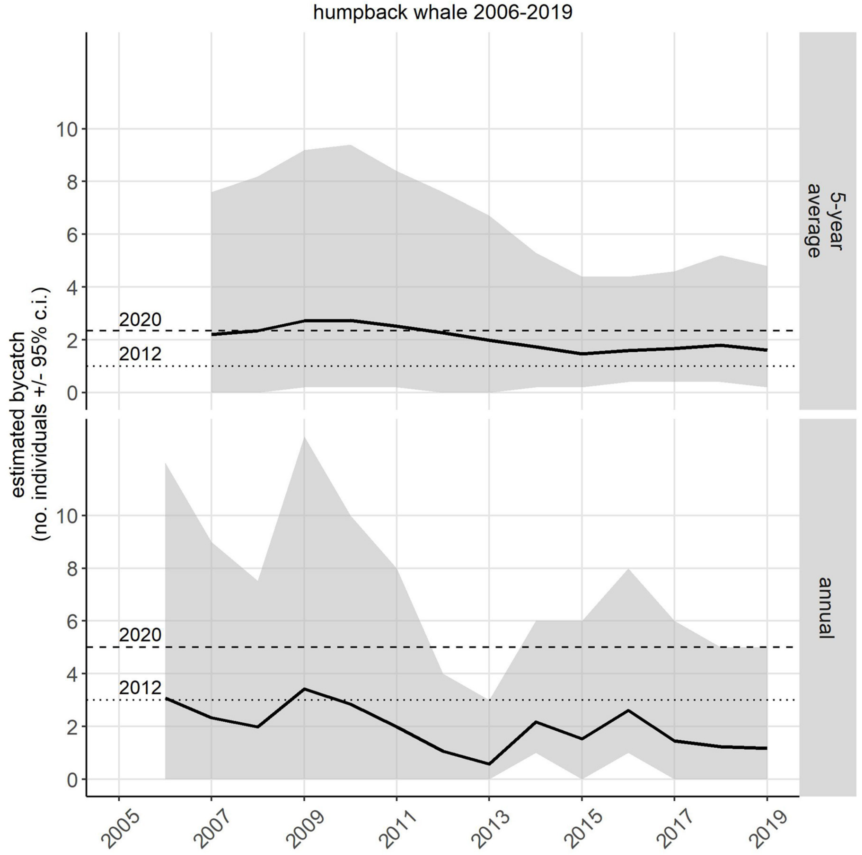


FIGURE 1 | Estimated bycatch [number of individuals, 95% credible interval (CI)] of humpback whales in the United States West Coast sablefish pot fishery. Estimates were made for LE and OA sectors separately and then the estimates from the two sectors were summed. The two Biological Opinions (2012 and 2020) governing the incidental take of humpback whales in this fishery specified a 5-year average (top panel) in 2012 (dotted line) as 1 whale/year, and 2.34 whales/year in 2020. The specified annual take threshold (bottom panel) in 2012 (dotted line) was 3 whales/year and in 2020 (dashed line) was 5 whales/year.

ranged from 0 to slightly more than 1 animal irrespective of the depth stratum and the OA sector estimates ranged from less than 1 to slightly more than 2.5 animals, irrespective of depth grouping (**Supplementary Figure 3**). In both sectors, estimates were largest in the years when observed takes occurred, again irrespective of the depth grouping (**Supplementary Figure 3**). The 5-year average of takes and probabilities of exceeding the 2020 5-year threshold of 2.34 takes/year (LE + OA) also appeared to be insensitive to the depth grouping, with similar trajectories in estimates from both depth scenarios (**Supplementary Figure 4**).

DISCUSSION

This work demonstrates how Bayesian models can be used to estimate rare bycatch events: in this case, humpback whale entanglements in the United States West Coast sablefish pot fishery. This approach can more accurately estimate bycatch and uncertainty than other methods (e.g., ratio estimators; Lewin et al., 2010; Carretta and Moore, 2014; Martin et al., 2015) and yields probabilities for unobserved entanglements. Our simulations show that when data are rare but not over-dispersed, simpler bycatch-generating processes (i.e., Poisson) are favored over more complex distributions (e.g., negative binomial). However, as expected, when data are rare and over-dispersed, more complex distributions need to be employed, but the precise distribution depends on the sparsity of the data (e.g., zero-inflated vs. negative binomial). For 2019, the most recent estimates available, a Poisson process estimated a 5-year average of 1.60 entanglements/year for 2015–2019, for both sectors combined, which is less than the 2020 5-year average threshold of 2.34 entanglements/year. However, the uncertainty around that 5-year average estimate suggests that entanglements could be as high as 4.8/year. Our analysis also suggests that the results are not sensitive to assumptions about the data. Splitting the data by depth and comparing the model using all depths to the model using only the shallow depth stratum demonstrated that bycatch rates, posterior predictive distributions, and the probability of exceeding the 2020 threshold did not depend on these stratification choices.

The NMFS began using Bayesian models to estimate humpback whale bycatch in the United States West Coast sablefish pot fishery in 2019. That analysis resulted in estimates

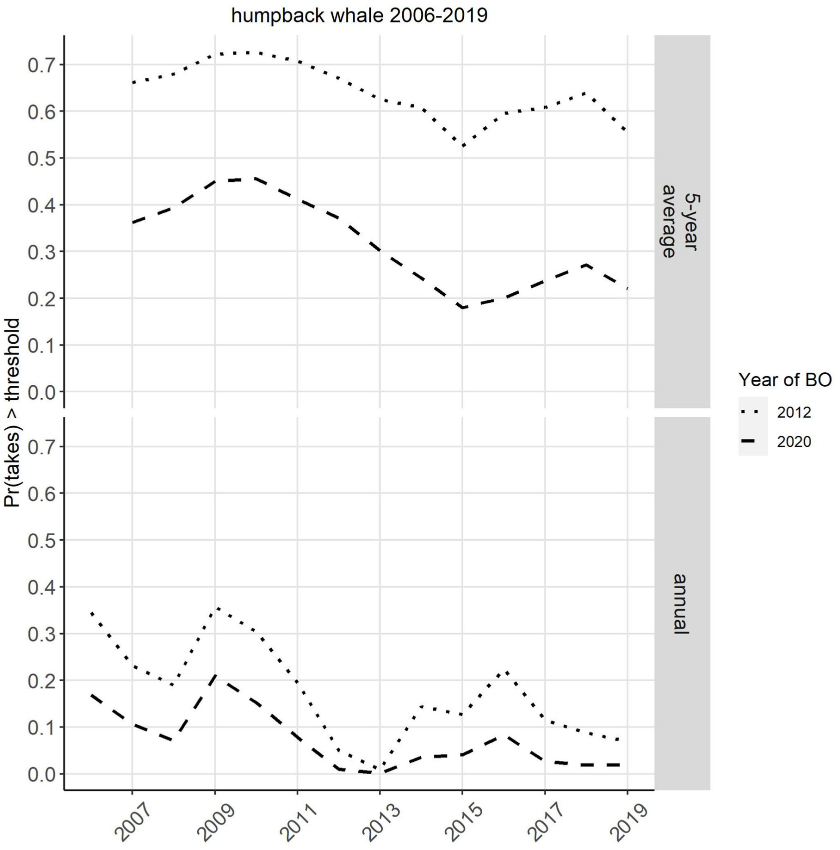


FIGURE 2 | Probability of the 5-year average take estimate (top) or annual take estimate (bottom) exceeding the take threshold specified in the 2012 (dotted line) or 2020 (dashed line) Biological Opinion (BO) Incidental Take Statement. Currently these fisheries operate under the 2020 BO, which superseded the 2012 BO.

of humpback whale bycatch in the fishery that exceeded the previously anticipated bycatch limits, in part, contributing to a new review of the fishery and humpback bycatch risk under the ESA. While the California/Oregon/Washington stock showed long-term increases in abundance from 1990 to 2008, estimates from 2008 to 2014 suggest a period of leveling-off, but data from 2014 to 2018 suggest another period of population growth along the United States West Coast (Carretta et al., 2020b; Calambokidis and Barlow, 2020). The most recent Potential Biological Removal (PBR) for humpback whales in United States waters is 16.7 whales/year (Carretta et al., 2020b). Estimates of mortality in the LE sablefish pot fishery are generally less than one whale per year (**Table 1**), whereas estimates of humpback mortality from the OA Fixed Gear pot fishery are between one and three whale mortalities per year (**Table 2**). Together, these two fisheries represent about 10% of the total PBR.

The goal of the MMPA is to reduce incidental mortality and serious injury of all marine mammals to insignificant levels approaching a zero rate. This goal has been defined as the threshold for mortality and serious injury at 10% of PBR for a stock of marine mammals (69 FR 23477). As a result, we estimate that the mortality and serious injury associated with the sablefish pot fishery is approaching this threshold by itself, while numerous other sources of mortality and serious injury are also occurring associated with other fishery and non-fishery sources. For example, the number of confirmed humpback whale entanglements from all sources was on the rise from 16 confirmed entanglements in 2014 to 48 confirmed in 2016 (Santora and Lawson, 2021), coinciding with the time period in which the LE and OA sablefish pot fisheries each recorded a humpback whale entanglement. Since 2016, confirmed humpback whale entanglements have been between 17 and 34/year (Santora and Lawson, 2021).

Analytical Challenges

Despite the flexibility of the models presented here, there still remain several challenges and limitations to this method. For the two sectors presented here, we were able to successfully compare models in terms of bycatch rates (time-varying vs. constant) and some processes (Poisson vs. negative binomial). However, this is unlikely to always be the case, and indeed we were unable to compare all possible processes (e.g., zero-inflated) using the humpback data and have encountered other rare bycatch modeling scenarios where direct comparison was impossible as none of the models passed the diagnostic criteria (Jannot et al., 2021). In the case where all models failed, yet bycatch estimates were still required for management purposes, we chose the simplest form of the models (constant rate, Poisson)

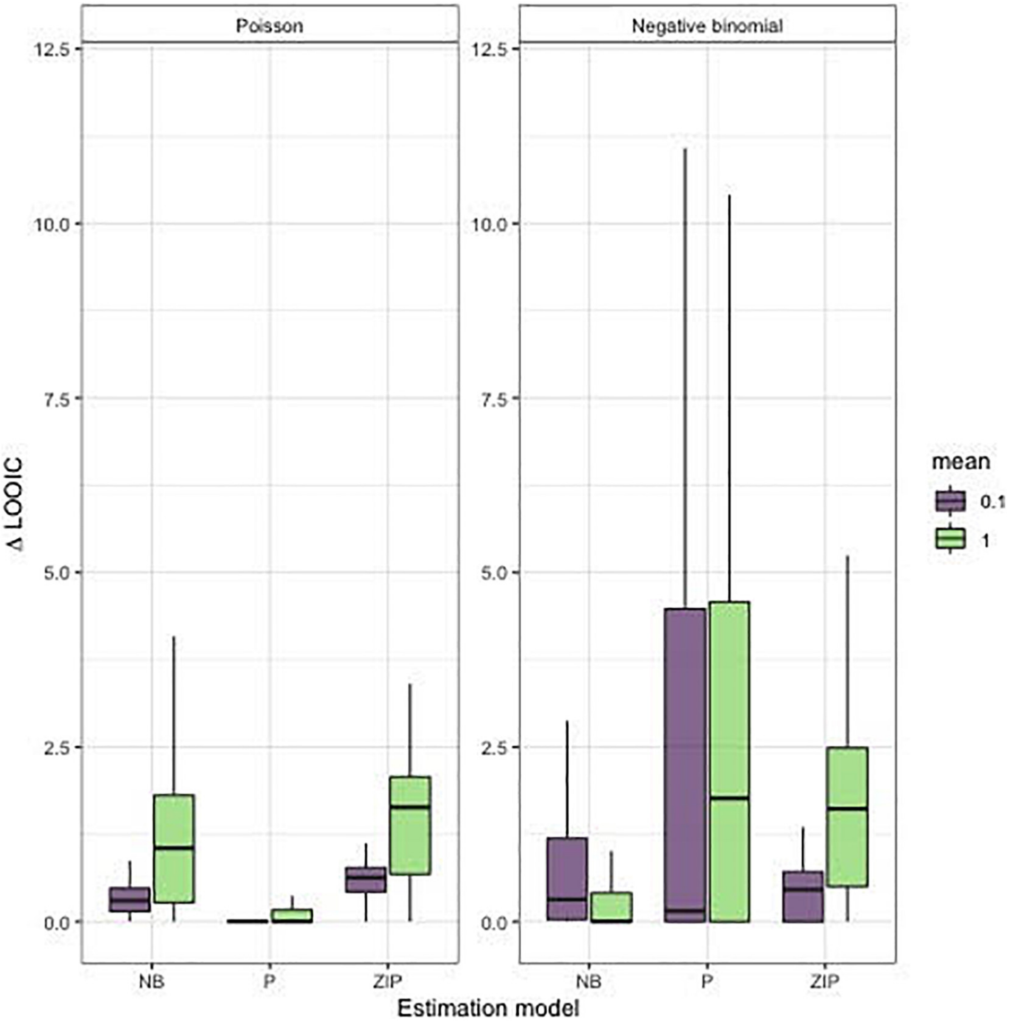


FIGURE 3 | Distribution of LOOIC differences for our model selection applied to simulated time series. Each boxplot represents 100 time series generated from each level of the mean bycatch rate (0.1, 1), estimation model ("NB," negative binomial; "P," Poisson; "ZIP," zero-inflated Poisson), and simulation model (Poisson, negative binomial).

and compared among effort metrics (Jannot et al., 2021). None-the-less, the specific nature of the data sparsity and dispersion can hamper the optimal performance of the models and limit our understanding of bycatch.

Another limitation of this method is the nature of the data itself. In the examples we have provided, we rely almost exclusively on observer data. In the LE and OA sectors presented here, we have a plethora of information from observed vessels that could provide insight into bycatch, such as multiple proxy metrics of fishing effort (# pots, weight of landed catch), depth, latitude, and duration as well as others. However, we have much less comparable data from the unobserved portion of the fleet which limits our understanding of the causes of bycatch, because, for example, we have to assume that the fishing depth distribution is similar among observed and unobserved vessels. The "observer effect" posits that observed vessels can behave quite differently than unobserved vessels (Hilborn et al., 2009; Faunce and Barbeaux, 2011). Therefore, any inferences about humpback whale bycatch must be tempered by the limited data available from unobserved vessels and the potential for an observer effect.

Fisheries that are similar but have no observed recorded takes also pose a challenge to this method. As mentioned above, there is a third pot sector, the CS pot fishery. Since 2011, this fishery has had 100% monitoring of catch at-sea. During 2011–2014 all CS pot fishery trips were monitored by fisheries observers. Since 2015, while all trips are monitored, only roughly 50% of trips have been monitored by observers and the remainder of trips are monitored by EM (Somers et al., 2020a). To date, the CS pot fishery has not recorded a humpback whale entangled in pot gear since the fishery inception in 2011. Despite the fishery being 100% monitored, there might still be unaccounted for bycatch. For example, a humpback whale could become entangled in pot gear while the gear was fishing, but unattended, and then swim away with gear attached and therefore, be unobserved and undocumented, which is a plausible scenario (see section "Unobserved Bycatch and Cryptic Mortality"). To estimate bycatch in the CS fishery, we would have to assume that bycatch rates are shared from either the LE or the OA fishery, either by taking an average across the sectors, or using a more precautionary approach, assuming the higher of the two bycatch rates. While the gear used (pots, lines, and floats) in the CS pot fishery is similar to the gear used in the LE and OA fisheries, CS vessels do not fish precisely the same as either fleet. For example, the LE fleet targets sablefish during a season (April–October) whereas the CS fleet can fish all year. In terms of fishing effort, CS vessels generally deploy similar numbers of pots as LE vessels, which is more pots per gear deployment than OA vessels. Also the CS fleet holds quota for other species besides groundfish, whereas the LE and OA fleets can only land other groundfish up to species-specific trip limits. In this way, the problem is analogous to the observer effect, to make estimates for the CS fleet, we would have to make untested assumptions about fishing effort and the manner in which pots are fished in the CS fleet based on information from the LE or OA fleet.

Unobserved Bycatch and Cryptic Mortality

One of the advantages to the method we present here is that it accounts for unobserved entanglements. Estimating unobserved entanglements is particularly important in the case of west coast humpback whales. Observers do not detect all humpback whales that have been entangled in sablefish pot gear, given that whales entangled in this gear type have been opportunistically reported at various locations off the west coast (Saez et al., 2021). Entanglements might go unobserved for multiple reasons. For example, observers may not be present on a trip when an entanglement occurs (only a portion of trips are observed; **Tables 1**, **2**). Whales could break free of the entanglement before observation (Saez et al., 2021), and entangled whales could then leave the fishing area with gear attached. In all three sectors (LE, OA, and CS), it is common for vessels (with or without observers present) to deploy gear at the fishing grounds and then leave the area to let the gear fish. OA pot vessels very often place the gear in a single location and then return throughout the year during days of good weather to retrieve the catch, resetting the gear back to its original location. In these cases, the gear is fishing unattended for a period of time, which can vary from a few hours or days potentially up to weeks (a.k.a. soak time). During these long soak times, whales could become entangled in gear and swim away without being observed.

Between 2006 and 2017, there were five confirmed reports of humpback entanglements with sablefish pot gear (Saez et al., 2021). These five observations are considered a minimum estimate, due to the opportunistic nature of reporting at-sea entanglements. These opportunistic observations are likely biased because they are not a random selection of observations in space or time. Thus, these observations cannot be directly incorporated into the models presented here. A total of 17 opportunistic records of humpback whale entanglements in fishing gear were reported in 2019. In the 2020 Biological Opinion, a very small percentage (less than 5%) of entanglements from 2011 to 2019 that were attributed to a fishery were associated with sablefish pot gear (NMFS, 2020a). Seven of the 17 reports could be attributed to a fishery but none (0) of those were associated with sablefish gear. The remaining 10 entanglement reports could not be attributed to a specific fishery. Based on this information, we assume that no more than one of the 10 entanglements with unidentified fishing gear would be expected to be associated with sablefish gear. The Bayesian framework generates posterior predictive distributions for unobserved entanglements in addition to estimates and associated uncertainty for observed entanglements. Therefore, estimates from these models do account for these unobserved entanglements.

Assessing the number of unobserved pot or trap gear entanglements of humpback whales from any fishery on the United States West Coast is difficult due to the nature of opportunistic reports (i.e., non-random) and the rarity of systematically observed incidents (human or EM). Undetected, a.k.a., cryptic, injury and mortality of marine mammals is challenging to estimate, but progress has been made for several populations (Williams et al., 2011; Peltier et al., 2012; Prado et al., 2013; Wells et al., 2015; Carretta et al., 2016; Young et al., 2019; Harting et al., 2021; Pace et al., 2021). Marine mammal carcass recovery rates (= detection rates) have been estimated with several approaches: tracking the fate of known individuals over time (Wells et al., 2015); combining abundance estimates and estimated annual survival in Monte Carlo simulations to estimate carcass numbers available for detection (Carretta et al., 2016; Harting et al., 2021), comparing observed stranding numbers to estimated mortalities from population models (Pace et al., 2021), comparing numbers of marked carcasses at sea with those arriving ashore (Prado et al., 2013) and using drift models to estimate the fraction of carcasses arriving ashore (Peltier et al., 2012; Young et al., 2019). Generally, published estimates of carcass recovery rates are quite low, ranging from near-zero for some pelagic species such as killer whales and false killer whales (Williams et al., 2011), <10% for common dolphins (Peltier et al., 2012), 33% for an embayment population of coastal bottlenose dolphins (Wells et al., 2015), 36% for North Atlantic right whales (Pace et al., 2021), and 46% for Hawaiian monk seals (Harting et al., 2021). Most species lack estimates of undetected mortality and serious injury and, for pelagic species, at-sea sightings and strandings provide minimum accounting of human-related mortality and serious injury due to low probabilities of stranding and detection (Faerber and Baird, 2010; Williams et al., 2011).

Our bycatch model includes estimates of interactions in unobserved portions of the sablefish fishery, but both observed and modeled interaction rates are based on time windows when observers are present. These interaction rates may be negatively biased because they exclude unobserved cases where whales swam off with gear and subsequently incur serious injury or die from chronic entanglement over a period of months (Moore and van der Hoop, 2012). Estimates of interaction rates also exclude unobserved cases of whales becoming entangled in lost gear, which is a special case of "fishing effort" outside the purview of observer programs. Baseline data on levels of lost gear generally shows that $<2\%$ of pot gear is lost (**Supplementary Table 1**), but assessment of how and why gear is lost (rough weather vs. whales swimming away with gear) is difficult.

Other United States West Coast Fisheries

From 2011 to 2019, NMFS received and evaluated 170 separate confirmed humpback whale entanglement reports from the United States West Coast (excluding re-sightings; NMFS, 2019; NWFSC, 2020b; Saez et al., 2021). With the limited exception of a few reports from fishery observers, including the two reports from sablefish fishery observers assessed here, most of these reports are products of opportunistic, i.e., non-random, sightings, and reportings from sources that include marine mammal stranding and response networks, members of the public, United States Coast Guard, law enforcement agencies, and marine researchers. Given that the majority of entanglement reports are opportunistic, NMFS assumes that many large whale entanglements are not observed or, if observed, are not likely to be reported except as required by fisheries observers or EM programs. Therefore, it is likely that reports of large whale entanglements represent an unknown fraction of the total number of whales that have been entangled over time (Saez et al., 2021). Currently, the number of total whale entanglements that occur along the United States West Coast relative to the number of entanglements reported is unknown (Saez et al., 2021). However, rope scarring from entanglements with fishing gear are evident on one third to one half of all humpback whales (Calambokidis et al., 2008), which may provide insight on the total number of whales that have been entangled at least once.

Numerous other United States West Coast commercial and recreational fisheries have been associated with the origins of whale entanglements reported through opportunistic sources, including Dungeness crab, spot prawn, spiny lobster, and rock crab pot or trap fisheries, along with various set and drift gillnet fisheries (Saez et al., 2021). A cursory review of the literature provided no other examples of the use of this method in fixed gear fisheries to estimate large whale bycatch. Many of these fisheries do not employ use of fishery monitoring schemes like the sablefish pot fishery presented here. Unobserved fisheries pose a challenge to estimating large whale bycatch. The Bayesian method we present here relies on systematically collected random samples of lethal entanglements in fishing gear, a measure of observation effort, and a measure of total fishing effort for the entire fleet. For the United States West Coast sablefish pot fishery we used data from fishery monitoring programs (e.g., human observers and EM) that systematically collect data on whale entanglements. For unmonitored fisheries, observed bycatch rates (θ) could be borrowed from observed fisheries and applied to unobserved fisheries. However, this would require having some measure of observed effort for the entanglements (E_y) which is unlikely in unmonitored fisheries. In many unmonitored fisheries, the only available data will be a count of entanglements reported and some measure of total fishing effort (number of: vessels, gear deployments, pot or traps; gear soak time; and total fleet landings). Entanglement estimates in most cases will not be collected systematically and therefore not related to observational effort in any meaningful way, making the Bayesian method presented here less than ideal for unmonitored fisheries. However, one case where our method might prove useful are species such as the North Atlantic right whale, where a large proportion of the population has been observed and identified by photograph (Knowlton et al., 2012). In theory, if observation and entanglement rates could be constructed from photographs (Knowlton et al., 2012) and entangling gear appropriately assigned to a fishery with available data on fishing effort, then bycatch estimates could be obtained for that fishery.

CONCLUSION

Large whale entanglement and bycatch in fishing gear presents a challenge to analysts and managers that need to estimate the number of, and mitigate for, these low frequency events. Currently, the most robust data estimates of whale entanglements mainly comes from at-sea fishery observers or EM devices. However, not all fisheries are monitored and for those fisheries that are monitored, it is often the case that not all vessels within a fishery are observed. The Bayesian method used here provides robust estimates of both observed and unobserved bycatch in partially monitored fisheries, thus overcoming some of the challenges posed by rare event data. This method is flexible and can be used on a wide-variety of commonly used generalized linear models and provides reasonable estimates of uncertainty around bycatch estimates as well as accounting for undetected bycatch by providing estimates of bycatch when the observed estimate is zero. More work needs to be done to develop methods that use non-random opportunistic observations of whale entanglement. However, the Bayesian time-series used here provides managers and analysts with an important tool to accurately assess the impacts of fishing on large whales.

DATA AVAILABILITY STATEMENT

The datasets presented in this article are not readily available because of the confidential nature of the NOAA west coast sablefish fishery observer data. All data to replicate our analyses is included in the tables in the article, and code is provided *via* our public R package. The raw data (e.g., at the level of individual sets) for the case studies presented here are only available upon request from the NOAA NWFSC FOS Program. Requests to access the datasets should be directed to JJ, jason.jannot@noaa.gov.

AUTHOR CONTRIBUTIONS

JJ and EW conceptualized the study, conducted the analyses, and created the tables and figures. JJ and KS prepared, processed, and summarized the data. JJ and EW wrote the manuscript with contributions from KS, BF, TG, DL, and JC. All authors contributed to manuscript revision, and read and approved the submitted version.

FUNDING

NOAA Fisheries Northwest Fisheries Science Center supported JJ, EW, KS, BF, and TG. NOAA Fisheries West Coast Regional Office supported DL. NOAA Fisheries Southwest Fisheries Science Center supported JC. Funding from the Northwest Fisheries Science Center Fisheries Observation Science Program and Conservation Biology Division were used for publication fees.

ACKNOWLEDGMENTS

We gratefully acknowledge the hard work and dedication of observers from the West Coast Groundfish Observer Program and the At-Sea Hake Observer Program, as well as contributions from observer program staff. We would like to thank the NOAA West Coast Strandings Network Coordinators: Kristin Wilkinson, Brent Norberg, Justin Viezbicke, and Justin Greenman for providing strandings data. We would also like to thank Jameal Samhouri and three reviewers for constructive comments on earlier versions of this manuscript.

SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2021.775187/full#supplementary-material

REFERENCES

Amande, M. J., Chassot, E., Chavance, P., Murua, H., De Molina, A. D., and Bez, N. (2012). Precision in bycatch estimates: the case of tuna purse-seine fisheries in the Indian Ocean. *ICES J. Mar. Sci.* 69, 1501–1510. doi: 10.1093/icesjms/fss106

Babcock, E. A., Pikitch, E. K., and Hudson, C. G. (2003). *How much Observer Coverage is Enough to Adequately Estimate Bycatch?*. Washington, DC: Pew Institute for Ocean Science and Oceana. doi: 10.1111/j.1748-7692.2010.00444.x

Bettridge, S. O. M., Baker, C. S., Barlow, J., Clapham, P., Ford, M. J., Gouveia, D., et al. (2015). *Status Review of the Humpback Whale (Megaptera novaeangliae) Under the Endangered Species Act*. La Jolla, CA: U. S. Department Of Commerce.

Borges, L., Zuur, A. R., Rogan, E., and Officer, R. (2005). Choosing the best sampling unit and auxiliary variable for discards estimations. *Fish. Res.* 75, 29–39. doi: 10.1016/j.fishres.2005.05.002

Breiman, L. (2001). Statistical modeling: the two cultures (with comments and a rejoinder by the author). *J. Stat. Sci.* 19:133. doi: 10.1214/ss/1009213726

Calambokidis, J., and Barlow, J. (2020). *Updated Abundance Estimates for Blue and Humpback Whales Along the U.S. West Coast using data through 2018*. NOAA Technical Memorandum NMFS-SWFSC-634. La Jolla, CA: U.S. Department of Commerce.

Calambokidis, J., Falcone, E. A., Quinn, T. J., Burdin, A. M., Clapham, P. J., Ford, J. K. B., et al. (2008). *SPLASH: Structure of Populations, Levels of Abundance and Status of Humpback Whales in the North Pacific*. Seattle, WA: U.S. Dept of Commerce - Western Administrative Center, 57.

Carretta, J. V., Danil, K., Chivers, S. J., Weller, D. W., Janiger, D. S., Berman-Kowalewski, M., et al. (2016). Recovery rates of bottlenose dolphin (*Tursiops truncatus*) carcasses estimated from stranding and survival rate data. *Mar. Mamm. Sci.* 32, 349–362. doi: 10.1111/mms.12264

Carretta, J. V., Delean, B., Helker, V. T., Muto, M., Greenman, J., Wilkinson, K. M., et al. (2020a). *Sources of Human-Related Injury and Mortality for U.S. Pacific West Coast Marine Mammal Stock Assessments, 2014-2018*. NOAA Tech. Memo. NOAA-TM-NMFS-SWFSC-631. La Jolla, CA.

Carretta, J. V., Forney, K. A., Oleson, E. M., Weller, D. W., Lang, A. R., Baker, J., et al. (2020b). *U.S. Pacific Marine Mammal Stock Assessments: 2019*. NOAA Tech. Memo. NOAA-TM-NMFS-SWFSC-629. La Jolla, CA.

Carretta, J. V., and Moore, J. E. (2014). *Recommendations for Pooling Annual Bycatch Estimates When Events are Rare*. NOAA Tech. Memo. NOAA-TM-NMFS-SWFSC-528. La Jolla, CA.

Carretta, J. V., Moore, J. E., and Forney, K. A. (2017). *Regression Tree and Ratio Estimates of Marine Mammal, Sea Turtle, and Seabird Bycatch in the California Drift Gillnet Fishery, 1990-2015*. NOAA Tech. Memo. NOAA-TM-NMFS-SWFSC-568. La Jolla, CA.

Cosandey-Godin, A., Krainski, E. T., Worm, B., and Flemming, J. M. (2015). Applying Bayesian spatiotemporal models to fisheries bycatch in the Canadian Arctic. Can. J. Fish. Aquat. Sci. 72, 186–197.

Curtis, K. A., and Carretta, J. V. (2020). ObsCovgTools: assessing observer coverage needed to document and estimate rare event bycatch. *Fish. Res.* 225:105493. doi: 10.1016/j.fishres.2020.105493

Dixon, P. M., Ellison, A. M., and Gotelli, N. J. (2005). Improving the precision of estimates of the frequency of rare events. *Ecology* 86, 1114–1123. doi: 10.1890/04-0601

Faerber, M. M., and Baird, R. W. (2010). Does a lack of observed beaked whale strandings in military exercise areas mean no impacts have occurred? A comparison of stranding and detection probabilities in the Canary and main Hawaiian Islands. *Mar. Mamm. Sci.* 26, 602–613. doi: 10.1111/j.1748-7692.2010.00370.x

Faunce, C. H., and Barbeaux, S. J. (2011). The frequency and quantity of Alaskan groundfish catcher-vessel landings made with and without an observer. *ICES J. Mar. Sci.* 68, 1757–1763. doi: 10.1093/icesjms/fsr090

Gilman, E., Suuronen, P., Hall, M., and Kennelly, S. (2013). Causes and methods to estimate cryptic sources of fishing mortality. *Fish. Biol.* 83, 766–803. doi: 10.1111/jfb.12148

Harting, A. L., Barbieri, M. M., Baker, J. D., Mercer, T. A., Johanos, T. C., Robinson, S. J., et al. (2021). Population-level impacts of natural and anthropogenic causes-of-death for Hawaiian monk seals in the main Hawaiian Islands. *Mar. Mamm. Sci.* 37, 235–250. doi: 10.1111/mms.12742

Hennig, C. (2020). *fpc: Flexible Procedures for Clustering. R Package Version 2.2-9*. Available online at: https://CRAN.R-project.org/package=fpc (accessed December 06, 2020).

Hilborn, R., Benoît, H. P., and Allard, J. (2009). Can the data from at-sea observer surveys be used to make general inferences about catch composition and discards? *Can. J. Fish. Aquatic Sci.* 66, 2025–2039. doi: 10.1139/F09-116

Jannot, J. E., Wuest, A., Good, T. P., Somers, K. A., Tuttle, V. J., Richerson, K. E., et al. (2021). *Seabird Bycatch in U.S. West Coast Fisheries, 2002–18*. NOAA Technical Memorandum NMFS-NWFSC-165. Seattle, WA: U.S. Department of Commerce.

Knowlton, A. R., Hamilton, P. K., Marx, M. K., Pettis, H. M., and Kraus, S. D. (2012). Monitoring North Atlantic right whale Eubalaena glacialis entanglement rates: a 30 yr. retrospective. *Mar. Ecol. Prog. Ser.* 466, 293–302. doi: 10.3354/meps09923

Komoroske, L. M., and Lewison, R. L. (2015). Addressing fisheries bycatch in a changing world. *Front. Mar. Sci.* 2:83. doi: 10.3389/fmars.2015.00083

Lewin, W.-C., Freyhof, J., Huckstorf, V., Mehner, T., and Wolter, C. (2010). When no catches matter: coping with zeros in environmental assessments. *Ecol. Indic.* 10, 572–583. doi: 10.1016/j.ecolind.2009.09.006

Martin, S. L., Stohs, S. M., and Moore, J. E. (2015). Bayesian inference and assessment for rare-event bycatch in marine fisheries: a drift gillnet fishery case study. *Ecol. Appl.* 25, 416–429. doi: 10.1890/14-0059.1

McCracken, M. L. (2004). *Modeling a Very Rare Event to Estimate Sea Turtle Bycatch Lessons Learned*. NOAA Tech. Memo. NMFS-PIFSC-3. Honolulu, HI.

Moore, J. E., Curtis, K. A., Lewison, R. L., Dillingham, P. W., Cope, J. M., Fordham, S. V., et al. (2013). Evaluating sustainability of fisheries bycatch mortality for marine megafauna: a review of conservation reference points for data-limited populations. *Environ. Conserv.* 40, 329–344. doi: 10.1017/S037689291300012X

Moore, J. E., Merrick, R. L., Barlow, J., Bettridge, S. O. M., Carretta, J. V., Eagle, T. C., et al. (2011). *Guidelines for Assessing Marine Mammal Stocks: Report of the GAMMS III Workshop, February 15-18, 2011*. La Jolla, CA: National Marine Fisheries Service.

Moore, M. J., and van der Hoop, J. M. (2012). The painful side of trap and fixed net fisheries: chronic entanglement of large whales. *J. Mar. Sci.* 2012:230653. doi: 10.1155/2012/230653

NMFS (1991). *Final Recovery Plan for the Humpback Whale (Megaptera novaeangliae). Prepared by the Humpback Whale Recovery Team for the National Marine Fisheries Service*. Silver Spring, MD: National Marine Fisheries Service, 105.

NMFS (2004). *Evaluating bycatch: A National Approach to Standardized Bycatch Monitoring Programs*. NOAA Tech. Memo. NMFSF/SPO-66. Silver Springs, MD: U.S. Department of Commerce, 108.

NMFS (2012). *Endangered Species Act (ESA) Section 7(a)(2) Biological and Section 7(a)(2) "Not Likely to Adversely Affect" Determination Continuing Operation of the Pacific Coast Groundfish Fishery*. West Coast Region: National Marine Fisheries Service.

NMFS (2019). *2018 West Coast Whale Entanglement Report. Spring 2019*. Long Beach, CA: NOAA Fisheries West Coast Region, 10.

NMFS (2020a). *Endangered Species Act (ESA) Section 7(a)(2) Biological and Conference Opinion Continuing Operation of the Pacific Coast Groundfish Fishery*. West Coast Region: National Marine Fisheries Service.

NMFS (2020b). 2019 West Coast Whale Entanglement Summary. Spring 2020. Long Beach, CA: NOAA Fisheries West Coast Region, 4.

NWFSC (2020c). *Training Manual West Coast Groundfish Observer Program*. Seattle, WA: Northwest Fisheries Science Center.

Pace, R. M. III, Williams, R., Kraus, S. D., Knowlton, A. R., and Pettis, H. M. (2021). Cryptic mortality of North Atlantic right whales. *Conserv. Sci. Pract.* 3:e346. doi: 10.1111/csp2.346

Parsa, M., Emery, T. J., Williams, A. J., and Nicol, S. (2020). An empirical Bayesian approach for estimating fleet- and vessel-level bycatch rates in fisheries with effort heterogeneity and limited data: a prospective tool for measuring bycatch mitigation performance. *ICES J. Mar. Sci.* 77, 921–929. doi: 10.1093/icesjms/fsaa020

Peltier, H., Dabin, W., Daniel, P., Van Canneyt, O., Dorémus, G., Huon, M., et al. (2012). The significance of stranding data as indicators of cetacean populations at sea: modelling the drift of cetacean carcasses. *Ecol. Indic.* 18, 278–290. doi: 10.1016/j.ecolind.2011.11.014

Prado, J. H. F., Secchi, E. R., and Kinas, P. G. (2013). Mark-recapture of the endangered franciscana dolphin (*Pontoporia blainvillei*) killed in gillnet fisheries to estimate past bycatch from time series of stranded carcasses in southern Brazil. *Ecol. Indic.* 32, 35–41. doi: 10.1016/j.ecolind.2013.03.005

R Core Team (2020). *R: A Language and Environment for Statistical Computing*. Vienna: R Foundation for Statistical Computing.

Rochet, M. J., and Trenkel, V. M. (2005). Factors for the variability of discards: assumptions and field evidence. *Can. J. Fish. Aquatic Sci.* 62, 224–235. doi: 10.1139/f04-185

Saez, L., Lawson, D., and DeAngelis, M. L. (2021). *Large whale Entanglements off the U.S. West Coast, from 1982-2017*. Silver Spring, MD: United States Department of Commerce, 50.

Santora, J., and Lawson, D. (2021). *IEA Integrated Ecosystem Assessment California Current Integrated Ecosystem Assessment Ecosystem Context for Reducing West Coast Whale Entanglements California Current Project [Online]*. Available online at: https://www.integratedecosystemassessment.noaa.gov/regions/california-current/cc-projects-whale-entanglement (accessed August 30, 2021)

Siders, Z. A., Ducharme-Barth, N. D., Carvalho, F., Kobayashi, D., Martin, S., Raynor, J., et al. (2020). Ensemble random forests as a tool for modeling rare occurrences. *Endang. Species Res.* 43, 183–197. doi: 10.3354/esr01060

Somers, K. A., Jannot, J. E., Richerson, K., Tuttle, V. J., and Mcveigh, J. (2020a). *Fisheries Observation Science Program Coverage Rates, 2002-19. U.S. Dep. Commer*. NOAA Data Report NMFS-NWFSC-DR-2020-03. Seattle, WA.

Somers, K. A., Whitmire, C. E., Richerson, K., Jannot, J. E., Tuttle, V. J., and Mcveigh, J. (2020b). *Fishing Effort in the 2002-17 Pacific Coast Groundfish Fisheries*. NOAA Technical Memorandum NMFS-NWFSC-153. Seattle, WA.

Stan Development Team (2021). *Stan Modeling Language Users Guide and Reference Manual, 2.27*. Available online at: https://mc-stan.org.

Stock, B. C., Ward, E. J., Eguchi, T., Jannot, J. E., Thorson, J. T., Feist, B. E., et al. (2020). Comparing predictions of fisheries bycatch using multiple spatiotemporal species distribution model frameworks. *Can. J. Fish. Aquatic Sci.* 77, 146–163. doi: 10.1139/cjfas-2018-0281

Stratoudakis, Y., Fryer, R., Cook, R., and Pierce, G. (1999). Fish discarded from Scottish demersal vessels: estimators of total discards and annual estimates for targeted gadoids. *ICES J. Mar. Sci.* 56:592. doi: 10.1006/jmsc.1999.0500

Vehtari, A., Gabry, J., Magnusson, M., Yao, Y., Bürkner, P., Paananen, T., et al. (2020). *"loo: Efficient leave-one-out cross-validation and WAIC for Bayesian models". R package version 2.2.0*.

Vehtari, A., Simpson, D., Gelman, A., Yao, Y., and Gabry, J. (2019). Pareto smoothed importance sampling. *arXiv*[Preprint] arXiv:1507.02646,

Wade, P. R. (1998). Calculating limits to the allowable human-caused mortality of cetaceans and pinnipeds. *Mar. Mamm. Sci.* 14, 1–37. doi: 10.1111/j.1748-7692.1998.tb00688.x

Wakefield, C. B., Hesp, S. A., Blight, S., Molony, B. W., Newman, S. J., and Hall, N. G. (2018). Uncertainty associated with total bycatch estimates for rarely-encountered species varies substantially with observer coverage levels: informing minimum requirements for statutory logbook validation. *Mar. Policy* 95, 273–282. doi: 10.1016/j.marpol.2018.05.018

Walmsley, S., Leslie, R., and Sauer, W. (2007). Bycatch and discarding in the South African demersal trawl fishery. *Fish. Res.* 86, 15–30. doi: 10.1016/j.fishres.2007.03.002

Ward, E., and Jannot, J. (2021). *bycatch: Using Bayesian Generalized Linear Models for Estimating bycatch Rates and Generating Fleet-Level Expansions R Package (v1.0.6)*. Zenodo. doi: 10.5281/zenodo.4743823

Wells, R. S., Allen, J. B., Lovewell, G., Gorzelany, J., Delynn, R. E., Fauquier, D. A., et al. (2015). Carcass-recovery rates for resident bottlenose dolphins in Sarasota Bay, Florida. *Mar. Mamm. Sci.* 31, 355–368. doi: 10.1111/mms.12142

Wickham, H. (2016). *ggplot2: Elegant Graphics for Data Analysis*. New York, NY: Springer-Verlag. doi: 10.1007/978-3-319-24277-4

Wickham, H., Averick, M., Bryan, J., Chang, W., Mcgowan, L. D., François, R., et al. (2019). Welcome to the tidyverse. *J. Open Source Softw.* 4:1686. doi: 10.21105/joss.01686

Williams, R., Gero, S., Bejder, L., Calambokidis, J., Kraus, S. D., Lusseau, D., et al. (2011). Underestimating the damage: interpreting cetacean carcass recoveries in the context of the Deepwater Horizon/BP incident. *Conserv. Lett.* 4, 228–233. doi: 10.1111/j.1755-263X.2011.00168.x

Young, C., Eguchi, T., Ames, J. A., Staedler, M., Hatfield, B. B., Harris, M., et al. (2019). Drift and beaching patterns of sea otter carcasses and car tire dummies. *Mar. Mamm. Sci.* 35, 1512–1526. doi: 10.1111/mms.12609

Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

frontiers in Marine Science

POLICY AND PRACTICE REVIEWS
published: 17 November 2021
doi: 10.3389/fmars.2021.757330

Best Practices for Assessing and Managing Bycatch of Marine Mammals

Paul R. Wade[1*†], Kristy J. Long[2†], Tessa B. Francis[3,4], André E. Punt[4], Philip S. Hammond[5], Dennis Heinemann[6], Jeffrey E. Moore[7], Randall R. Reeves[8], Maritza Sepúlveda[9], Genoa Sullaway[4], Guðjón Már Sigurðsson[10], Margaret C. Siple[4,11], Gísli A. Víkingsson[10], Rob Williams[12] and Alexandre N. Zerbini[1,13,14]

[1] Marine Mammal Laboratory, AFSC, NOAA, National Marine Fisheries Service, Seattle, WA, United States, [2] Office of Protected Resources, NOAA, National Marine Fisheries Service, Silver Spring, MD, United States, [3] Puget Sound Institute, University of Washington Tacoma, Tacoma, WA, United States, [4] School of Aquatic and Fishery Sciences, University of Washington, Seattle, WA, United States, [5] Sea Mammal Research Unit, Scottish Oceans Institute, University of St Andrews, St Andrews, United Kingdom, [6] U.S. Marine Mammal Commission, Bethesda, MD, United States, [7] Marine Mammal and Turtle Division, SWFSC, NOAA, National Marine Fisheries Service, La Jolla, CA, United States, [8] Okapi Wildlife Associates, Hudson, QC, Canada, [9] Facultad de Ciencias, Universidad de Valparaíso, Valparaíso, Chile, [10] Pelagic Division, Marine and Freshwater Research Institute, Hafnarfjörður, Iceland, [11] Resource Assessment and Conservation Engineering Division, AFSC, NOAA, National Marine Fisheries Service Fisheries, Seattle, WA, United States, [12] Oceans Initiative, Seattle, WA, United States, [13] Cooperative Institute for Climate, Ocean and Ecosystem Studies, University of Washington, Seattle, WA, United States, [14] Marine Ecology and Telemetry Research, Seabeck, WA, United States

OPEN ACCESS

Edited by:
Lyne Morissette,
M – Expertise Marine, Canada

Reviewed by:
Simon David Goldsworthy,
South Australian Research and Development Institute, Australia
Pierre Pistorius,
Nelson Mandela University,
South Africa

***Correspondence:**
Paul R. Wade
paul.wade@noaa.gov

†These authors have contributed equally to this work and share first authorship

Specialty section:
This article was submitted to Marine Megafauna, a section of the journal Frontiers in Marine Science

Received: 12 August 2021
Accepted: 04 October 2021
Published: 17 November 2021

Citation:
Wade PR, Long KJ, Francis TB, Punt AE, Hammond PS, Heinemann D, Moore JE, Reeves RR, Sepúlveda M, Sullaway G, Sigurðsson GM, Siple MC, Víkingsson GA, Williams R and Zerbini AN (2021) Best Practices for Assessing and Managing Bycatch of Marine Mammals. Front. Mar. Sci. 8:757330. doi: 10.3389/fmars.2021.757330

Bycatch in marine fisheries is the leading source of human-caused mortality for marine mammals, has contributed to substantial declines of many marine mammal populations and species, and the extinction of at least one. Schemes for evaluating marine mammal bycatch largely rely on estimates of abundance and bycatch, which are needed for calculating biological reference points and for determining conservation status. However, obtaining these estimates is resource intensive and takes careful long-term planning. The need for assessments of marine mammal bycatch in fisheries is expected to increase worldwide due to the recently implemented Import Provisions of the United States Marine Mammal Protection Act. Managers and other stakeholders need reliable, standardized methods for collecting data to estimate abundance and bycatch rates. In some cases, managers will be starting with little or no data and no system in place to collect data. We outline a comprehensive framework for managing bycatch of marine mammals. We describe and provide guidance on (1) planning for an assessment of bycatch, (2) collecting appropriate data (e.g., abundance and bycatch estimates), (3) assessing bycatch and calculating reference points, and (4) using the results of the assessment to guide marine mammal bycatch reduction. We also provide a brief overview of available mitigation techniques to reduce marine mammal bycatch in various fisheries. This paper provides information for scientists and resource managers in the hope that it will lead to new or improved programs for assessing marine mammal bycatch, establishing best practices, and enhancing marine mammal conservation globally.

Keywords: bycatch, management, assessment, marine mammal, framework, MMPA import rule, fisheries, USA Marine Mammal Protection Act

INTRODUCTION

Human activities can intentionally or unintentionally harm marine mammals. Commercial hunting led to the decline of most species of large whales and many species of pinnipeds, and led or contributed to the extinction of a few species, namely Steller's sea cow (*Hydrodamalis gigas*), the Caribbean monk seal (*Neomonachus tropicalis*), the Japanese sea lion (*Zalophus japonicus*), and the sea mink (*Neovison macrodon*) (Le Boeuf et al., 1986; Mead et al., 2000; Turvey and Risley, 2006; Lowry, 2017). The risk to marine mammals from commercial hunting has been greatly reduced for most species since the establishment of agreements, such as the International Convention for the Regulation of Whaling, and various decisions under those conventions, such as the 1982 moratorium on commercial whaling. Now, for many species of marine mammals, the greatest threat is from fisheries bycatch, when marine mammals die from injuries sustained from becoming hooked, entrapped, or entangled in fishing gear (Read et al., 2006; Read, 2008). Many species of marine mammals have experienced severe declines in abundance caused by fisheries bycatch. For example, entanglement in fishing gear was a major contributor to the recent extinction of the baiji or Yangtze river dolphin (*Lipotes vexillifer*) (Smith et al., 2017), and this same threat is largely or entirely responsible for the Critically Endangered status and near-extinction of the vaquita (*Phocoena sinus*) (Rojas-Bracho and Taylor, 2017; Taylor et al., 2017) and the Atlantic humpback dolphin (*Sousa teuszii*) (Collins et al., 2017). Brownell et al. (2019) concluded that bycatch in gillnets is the greatest threat to most of the 13 small cetaceans presently listed as Critically Endangered on the International Union for Conservation of Nature (IUCN) Red List.

Therefore, it is important to evaluate whether the number of marine mammals killed by fisheries bycatch is leading to population declines (or impeding recovery) and this can only be accomplished by conducting an appropriate assessment. An assessment, often called a *stock* or *population assessment*, is an evaluation of the status of the population relative to management or conservation goals. Most commonly, marine mammal assessments involve quantitative methods to estimate the extent of population depletion, or to estimate how much human-caused mortality, intentional or incidental, can be allowed while achieving management or conservation goals (Wade, 2018). Moreover, it is important to develop assessment methods that are practical and can be applied worldwide given that marine mammal bycatch is a global problem.

Assessment methods for marine mammal populations have changed substantially over recent decades. In the early 1960s, the rapid decline in the numbers of whales of hunted species spurred the International Whaling Commission (IWC) to invite fisheries stock assessment scientists (i.e., the "Committee of Three"; Nagtzaam, 2009) to help evaluate the status of whale stocks and recommend quotas, leading to some of the first quantitative stock assessments ever conducted for whales. Similar methods were then adopted for what were perhaps the first assessments of the impact of bycatch on marine mammal populations, those of Smith (1979, 1983) for dolphin populations in the eastern tropical Pacific Ocean killed in a tuna purse seine fishery. These assessments were "back-calculations" in which a population model, an estimate of current abundance (and trends, if available), and a complete historical record of estimates of bycatch mortality were used to calculate the pre-exploitation population size. The ratio of current to pre-exploitation abundance (referred to as the 'depletion level') was used to summarize population status. This type of assessment was used to address one of the primary objectives of the United States Marine Mammal Protection Act (MMPA), which directs that marine mammals should not be permitted to diminish below their "optimum sustainable population" (OSP), defined as being between the maximum net productivity level (MNPL) and the maximum number of individuals that the environment can support (the carrying capacity of the environment, *K*). Under the MMPA, a population that falls below MNPL (often considered to be 50% of *K*) is designated as *Depleted,* and management actions designed to protect and recover *Depleted* populations may be taken.

Scientists and managers rarely have enough data to assess the depletion level of bycaught species, because there is seldom a record of bycatch going back in time to the start of all fisheries. For example, over the first 22 years where the MMPA was in force in the United States, only 12 (8%) of all marine mammal populations in US waters were assessed relative to MNPL (Taylor et al., 2000). Basing management on a finding of *Depletion* is also not proactive in preventing depletion in the first place. Using fisheries assessment terminology, a depleted population is analogous to a population that is *overfished*. What is missing from this approach is a way to evaluate whether the level of bycatch is high enough to eventually lead to depletion, which is analogous to a fish population that is experiencing *overfishing* (Methot et al., 2014).

A different and simpler approach to assessing marine mammal bycatch is to develop a bycatch reference point based on data that can be collected and analyzed at any time, especially data that can be used to estimate abundance (Taylor et al., 2000). Marine mammal scientists have developed methods for conducting population surveys and estimating abundance (Hammond et al., 2021). Similarly, fisheries observer programs have collected data on marine mammal bycatch in many types of fisheries for decades, and robust statistical techniques have been developed to estimate the annual bycatch in a fishery (Moore et al., In review, Frontiers in Marine Science)[1]. Once estimates of abundance are available, it is relatively straightforward to calculate a bycatch reference point, which can be compared to the estimated bycatch mortality to determine if the bycatch level is too high (i.e., if it is likely to slow

---

[1] Moore, J. E., Heinemann, D., Francis, T. B., Hammond, P. S., Long, K. J., Punt, A. E., et al. (2010). *In internal review. Estimating bycatch mortality for marine mammal stock assessment: concepts and best practices. To be submitted to a Special Research Topic in the Marine Megafauna section of the journal Frontiers in Marine Science.*

recovery or lead to the long-term decline of the population or stock productivity).

The 1994 amendments to the US MMPA mandated, for the first time in the United States, the use of a reference point to evaluate human-caused mortality (e.g., bycatch), termed the Potential Biological Removal (PBR) level. Outside the United States, several similar bycatch reference points have been used, such as for evaluating bycatch of harbor porpoises (*Phocoena phocoena*) in gillnet fisheries in the Baltic and North seas (ASCOBANS, 2000), bycatch of New Zealand sea lions (*Phocarctos hookeri*) in a squid trawl fishery (Gales, 1995; Harcourt, 2001), and bycatch of several species of dolphin in the tuna purse seine fishery in the eastern tropical Pacific[2]. Many of these reference points parallel those used for fisheries stock assessment; for example, the concept of MNPL, which underpins the PBR approach, is nearly identical to the concept of Maximum Sustainable Yield Level often used in assessments of fish stocks.

The PBR reference point was developed to assess mortality of marine mammal populations, but PBR and similar mortality reference points have been recommended more generally for management of exploited species (Milner-Gulland and Akçakaya, 2001). PBR has been used to evaluate bush-meat hunting in tropical forests (Parry et al., 2009; Weinbaum et al., 2013) and to assess fisheries bycatch of seabirds (e.g., Dillingham and Fletcher, 2008; Barbraud et al., 2009; Zydelis et al., 2009). Several reference points have been proposed, more broadly, to evaluate bycatch of all marine megafauna, not just marine mammals (Moore J. E. et al., 2013; Curtis et al., 2015).

The urgent need for quantitative assessments of marine mammal bycatch in fisheries is bound to substantially increase worldwide. The import provisions of the US MMPA require that imported fish and fish products be evaluated with respect to US standards for managing marine mammal bycatch; the regulations to implement this requirement were issued in 2016 (50 CFR §216.24; hereafter referred to as the "MMPA Import Provisions"). These regulations require nations that export fish and fish products to the United States, and that are identified by the National Oceanic and Atmospheric Administration (NOAA) as having fisheries that are known or likely to involve marine mammal bycatch (called "export fisheries"), have a regulatory program governing marine mammal bycatch that is comparable in effectiveness to the regulatory program governing US commercial fisheries. A fishery may also be classified as an export fishery if there is insufficient information on marine mammal bycatch rates to determine whether the fishery has no known or remote likelihood of marine mammal bycatch and thus could be exempt from the requirement. To receive an authorization to export fish or fish products to the United States, an export fishery must be governed by a regulatory program that meets certain conditions for assessing marine mammal populations by estimating bycatch, calculating bycatch limits, and reducing bycatch below such limits in export fisheries or by implementing alternative measures (e.g., eliminate the potential for bycatch). By the end of 2022 the United States intends to make comparability findings for all export fisheries to determine whether a harvesting nation's marine mammal bycatch program is comparable in effectiveness to that of the United States. If an export fishery fails to receive a comparability finding, the products from that fishery would be prohibited from entering the United States. One of the first steps NOAA took in implementing the MMPA Import Provisions was to create a List of Foreign Fisheries[3], which currently includes more than 1,800 fisheries from 131 nations listed as export fisheries, all of which will have to be evaluated for comparability to US standards[4].

The motivation for this paper is, in part, the recognition that in many parts of the world, the MMPA Import Provisions will mean that fisheries managers need to conduct marine mammal assessments for the first time (Williams et al., 2016). In the United States, the change to using a reference point for evaluating bycatch, rather than depletion level, immediately resulted in an increase in the number of populations assessed from 12 to 112, primarily because PBR is a relatively easy method to implement (Taylor et al., 2000). By describing the simplest framework for managing marine mammal management based on the use of a reference point, we hope to increase the number of populations that are assessed worldwide.

Collectively, we, the authors, have >250 years of contributing to marine mammal assessments, and we attempt to synthesize lessons learned in a way that we hope will be useful to those who are relatively new to managing marine mammal bycatch. We have focused in the Introduction on the rationale for using reference points to evaluate bycatch levels, primarily because using reference points requires the least amount of data compared to other, more complicated assessment methods. However, calculating a reference point is just one, albeit important, part of a larger and more complex process. Considerable information gathering and scientific research must occur before a reference point can be calculated. Then, once a reference point has been calculated and bycatch levels assessed, other steps must occur before bycatch can be reduced. Therefore, we describe and provide guidance on the entire framework, including (1) planning for an assessment of bycatch, (2) collecting appropriate data (e.g., abundance and bycatch estimates), (3) conducting the assessment of bycatch (by calculating a reference point), and (4) using the results of the assessment to guide marine mammal bycatch reduction (**Figure 1**). Although many of the examples discussed stem from the US approach to addressing marine mammal bycatch, our intention is to provide a general framework for assessing marine mammal bycatch more broadly.

We briefly discuss alternative, more complicated assessment methods, and we also discuss an alternative approach of mitigating bycatch without conducting an assessment, though adopting this approach can be problematic. Although **Figure 1** contains eight discrete steps, in some cases, where some information is already known about both the affected marine mammal community and fishing activity, it may be possible to start the process at later steps.

---

[2] Agreement on the International Dolphin Conservation Program, https://www.iattc.org/PDFFiles/AIDCP/_English/AIDCP.pdf

[3] https://www.fisheries.noaa.gov/foreign/international-affairs/list-foreign-fisheries

[4] 85 FR 63527, October 8, 2020, https://www.govinfo.gov/app/details/FR-2020-10-08/2020-22290

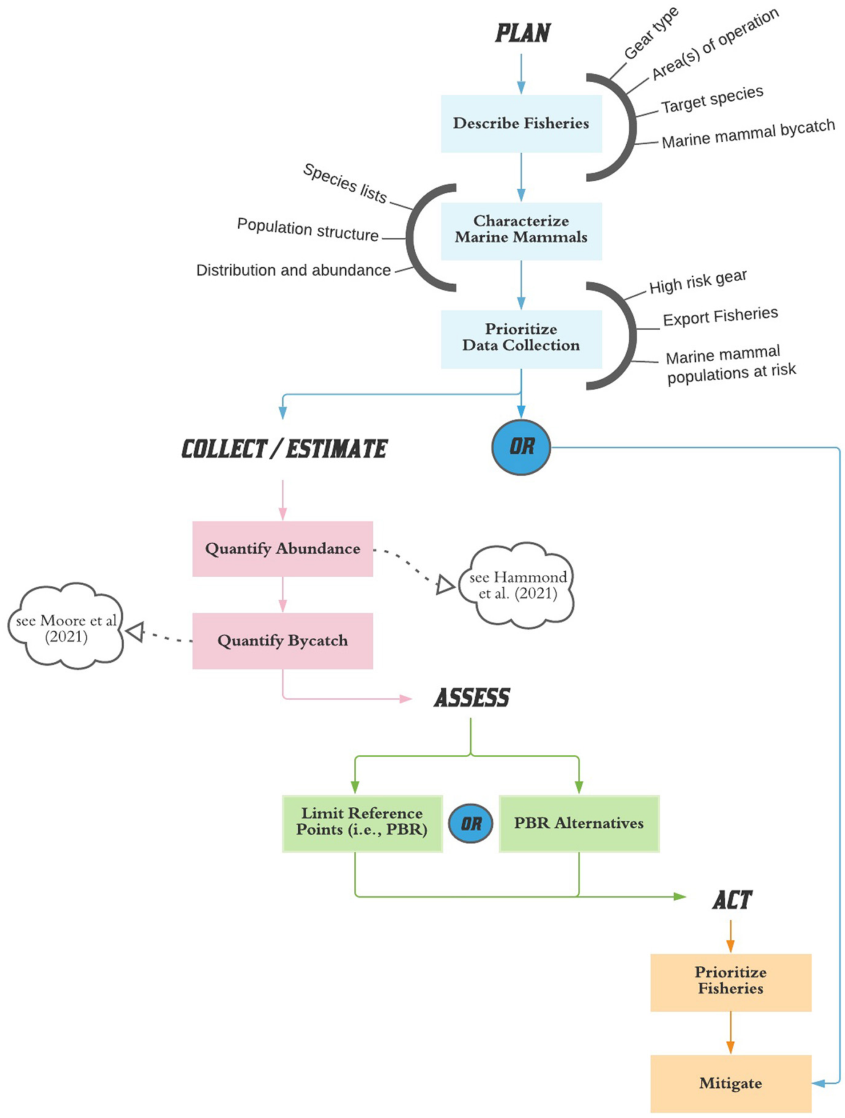


FIGURE 1 | A flow chart illustrating the process for assessing and managing bycatch of marine mammals. Names of flowchart components correspond to the names of the section headers in the paper.

PLANNING FOR AN ASSESSMENT OF BYCATCH

Identify Fisheries That May Interact With Marine Mammal Populations

The first step is to summarize what is known about the fishing gear used in a specific area. Fisheries are sometimes categorized (and managed) by target species, and therefore might deploy multiple gear types, but for evaluating bycatch of marine mammals it is important to identify fisheries by individual gear type because of the different risks posed by each gear type. Enough is known about marine mammal bycatch worldwide to reasonably predict which types of fishing gear will have the highest bycatch risk for particular marine mammals (Read, 2008; Brownell et al., 2019). Identifying fisheries that use high-risk gear and show substantial spatial and temporal overlap with the distribution of marine mammals provides a strong indicator of the likelihood of a bycatch problem. For example, global bycatch risk for odontocetes (toothed whales, dolphins, and porpoise) from small-scale gillnet fisheries has been estimated from species occurrence and bycatch susceptibility combined with estimates of fishing pressure (Temple et al., 2021). High-risk gear types, such as gillnets, purse seines, and trawls, not only have the capacity to take large numbers of marine mammals, but the potential for mortality is also very high due to the length of time the gear is fished and the inability of captured marine mammals to reach the surface to breathe. Other gear types, such as longlines, traps/pots, and pound nets, have variable impacts depending on which marine mammals occur in the fishing area (**Box 1**).

BOX 1 | Default priority level for data collection for different fishing gear types, based on likely risk for marine mammal bycatch. This is intended as a starting point, given that there can be different risks to different species depending on the fishing location, amount of fishing effort, how the gear is configured and fished, and other considerations. For example, the risk from buoy lines of trap/pot gear can be a high priority for large whales, such as the considerable risk such gear poses to North Atlantic right whales. In contrast, the same trap/pot gear may be a low priority for species such as seals and sea lions. Similarly, the risk from longline gear is particularly high for species that depredate longline gear, such as pilot whales, false killer whales, and sperm whales, but may be a low priority for species such as large whales.

Gear type	Risk priority	Considerations
Gillnet	High	All types, including drift, set, anchored, and trammel, are generally high risk for all species of marine mammals.
Trawl	High	Bottom or mid-water. Risk can vary depending on the speed of the trawl and size of the opening, with higher risk associated with faster tow speeds and wider trawl mouths.
Purse seine	High	Risk is variable, but can be high particularly if there is intentional encirclement, such as of dolphin schools in the eastern tropical Pacific or if the fishery targets fish that are also marine mammal prey, leading to inadvertent capture of marine mammals.
Trap/pot	Medium	Risk is species dependent. Buoy lines from trap/pot gear can be a high risk for large whales, such as right and humpback whales, and in the United States there is bycatch of bottlenose dolphins in some pot gear. Additionally, some pinnipeds can become entangled and drown after entering pots.
Longline	Medium	Bottom or pelagic. Many species can be captured, but higher risk is mainly associated with species that frequently depredate catch (e.g., pilot whales, sperm whales, killer whales, false killer whales).
Fyke (trap) net	Medium	Can pose substantial risk for species like harbor porpoises and seals.
Dredge	Low	Though similar to trawling, lower tow speed and narrow opening usually leads to low risk.
Hook and line	Low	Includes trolling. There are reports of bycatch with what are likely depredating sea lions, dolphins, and killer whales.
Demersal seines	Low	Includes Danish and Scottish seines.
Pound net	Low	There are reports of interactions with some dolphin species and harbor porpoises.
Cast and ring net	Low	
Jigs	Low	
Handline	Low	

After gear types have been identified, fisheries are often described by area of operation and/or target species. There is not necessarily a single best way to do this; it may be most sensible to anticipate what categorization would facilitate implementation of mitigation measures, should they be necessary. Once a list of fisheries has been developed, available information on the number of boats or individual fishery participants, the level of fishing effort, and the seasonal and spatial distribution of that fishing effort should be summarized to give an indication of the potential for bycatch of marine mammals (see for example, the US "List of Fisheries")[5].

Next, available information on marine mammal bycatch in each specific fishery should be compiled. Assuming that bycatch has not been directly studied or observed previously, indirect or anecdotal information may be available, which can sometimes provide a good indication of substantial marine mammal bycatch in a specific fishery or area (**Box 2**). Records of marine mammal strandings are sometimes available, either through systematic stranding programs, anecdotal reports to fisheries agencies, accounts published in scientific literature, or reports in traditional and social media outlets. Stranded animals or even live pinnipeds when hauled-out, can be evaluated for evidence of fishing gear interactions (e.g., net marks on the body, recovered hooks/line/net) that sometimes can identify whether an animal died due to an interaction with fishing gear, and can implicate which type of gear was involved (Page et al., 2004; Moore M. J. et al., 2013; Ashe et al., 2021). Even if stranded animals do not provide enough information to identify gear type, sometimes the spatial and temporal co-occurrence of strandings and the operation of a fishery can suggest a fishery has substantial bycatch that should be investigated further. Examination of beach-cast carcasses can reveal that fisheries interactions exist, but may be unreliable for estimating the extent of bycatch

BOX 2 | How does one know whether marine mammal bycatch occurs in a fishery?

- Talk to fishermen – not all people associated with a fishery are willing to self-report marine mammal bycatch, but some do.
- Are there regular strandings of marine mammals on the coast in certain areas where fisheries operate?
    - ○ Detailed examination of fresh carcasses can reveal marking and other information that indicates whether fishing gear, and what type, caused the death of an animal.
- Are there fisheries using gear types known to be high risk for marine mammal bycatch?
    - ○ Some types of gear, such as gillnets, nearly always catch marine mammals if they co-occur.
- Search the popular media or social media for anecdotes from the public, and look for accounts published in scientific literature or other forums.
- Rapid assessment techniques can be used if no other information is available.

---

[5] https://www.fisheries.noaa.gov/national/marine-mammal-protection/list-fisheries-summary-tables

mortality, not least because the probability of a carcass stranding can vary among species by orders of magnitude (Williams et al., 2011).

A formal fishery observer program to estimate marine mammal bycatch rates (described below under "Quantifying marine mammal bycatch") is the most reliable way to evaluate whether a fishery has substantial bycatch, but when setting initial priorities, some other rapid-assessment methods can be used to identify if bycatch occurs. Fishermen can be requested or mandated to self-report bycatch of marine mammals. However, because self-reporting rates are often low, the use of such data typically results in negatively biased estimates of bycatch rates (Walsh et al., 2002; Emery et al., 2019; see Mangi et al., 2016), but such data can be valuable to identify whether bycatch occurs. Similarly, dockside interviews with fishermen can be conducted to collect information about marine mammal bycatch. Such interviews, especially if conducted by people known and trusted by the fishing community, can often reveal much about marine mammal bycatch (e.g., Moore et al., 2010; Pardalou and Tsikliras, 2018). Methods can be combined; for example, Mustika et al. (2014) describe a pilot study to identify the extent of small cetacean bycatch in Indonesia through fishermen interviews and stranding data. Another possibility arises if a marine mammal survey is conducted; data on direct occurrence of fishing boats can then also be collected to document the distribution of fisheries and their co-occurrence with marine mammals, to identify important areas of overlap (e.g., Goldsworthy and Page, 2007; de Boer et al., 2016; Baird et al., 2021). Similarly, Braulik et al. (2018) describe an approach for a rapid assessment that integrated collection of data on cetaceans from visual, acoustic, and interview surveys with existing information from multiple sources, to provide low-resolution data on the relative abundance of cetaceans as well as on threats such as bycatch.

Hines et al. (2020) have developed a geographic information systems tool based on open-source software for analyzing bycatch in small-scale fisheries, called Bycatch Risk Assessment (ByRA). The tool combines data on spatial locations of fishing vessels from marine mammal surveys with information from interviews with fishermen or other experts to create a GIS layer of fisheries risk, which is combined with a habitat model from survey data and environmental variables to predict the distribution of marine mammal species. Bycatch risk is evaluated based on the spatial and temporal coincidence of ranked probabilities of overlap between a species' occurrence and fishing; such analyses can be used to set priorities for collecting data on bycatch rates and fishing effort, and can identify areas deserving of management efforts and further research. Verutes et al. (2020) show an example of the use of the ByRA tool in a case study examining risk to Irrawaddy dolphins (*Orcaella brevirostris*) and dugongs (*Dugong dugon*) from five small-scale fishing gear types in Malaysia and Vietnam.

Initially Characterize the Marine Mammal Community

The marine mammal community needs to be described and characterized to create a list of all the marine mammal species that occur in the region, and a description of the population structure (number and boundaries of discrete populations) within each species in the region. Information about a population learned from any surveys (formal or informal) should be summarized, especially related to the population's distribution and abundance. It is also important to summarize anything known about the population structure. Many types of information can be used to identify populations of a species, including distribution (especially a hiatus in occurrence), movements, population trends, morphology, life history, genetics, acoustic signatures, chemical signals including contaminants, and habitat preferences (Martien et al., 2019). Additional information should be summarized, such as anything known about the spatial and seasonal occurrence of each population.

Even if no formal surveys have been conducted, it should at least be possible to describe which species are known to occur in a region, and in which marine zone each species is expected to be found, including the (1) Coastal (Littoral), (2) Shelf (Neritic), (3) Continental slope, or (4) Oceanic zones. For example, sperm whales (*Physeter macrocephalus*) are rarely found in the Coastal or Shelf marine zones. Anecdotal information is often available to document occurrence of most species. In most coastal areas, people who are on the water regularly, such as fishermen, will be familiar with which marine mammal species occur in their area. Because pinnipeds haul-out on land to give birth, molt, or rest, the pinniped species that occur in an area are usually well known, though their at-sea distribution may be unknown. Similarly, the occurrence of coastal cetaceans that can be seen from shore, such as bottlenose dolphins (*Tursiops* spp.), humpback dolphins (*Sousa* spp.), franciscana (*Pontoporia blainvillei*), harbor porpoise (*Phocoena phocoena*), or Burmeister's porpoises (*Phocoena spinipinnis*) will likely be well known in a region. However, given that fisheries often occur throughout a nation's Exclusive Economic Zone (EEZ), which extends up to 200 nautical miles from shore, it will be necessary to characterize the marine mammal community within the EEZ. Unless dedicated surveys have occurred in those areas, little documentation may exist about which species occur there, particularly for relatively cryptic species such as beaked whales. There are several resources that can be used to create a complete list of species likely to be found in a certain region, which can serve as a starting point (**Supplementary Material S1**).

Although information about bycatch of marine mammals in specific fisheries will have already been summarized in the previous step (above), it is also useful to summarize information about bycatch specific to each species. For example, a summary of fisheries known to take a particular marine mammal may point to a priority species if it is killed as bycatch in many large fisheries. Additionally, some information might be available for a species that is not tied to a specific fishery; this might include stranding records that indicate bycatch of the species, but not which specific gear or fishery.

Any information about other sources of anthropogenic mortality should also be included. Finally, anything known about the conservation or management status of the population should be summarized, including IUCN Red List status, and, if relevant, any status relative to domestic legislation or assessment frameworks.

Prioritizing Data Collection

Data on abundance and bycatch are essential to assessing and mitigating bycatch impacts. If those data are not available, programs to estimate abundance and fisheries bycatch will need to be developed. Because it is usually impractical to immediately collect all necessary data, some decisions will need to be made about which marine mammal surveys to first conduct, and which fisheries to first observe. Obviously, creating a meaningful list of priorities will be more difficult the less that is known, but even with little information, it is still possible to establish priorities based on several considerations. Here, we start with how to set priorities that are most feasible in a situation where little or no abundance or bycatch data are available.

The most important initial step is to compile information about the types of fisheries that occur in an area, and identify those that are likely to have the greatest potential for bycatch of marine mammals. If little information is available about bycatch rates, or even if bycatch occurs, we recommend the risk categories based on gear type (**Box 1**). Where no evidence of bycatch is known, but monitoring has been sparse or non-existent, it is important not to assume that bycatch does not occur. In general, one needs to be cautious because no data or incomplete data does not necessarily indicate a lack of bycatch impacts. Basing priorities on overlap between fisheries known to have substantial marine mammal bycatch in other regions ('risky fisheries') and marine mammal density distribution will avoid this pitfall (e.g., Hines et al., 2020), though this is often insufficient (Williams et al., 2014). Nonetheless, in the absence of other information, the overlap between risky fisheries and the range of marine mammal species can represent a starting point for collecting data.

For nations exporting seafood products to the United States, if NOAA's List of Foreign Fisheries (LOFF) identifies a fishery as an export fishery, this is a good indicator to begin assessing that fishery for marine mammal bycatch. The LOFF uniformly classified all gillnet, driftnet, set net, fyke net, trammel net and pound net fisheries as export (rather than exempt) fisheries because the likelihood of marine mammal bycatch is more than remote. For other gear types, including trap/pot, longline and troll line, purse seine, and all trawl, the LOFF classified these as export fisheries with limited exceptions; these limited exceptions include when a harvesting nation provided information that the fishery did not overlap with marine mammals, had very low documented bycatch rates, was analogous to a US commercial fishery that had low documented bycatch rates, or had implemented mitigation measures to prevent bycatch. On the LOFF, highly selective fisheries that have a remote likelihood of marine mammal bycatch (i.e., low priority for data collection or not a priority at all) are exempt fisheries and include the following gear types: hand collection, diving, manual extraction, hand lines, hook and line, jigs, dredges, clam rakes, beach-operated hauling nets, ring nets, beach seines, small lift nets, cast nets, small bamboo weir, floating mats for roe collection, and most forms of aquaculture.

Gillnet fisheries have long been recognized to have high bycatch mortality rates in nearly all configurations, including drift gillnets and anchored/set gillnets (Perrin et al., 1994; Reeves et al., 2013). Substantial bycatch has been documented in areas where coastal gillnet fisheries overlap distributions of coastal marine mammals, such as harbor porpoises or bottlenose dolphins (Brownell et al., 2019), and proximity to the shore often leads to evidence of such bycatch from strandings usually with visible net marks on the body (de Quiros et al., 2017). However, it is well known that pelagic gillnets can also have high bycatch of marine mammals, such as >100,000 cetaceans per year in tuna gillnets in the Indian Ocean (Anderson et al., 2020), so any type of gillnet fishery is potentially high risk. Some types of purse seine fisheries can have substantial bycatch particularly if there is intentional encirclement, such as of dolphin schools in the eastern tropical Pacific (Perrin, 1969; Wade, 1995) or if the fishery targets fish that are also marine mammal prey, leading to inadvertent capture of marine mammals. Midwater or surface trawl fisheries will sometimes have high marine mammal bycatch, depending on the gear, with larger openings and higher trawl speeds increasing risk; for this reason, some pair-trawl configurations have had particularly high bycatch and bycatch mortality rates (e.g., De Boer et al., 2012). Some bottom (demersal) trawl gear can have relatively high risk to marine mammal species that forage on or near the sea floor (e.g., Franco-Trecu et al., 2019).

Other gear types can pose a medium to high risk to marine mammals depending upon their configuration and operation. Longline fisheries can have substantial bycatch of marine mammals, especially odontocetes (toothed whales) known to take bait or target fish from fishing gear (Hamer et al., 2012). Many hooked and/or entangled marine mammals are able to reach the surface to breathe, but even those that are released alive or self-release with some gear remaining attached (e.g., a hook and some amount of line) may have suffered serious injuries that are likely to lead to death. Pot fisheries can trap and drown sea lions in the pot itself (Campbell et al., 2008). Trap/pot fisheries can entangle baleen whales (Johnson et al., 2005) and small cetaceans in buoy lines that fishermen use to locate and retrieve traps/pots from the bottom, or in ground lines used to connect traps/pots; when these pot fisheries occur at high densities, they can pose substantial risk to large whale populations (e.g., Kraus et al., 2005). After becoming entangled, baleen whales generally swim off with gear attached that can impede feeding, reproduction, and/or swimming, cause substantial injuries and suffering, and ultimately lead to death (van der Hoop et al., 2016). When marine mammal populations are small, such as the North Atlantic right whale (*Eubalaena glacialis*), mortality of a few individuals a year can have substantial population-level impacts.

Several gear types are thought to have low to medium risk of bycatch, depending on the specific gear and mode of operation. Dredge gear is somewhat similar to demersal trawl gear, but it has smaller openings, lacks large trawl doors, and is usually towed at a lower speed, so the risk to marine mammals is generally thought to be lower. Pound nets can trap small coastal cetacean and pinniped species; there have been some cases where substantial catches have occurred locally. Similarly, beach seines generally pose medium or low risk to marine mammals, but they can substantially impact small, localized populations of small cetaceans (Pierce et al., 2020). Troll fisheries tend to pose lower risk to marine mammals, though there are cases where trolling with hook-and-line gear could be considered impactful especially if the vessel uses dolphins to locate fish and maneuvers through a group of marine mammals (Baird and Webster, 2020). Hook-and-line fisheries are considered low risk, although there are some well documented cases where depredation in recreational fisheries has led to interactions and serious injuries, such as for bottlenose dolphins (*Tursiops truncatus*) in the Gulf of Mexico (Wells et al., 2008) and killer whales (*Orcinus orca*) in British Columbia. Other types of fishing gear or methods (e.g., jigs, handlines), many manually deployed, may also be considered low-risk because of the short duration of deployment or because marine mammals generally do not occur in areas where these gears are deployed.

The spatial and temporal distribution of effort in high- and medium-priority gear-type fisheries should be compared to the known or suspected distribution of marine mammal populations. Any obvious hotspots of co-occurrence of high densities of marine mammals and high-priority fisheries can contribute to a preliminary list of the highest priority areas to investigate. Similarly, co-occurrence with medium-priority fisheries will provide a preliminary list of the second highest priority areas. After this initial evaluation, any additional information can be used to finely tune priorities for data collection. Evidence of concurrent strandings or anecdotal reports of bycatch in identified areas could elevate the priority of a specific area or fishery. As mentioned above, it could be helpful to formalize this step in a GIS-based decision framework, such as the toolbox described by Hines et al. (2020). If fisheries bycatch mortality of a certain species is known to occur at an appreciable but unquantified rate, a decline has been noted in the relative abundance of that species, and there are no other obvious explanations for what has caused that decline, this would indicate that data collection on bycatch rates of that species is a high priority.

The next step is to evaluate other considerations. Marine mammal populations that are small or declining, and/or have already been identified as a conservation concern, would be a higher priority to assess than those with a larger population size or increasing trends. The goal is to develop a list of priorities for fisheries to monitor and marine mammal populations to assess given potential bycatch impacts.

COLLECTING APPROPRIATE DATA

Quantify Marine Mammal Bycatch

It is critical to estimate the magnitude of annual bycatch in fisheries to assess marine mammal bycatch using a mortality reference point. Fully describing how to observe fisheries, collect effort data, and estimate bycatch rates and total bycatch are beyond the scope of this paper. See text footnote 1 provide a comprehensive guide to these processes, including identifying minimum requirements for obtaining credible estimates of bycatch and best practices. See text footnote 1 also focus on empirical studies that have generated quantitative results (with uncertainties and limitations specified). Here, we briefly outline the primary steps involved in quantifying bycatch to describe the scope of the process.

The standard way to quantify marine mammal bycatch in a fishery is through a two-part process, including (1) observing fishing operations and bycatch for a portion of a fishery, and (2) collecting effort data for the entire fishery. With these two types of data, a bycatch rate for the observed portion of the fishery can be estimated, and it can be applied to some measure of total fishing effort to estimate total marine mammal bycatch. Among the primary approaches to data collection are on-board observer programs (commercial or research vessels), on-board camera systems, observer programs from secondary platforms (if observers cannot be placed on fishing boats), logbook records (self-reporting), and structured interviews with fishermen, including dockside surveys.

The proportion of fishing effort that needs to be observed will vary among fisheries. In general, the larger a fishery is (more vessels and/or fishing trips), the smaller the percentage of the total fishing effort that must be observed to adequately characterize bycatch rates. It is important that the observations randomly sample the entire fishery to produce unbiased estimates of bycatch mortality. This often requires understanding the fishery in great detail to, for example, ensure that fishing in all spatial areas or seasons are sampled (e.g., such as ensuring that fishing trips originating from all fishing ports are sampled). It is common to use stratified sampling designs, for example, to observe various-sized vessels as separate strata with different observation rates, or to sample at higher rates in areas with the greatest amount of fishing effort, or in areas that are suspected or known to have the highest rate of marine mammal bycatch.

Estimating or quantifying bycatch in fisheries that are not directly observable is particularly challenging; the most common example is entanglement of large whales in buoy lines attached to pot gear. Evidence that large whales are entangled in lines can be seen from specific types of scars seen on the whales (Knowlton et al., 2012), but this cannot determine which fisheries are responsible for the entanglement. Typically, pot gear can be left in the water unattended for considerable periods of time, so quite often no fishing vessel is in the vicinity of the gear when a large whale entanglement occurs. Entangled whales are often later encountered, either alive, floating dead, or stranded, so one approach is to mark the gear in a way that allows the specific fishery to be identified from lines and other gear still attached to the whale (e.g., Johnson et al., 2005).

It may also be necessary to address situations where marine mammals are observed injured and released alive, but their fate post-release is unknown and they may ultimately die. There are guidelines in the United States for assessing injuries and determining whether a given injury is likely to lead to death (National Marine Fisheries Service, 2012). For some injuries it is possible to estimate the proportion of marine mammals injured that ultimately died from the injury based on extensive population monitoring and known outcomes for individuals that were documented as injured in fishing gear (National Marine Fisheries Service, 2012).

There are several ways to analyze bycatch data, and statistical methods are rapidly improving (see text footnote 1). Simple ratio estimators are sufficient in a properly designed study; so-called design-based methods assume that bycatch in the observed portion of the fishery can be extrapolated to the whole fishery because the fishery is sampled in a representative way. Other approaches, generally referred to as model-based estimators, will work better when sampling cannot meet this standard. In some situations, model-based estimators can improve precision and reduce bias in bycatch estimates. There are strengths and weaknesses of each approach; see text footnote 1 for further discussion.

Some elements of characterizing a fishery (see **Supplementary Material S2**) are essential for estimating bycatch (e.g., fishing effort) whereas others are more relevant for developing mitigation measures (e.g., nature of interactions, such as whether hooked or entangled, the amount of gear remaining on the animal, whether depredation occurred), so it is worth collecting those types of data, too, even if they are not directly used for estimating bycatch. Observer programs may need to be modified to provide sufficient data to evaluate whether mitigation measures are effective in reducing the bycatch to below the reference point or to meet conservation goals.

Quantify Abundance of Marine Mammal Populations

A fundamental requirement to assess the status of a population is the availability of a nearly unbiased estimate of absolute abundance. Hammond et al. (2021) provide a comprehensive guide to estimating the abundance of marine mammal populations, and they identify minimum requirements for obtaining credible estimates of abundance and suggest best practices. That review also describes many examples of studies that have resulted in credible abundance estimates (with uncertainties and limitations specified). Here, we briefly outline some common abundance quantification techniques and applications; see Hammond et al. (2021) for guidance on how to select a method for a particular situation.

For cetaceans, several techniques and field methods are used depending on the marine mammal species and regions (as an illustration, see **Supplementary Material S4** for a summary of the methods used for all marine mammal populations in the United States). Most cetacean abundance estimates are made using either line transect (distance sampling) methods (74% of US stocks, **Supplementary Table 1**) (Buckland et al., 2001) or mark-recapture methods (25% of US stocks, **Supplementary Table 1**) based on the identification of individuals (Hammond et al., 1990). Line-transect surveys involve conducting a survey with observers along transects that sample the area for which an abundance estimate is desired, while collecting data on the perpendicular distance (the distance away from the trackline) of each marine mammal group that is seen. The platform to use for cetacean line-transect surveys varies depending upon the region to be surveyed (e.g., ships in oceanic regions; ships and airplanes in continental slope and shelf regions; ships, small boats, and airplanes in coastal and nearshore regions). Table 1 in Hammond et al. (2021) provides a list of example studies of line-transect methods used to estimate the abundance of cetacean populations from small boats, ships, and airplanes.

There is an extensive literature and history of using mark-recapture methods to estimate abundance of terrestrial wildlife, where animals are captured, marked in some way (such as with a tag), released, and then recaptured at a later sampling time (e.g., Otis et al., 1978; Schwarz and Seber, 1999; Chao, 2001). Many of these methods have been adapted for use with marine mammals and are usually based on the sighting and resighting of individuals over time. Individuals are normally identified from natural markings observed in photographs, but sometimes through genotyping of skin biopsies (e.g., Hamner et al., 2014) or from marks placed on animals. Identification of cetaceans can be based on scars, notches in the dorsal fin/ridge, color patterns, the shape of the dorsal fin or flukes or other natural marks. Table 2 in Hammond et al. (2021) provides a list of example studies where mark-recapture analyses have been applied to photo-identification data to estimate the abundance of marine mammal populations.

Other methods for estimating cetacean abundance exist. For example, boat surveys are often used to count freshwater dolphins and porpoises, without using line transect methods, but instead use correction factors applied to the counts. Alternatively, in a few cases the abundance of baleen whale populations that migrate close to headlands (e.g., gray whales, *Eschrichtius robustu*s, and some populations of humpback whales, *Megaptera novaeangliae*) can be estimated using land-based counts.

The abundance of a pinniped population is often estimated based on a count of individuals when they are hauled out on land or ice. Depending upon the species, the counts can be conducted from land, boats, drones, or airplanes. The surveys may be timed to coincide with times when the most individuals are hauled out, such as during molting, and all age classes are counted (54% of US pinniped stocks, **Supplementary Table 1**). These counts are often corrected for the proportion of animals that are hauled out at the time of the survey, which can be estimated using telemetry data from tags placed on the animals. Abundance can also be estimated from counts of the number of pups on a rookery (21% of US pinniped stocks, **Supplementary Table 1**); this method can be useful when not all age and sex classes haul out at the same time. A population model, using estimates of survival and birth rates, must be used to extrapolate a pup count to an estimate of total abundance. Data needed to estimate those parameters can be difficult to collect, and parameters may change over time, so this method is not used as commonly as haul-out counts. The abundance of several ice seal species has been estimated using line- or strip-transect surveys conducted from airplanes (10% of US pinniped stocks, **Supplementary Table 1**). This can also include the use of tag telemetry data to estimate the proportion of the population that is hauled out on the ice at the time of the survey. Mark-recapture methods have also been used to estimate the abundance of some pinnipeds, using marks applied to the animals or natural coloration patterns [see Table 2 in Hammond et al. (2021) for examples].

The abundance of sirenians or marine mustelids (e.g., sea otters, *Enhydra lutris*) is usually estimated in a similar manner to cetaceans, using line-transect or strip-transect surveys from airplanes or boats.

CONDUCTING AN ASSESSMENT OF BYCATCH

The United States Example – The Stock Assessment Process and the PBR Reference Point

Moore M. J. et al. (2013) and Curtis et al. (2015) offered a set of guidelines for conducting reference-point based management, outlining a general assessment framework of which the US PBR approach can be considered a special case. Their reviews include extensive discussion of the basis for different conservation objectives and risk tolerances that can be used to tune the reference point estimators as a function of these and species conservation status (e.g., IUCN Red List status). For convenience, and because it would be relevant to the MMPA Import Provisions, we briefly summarize how the PBR bycatch reference point is derived for US marine mammal populations, and how assessments are conducted.

The US MMPA requires a "Stock Assessment Report" for each marine mammal population[6] in US waters (see **Supplementary Material S3** for a list of contents for such reports), which makes transparent how much (or little) is known about each population and whether bycatch is a concern. To assist in this, the National Marine Fisheries Service has published "Guidelines for Preparing Stock Assessment Reports pursuant to the 1994 amendments to the MMPA,"[7] which we refer to as 'the Guidelines.' The Guidelines serve as a practical guide for how assessments are conducted in the United States, and here we highlight some of the important issues.

Understanding population structure is an essential first step in conducting an assessment. The Guidelines expand on the MMPA definition of a stock: "*For the purposes of management under the MMPA, a stock is recognized as being a management unit that identifies a demographically independent biological population*." Data on population structure are often lacking, but defaulting to a single population for an entire ocean basin is likely inappropriate given what is known about population structure for most species. The Guidelines note that for some species genetic and other biological information has confirmed the likely existence of stocks of relatively small spatial scale, and therefore a species' range within an ocean should be divided into stocks that represent defensible management units. A guide for how to delineate demographically independent populations of marine mammals can be found in Martien et al. (2019).

Once stocks are defined, it is possible to summarize information on abundance, fisheries bycatch mortality, and other information. Calculating the bycatch reference point PBR depends upon three values:

$$\mathrm{PBR} = \mathrm{N_{min}} \cdot 1/2\mathrm{R_{max}} \cdot \mathrm{F_r},$$

where N_{min} is a minimum estimate of abundance that provides assurance the true population size is larger, R_{max} is the maximum net reproductive rate, and F_r is the recovery factor (in the United States the F_r must range between 0.1 and 1.0, but in theory it can approach 0). In the Guidelines, N_{min} is defined to be the 20th percentile of an abundance estimate. The 20th percentile was specified using simulations that showed that populations that experienced mortality at the level of a PBR calculated with that value of N_{min} would stay at or recover to MNPL with 95% probability (Wade, 1998; Taylor et al., 2000). This type of simulation performance testing is known as management strategy evaluation, which has become common in fisheries management and conservation (Bunnefeld et al., 2011).

The Guidelines provide default values for R_{max} (0.04 for cetaceans and manatees; 0.12 for pinnipeds and sea otters) and require reliable stock-specific information to use an estimated R_{max} in lieu of the defaults. In practice, relatively few population assessments use stock-specific estimates of R_{max}. Notable exceptions include some endangered small populations that are not recovering (e.g., southern resident killer whales), and populations of humpback whales that have been documented to increase at rates greater than 0.04.

The intent of the US MMPA in including F_r when calculating PBR was to ensure the recovery of a population to its Optimum Sustainable Population level (by providing a precautionary buffer against potential biases or other non-ideal circumstances) and to expedite recovery (minimize recovery time) for endangered, threatened, and/or depleted populations. Therefore, the Guidelines specify that the default value for F_r for populations of unknown status (i.e., not known to be above MNPL) should be 0.5; this value was chosen because Wade (1998) indicated this value would make the PBR, and the achievement of the objectives of the MMPA, robust to common biases and issues with the estimates of PBR and bycatch levels. The Guidelines note that "*Recovery factors of 1.0 for stocks of unknown status should be reserved for cases where there is assurance that N_{min}, R_{max}, and the estimates of mortality and serious injury are unbiased and where the stock structure is unequivocal*." Species listed as endangered under the US Endangered Species Act (ESA) are given a value for F_r of 0.1 to promote recovery and minimize the probability of extinction. Although there is not a one-to-one correspondence between listings under the US ESA and the IUCN Red List, it is our view that it would be a reasonable starting point to set $F_r = 0.1$ for any species or population listed as Critically Endangered or Endangered on the IUCN Red List.

---

[6] The US MMPA defines the terms "population stock" and "stock" to be synonymous; that is, "The term "population stock" or "stock" means a group of marine mammals of the same species or lower taxon in a common spatial arrangement, that interbreed when mature." We use the term population throughout unless referring to a specific US MMPA directive that uses the word stock.

[7] https://www.fisheries.noaa.gov/national/marine-mammal-protection/guidelines-assessing-marine-mammal-stocks

The US MMPA requires that all human-caused mortality and serious injury[8] of the population be estimated, including bycatch mortality, and that all fisheries interacting with marine mammals be described, including number of vessels/participants in a fishery and seasonal/area differences in fishery mortality. The Guidelines recommend, where appropriate to improve precision, using average annual mortality over the last 5 years that are available. A stock is designated as *Strategic* if this level of annual mortality exceeds PBR. In the US, PBR is not used as a cap, such that a fishery would close if incidental mortality and serious injury exceeded PBR. Instead, fisheries with bycatch of a *Strategic* stock may be subject to specific requirements (see below), including those for reducing bycatch to below that stock's PBR.

Alternative Assessment Methods

It is worth considering other assessment methods, if they are available and appropriate. In data-poor situations, for example, methods can be developed that rely only on trends in abundance (e.g., Punt et al., 2021b). Those methods ultimately provide lower and more variable reference points for bycatch mortality and are more sensitive to uncertainties than the PBR method. Methods based on trends also have the disadvantage that if that mortality is reduced, an increasing trend does not guarantee the population has recovered significantly. The reverse situation is different – if a decline in population size of 50% or greater has been observed, it is safe to assume that the population is below MNPL, and therefore depleted. If fisheries bycatch mortality is known to occur at an appreciable but unquantified rate and there are no other explanations for what has caused a decline, this would indicate that data collection on bycatch rates is a high priority (see above).

Alternative methods are also available in more data-rich situations. For example, Brandon et al. (2017) show that incorporating multiple abundance estimates for data-rich populations can lead to increased stability of calculated values for PBR through time, which could reduce regulatory uncertainty that may be associated with some human activities managed using the PBR reference point. With more data or resources, assessments that are more sophisticated (and complicated) than simple reference points such as PBR are possible (e.g., Goldsworthy and Page, 2007; Punt et al., 2020, 2021a).

USING THE RESULTS OF THE ASSESSMENT TO GUIDE MARINE MAMMAL BYCATCH REDUCTION

Identifying High Priorities for Bycatch Reduction

Once the assessments have been completed, it is important to identify which fisheries are priorities for bycatch reduction, as there are usually insufficient resources to attempt to mitigate bycatch in all fisheries immediately. For example, the US MMPA specifies the highest priorities for bycatch reduction are fisheries with bycatch mortality exceeding a stock's PBR, marine mammal stocks with small population size, and those stocks that are declining most rapidly.

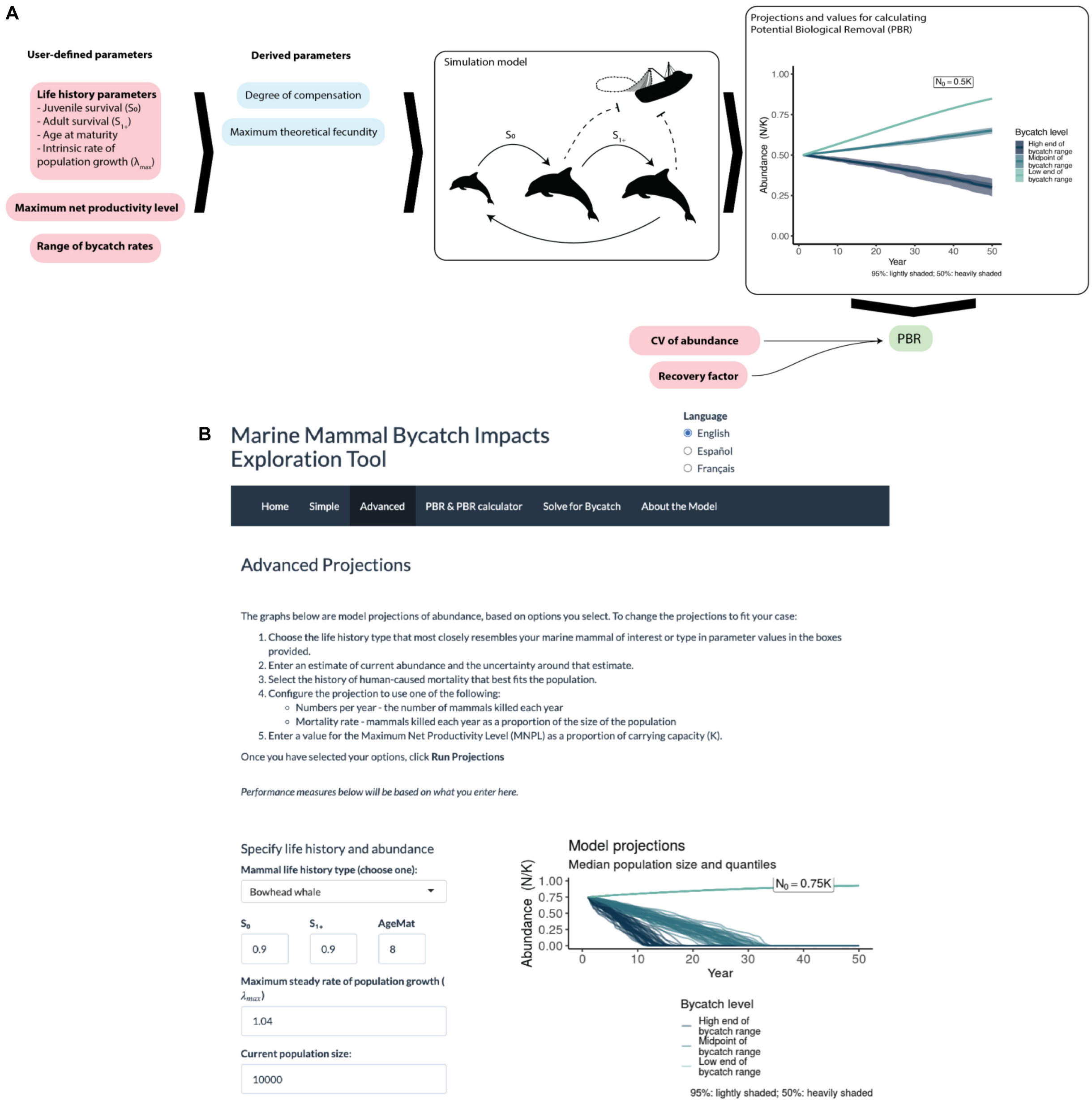
Fisheries can also be ranked by the level of bycatch mortality as a percentage of a given population's PBR. If bycatch levels exceed PBR, the fishery would be a high priority for monitoring and mitigation whereas if bycatch levels were more than 50% of PBR (but less than PBR), that fishery would be a medium priority for monitoring and mitigation. Obviously, this type of ranking can be extended if appropriate, to recognize the higher priority of situations where bycatch mortality is even greater (e.g., 2 or 3 times PBR). Another type of high priority fishery is one that has relatively high levels of bycatch of an endangered species or population.

Complications arise when assessments indicate bycatch exceeds reference points for several species. Clearly, if a single fishery has bycatch that exceeds the reference point of multiple populations, that fishery should be a high priority for management action or bycatch reduction. For example, in the United States, a plan was developed to reduce bycatch of Baird's beaked whales (*Berardius bairdii*), Cuvier's beaked whales (*Ziphius cavirostris*), *Mesoplodon* species of beaked whales, short-finned pilot whales (*Globicephala macrorhynchus*), pygmy sperm whales (*Kogia breviceps*), sperm whales, and humpback whales in the California/Oregon swordfish drift gillnet fishery. The large number of species taken at relatively high levels in a high-risk gear type made it clear that this fishery was a priority for bycatch reduction.

In the absence of high-quality survey data on abundance and/or bycatch, population projections can be used to test bycatch scenarios and identify priorities for bycatch reduction. Simulation-based tools provide a way to examine potential outcomes and inform management decisions related to bycatch limits or bycatch reduction measures, such as gear modifications meant to reduce risk to an acceptable level. Scenario analysis, in which populations are projected forward under different management decisions, allows stakeholders to see the relative impacts of these management decisions even when data are sparse for populations of interest. For example, Siple (2021) developed the Marine Mammal Bycatch Impacts Exploration Tool (MMBIET) and a corresponding R package for projecting marine mammal populations subjected to different bycatch levels (**Figures 2**, **3**). Projections can be used to determine performance in terms of population recovery and expected long-term depletion levels. In terms of reference points, the tool calculates N_{min} based on an abundance estimate and its coefficient of variation, and then calculates PBR based on a value for R_{max} (with default values based on the assumptions in the MMPA) and F_r chosen by the user (**Figure 2A**). Projection outputs and performance can then be plotted and explored (**Figure 2B**). While this tool is useful for exploring risk, the values it provides will only be as good as its inputs (e.g., if the current abundance of the population is incorrectly specified, MMBIET may provide an incorrect estimate of PBR or incorrect times to recovery). Thus, interactive tools such as MMBIET allow users to explore scenarios and identify robust management strategies, provided they are used correctly.

---

[8] "Serious injury" is defined to be an injury that is likely to lead to death.



Marine Mammal Bycatch Impacts Exploration Tool

Language
● English
○ Español
○ Français

Advanced Projections

The graphs below are model projections of abundance, based on options you select. To change the projections to fit your case:

1. Choose the life history type that most closely resembles your marine mammal of interest or type in parameter values in the boxes provided.
2. Enter an estimate of current abundance and the uncertainty around that estimate.
3. Select the history of human-caused mortality that best fits the population.
4. Configure the projection to use one of the following:
   - Numbers per year - the number of mammals killed each year
   - Mortality rate - mammals killed each year as a proportion of the size of the population
5. Enter a value for the Maximum Net Productivity Level (MNPL) as a proportion of carrying capacity (K).

Once you have selected your options, click **Run Projections**

Performance measures below will be based on what you enter here.

Specify life history and abundance

Mammal life history type (choose one): Bowhead whale

S_0 0.9

S_{1+} 0.9

AgeMat 8

Maximum steady rate of population growth (λ_{max}) 1.04

Current population size: 10000

Model projections

Median population size and quantiles

FIGURE 2 | (A) Flow of inputs and information in the Marine Mammal Bycatch Impacts Exploration Tool (MMBIET; Siple, 2021). Pink boxes indicate user-specified inputs; blue boxes indicate derived life history parameters. Users provide life history information by selecting a default life history type or by specifying parameter values manually. They also choose the Maximum Net Productivity Level (MNPL) and a range of bycatch rates to explore. The tool uses these values to conduct projections of marine mammal abundance into the future. The simulated abundance and CV of abundance selected by the user are used to calculate Potential Biological Removal (PBR). **(B)** Example screenshot from MMBIET. Open source code can be found at www.github.com/mcsiple/mmrefpoints.

Similarly, demonstrated co-occurrence of high-risk fisheries and marine mammal populations, either through qualitative evaluation or structured methods such as GIS mapping tools (e.g., Hines et al., 2020; Verutes et al., 2020; Welch et al., 2020), can help identify priority spatial areas for bycatch reduction. Bycatch mitigation, which often takes many years to accomplish, could begin in these areas while research continues. Eventually, it will be necessary to quantify bycatch rates to evaluate how much bycatch needs to be reduced to ensure it is below a reference point such as PBR. Valuable information can be gained in the meantime by initiating work on fisheries that are predicted to have unsustainable levels of bycatch.

Mitigating Marine Mammal Bycatch

Once fisheries have been prioritized, the next step is to set specific goals for mitigating bycatch. A reasonable immediate goal would be to reduce the bycatch level to below the reference point. However, reducing the bycatch to just barely below the reference

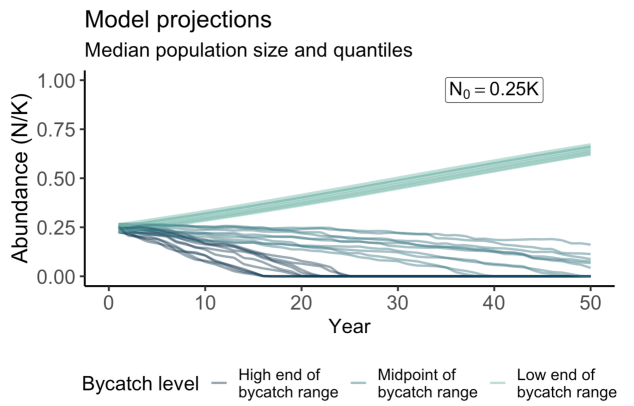


FIGURE 3 | An example of model projections for a bottlenose dolphin population, starting at 25% of carrying capacity, with a bycatch rate between 10 individuals per year (light green lines) to 1,300 individuals per year (dark blue lines). The lowest bycatch rate, which is below PBR for this theoretical population (PBR = 183 individuals), leads to a recovery to MNPL, which is set at half of the carrying capacity (*N/K* = 0.5). For these projections, the starting population size is 20,000 individuals, the CV of abundance is 0.1, and the CV of bycatch is 0.5.

point is not ideal, due to uncertainty in estimating abundance and mortality. Thus, it would be better to attempt to reduce bycatch to a level well below the reference point. Under the US MMPA, the immediate goal is reducing bycatch to below a stock's PBR, but the long-term goal is to reduce it even further to insignificant levels (i.e., 10% of PBR).

To implement effective bycatch reduction measures for fisheries, it is useful or even necessary to involve many stakeholders in the process, including fishing industry representatives, government fisheries managers and scientists, non-governmental organizations, and academic scientists. For this reason, the 1994 US MMPA amendments established a framework for reducing fisheries bycatch mortality that exceeded a stock's PBR level; this framework relies on stakeholder-based "Take Reduction Teams" to recommend consensus-based Take Reduction Plans that include mitigation measures for reducing bycatch mortality of particular marine mammal stocks in certain fisheries. Including fishermen, who bring expert knowledge and creativity, on Take Reduction Teams is critical to the success of take reduction planning.

There are various ways to mitigate marine mammal bycatch, and several recent reviews of methods are available (FAO, 2018, 2021; Leaper and Calderan, 2018; Hamilton and Baker, 2019). One with generally good success is time and/or area closures, meaning, for example, to prohibit fishing in "hotspots" or areas of substantial overlap in spatial distribution between fishing effort and marine mammal abundance at certain times of the year. If such hotspots in co-occurrence can be identified, closing the areas to fishing would, in some situations, reduce bycatch considerably. FAO (2018) provides nine examples of time/area closures from six countries. If needed, time/area closures can sometimes be implemented fairly rapidly with a high chance of success at reducing bycatch. For example, the International Council for the Exploration of the Sea (ICES) recommended emergency temporal closures of all fisheries of concern (trawls, gillnets, trammel nets, and purse seines) and the application of pingers on pair trawlers to mitigate bycatch outside the closure periods to immediately reduce bycatch of common dolphins in the Bay of Biscay (ICES, 2020). One implementation challenge is the potential for edge effects, where fishing effort is concentrated along the border of the closed area; in such cases, the bycatch reduction may not be as great as would be expected if fishing effort were dispersed across the remaining open areas.

There may be undesirable economic impacts on a fishery if fishing cannot occur in other areas or in different time periods, so stakeholders often favor modifying fishing gear and practices to avoid the potentially large economic impacts of time/area closures. This can include changing hook types or gillnet mesh size, or switching to buoyless (often referred to as ropeless) trap/pot fishing (e.g., Myers et al., 2019). Gear modification is an available and common mitigation approach in the absence of assessment (see below). For example, much research has focused on modifying longline gear to reduce the likelihood that animals get hooked (e.g., Bigelow et al., 2012; McLellan et al., 2015).

Acoustic alarms, such as pingers, have been used to alert certain species of marine mammals to nets in the water or to scare them away from nets (e.g., Carretta et al., 2008; Palka et al., 2008; FAO, 2021). However, their effectiveness in mitigating bycatch varies between locations and species (Berrow et al., 2008; Carretta and Barlow, 2011; Dawson et al., 2013). A review of multiple studies concluded that pingers were effective in reducing bycatch of harbor porpoises, beaked whales, common dolphins and franciscanas, but were not effective for bottlenose dolphins, and the authors concluded it was not possible to predict efficacy for other species; they suggested pingers might be most effective for species that are neophobic or easily startled such as the harbor porpoise (Dawson et al., 2013). Additionally, if there is a high density of fishing effort in a particular geographic region, one would need to consider the benefits of bycatch reduction versus the cost to the marine mammals of being displaced by pingers away from a significant portion of their habitat, which may be important for foraging or other essential activities. For example, harbor porpoises are highly susceptible to bycatch in gillnet fisheries, with most animals suffocating in the nets, and pingers have been shown through multiple experiments to effectively reduce bycatch; in such cases it may be more beneficial to implement mitigation measures despite any sub-lethal impacts such as displacement.

Another type of mitigation that can be applied in the absence of an assessment is gear switching, where a fishery switches from a relatively high-threat to a relatively low-threat gear type. Bycatch of vaquitas in gillnets in the upper Gulf of California has led to a severe reduction in the population to the point where just a few animals remain. The species is critically endangered, and it faces imminent extinction (D'Agrosa et al., 2000; Taylor et al., 2017). A small shrimp trawl that can be towed from artisanal-style fishing boats was recommended to replace the gillnets. Similarly, Berninsone et al. (2020) evaluated switching from gillnets to longlines to mitigate bycatch of franciscanas in Argentina.

In some cases, certain gear/fishery types have been eliminated altogether. For example, the US Atlantic Offshore Cetacean Take Reduction Team was convened to reduce the incidental mortality and serious injury of several species of marine mammal in the Atlantic pelagic driftnet, pelagic longline, and pair trawl fisheries. The pair trawl fishery was an experimental fishery for tuna, and because of the large separation between the vessels and the high tow speeds (2.5–5 knots), the fishery had a high dolphin bycatch rate. For a variety of reasons, the pair trawl fishery was not authorized as a permanent fishery. The pelagic driftnet fishery (targeting swordfish) was eventually closed to reduce marine mammal bycatch (McDonald et al., 2016). Effectively, two gear types with high bycatch rates were eliminated, leaving mitigation and bycatch reduction to focus on the longline fishery[9].

Mitigation methods, such as the use of electronic acoustic pingers, can be prohibitively expensive to implement. Awareness of this has led to recent research to develop lower-cost solutions, such as glass and plastic recycled bottle alarms and acoustic reflectors to use on gillnets instead of electronic pingers (FAO, 2018). Additional details on mitigation methods can be found in FAO (2018, 2021), which provide a thorough discussion of techniques used to address marine mammal bycatch, including a comprehensive review of mitigation techniques (Appendix 3 of FAO, 2018), and a table documenting 69 studies where an attempt was made to reduce marine mammal bycatch (Appendix 4 of FAO, 2018). Werner et al. (2015) provided specific advice for mitigating marine mammal bycatch and depredation in longline fisheries.

Consideration needs to be given to the timing and sample size to detect any change in the bycatch rate, assess eventual effectiveness, and determine whether the mitigation is successful in meeting conservation goals. It may be difficult to measure a statistically significant reduction in the bycatch rate immediately if bycatch rates are high relative to the mortality reference point, but observations of bycatch are still relatively rare; managers should consider this when evaluating the effectiveness of mitigation measures. Additionally, effectiveness should be regularly assessed at meaningful intervals to ensure that any bycatch reduction continues through time.

There are implementation considerations for all mitigation measures that managers should be aware of and plan for before implementation begins. Such considerations span many sectors – regulatory, enforcement, socioeconomic, and safety to name a few – and include subjects such as the mechanisms to enact new measures (e.g., regulations, proclamations, voluntary compliance), safety of fishermen when using modified gear or fishing practices, enforcement strategies (e.g., via marine patrol, vessel monitoring systems), supply chain impacts (e.g., time needed to manufacture a new gear modification), the time and money it will take for fishermen to incorporate gear modifications into existing gear, etc.

Mitigating Without Assessment

It may be appropriate to pursue bycatch mitigation directly without conducting an assessment. The main issue here would be determining how much mitigation to do; in other words, how do you know how much bycatch needs to be reduced if you have not calculated a bycatch reference point?

Some marine mammal species only occur in an area seasonally because they migrate to another location for a portion of the year, such as humpback whales that arrive at some nearshore areas in low latitudes for part of the winter. If a nearshore fishery overlapped with the distribution of a marine mammal population and was known to cause bycatch, that area could be closed for the entire time period in which the marine mammals are present. In this case, the bycatch would be entirely mitigated (reduced to zero), and it would not be necessary to calculate a mortality reference point as a reduction target. This is of course a relatively extreme mitigation strategy and considerable thought would need to be given as to whether the fishery could adapt.

Another example would be a change in fishing gear. For example, species that depredate longline gear, such as killer whales, pilot whales, and sperm whales, can be bycaught (by hooking or entanglement, or both). Switching to pot gear would eliminate bycatch in the longline gear but it would be necessary, of course, to evaluate whether pot fishing poses entanglement risk to large whales as well as whether it provides revenue similar to that provided by the longline fishery.

Another example would be mitigating bycatch of cetaceans in gillnet gear. As mentioned earlier, in many locations "pingers" have been used to reduce bycatch rates of cetaceans, but they have not been effective for all species. Therefore, it may not be possible to conclude that mandating the use of pingers would be sufficient mitigation without first conducting an assessment of effectiveness. It is possible that showing that circumstances are similar to those of another location where pingers have been evaluated as effective would give reasonable confidence that pingers would reduce mortality, such as successful examples with harbor porpoise bycatch reduction. However, it would still be difficult to determine whether the reduction was sufficient without an actual assessment to provide a quantitative target for bycatch reduction.

One successful example of mitigating without assessment occurred in the southern and western Australian rock lobster fisheries, which killed Australian sea lions (*Neophoca cinerea*) that became trapped and drowned when entering pots; rather than attempt to observe the fishery and estimate bycatch, a sea lion excluder device was mandated in the fishery and that essentially eliminated bycatch (Campbell et al., 2008).

DISCUSSION

We have outlined the full process of assessing bycatch of marine mammals in fisheries. We have tried to concisely provide enough

---

[9] https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-take-reduction-plans-and-teams

detail to make clear the issues and problems. Where detail is important, we have tried to provide clear pointers to the published literature and other sources of information that are particularly useful.

Although the need for more and better assessments of marine mammal bycatch worldwide has long been recognized, this paper was motivated by the recent US MMPA Import Provisions. The scale of marine mammal bycatch assessments that countries will need to perform to continue exporting seafood to the United States has increased, possibly by an order of magnitude. Countries have already been taking at least a few of the steps described above, particularly with the need to describe fisheries that interact or may interact with marine mammal populations, information that is needed for the List of Foreign Fisheries.

Challenges and Future Work

If we consider fisheries worldwide, the sheer magnitude of managing bycatch is daunting. There are thousands of fisheries, and relatively few of these have been monitored for marine mammal bycatch. Initiating observer programs to estimate bycatch mortality in so many fisheries and countries represents an enormous task that would require a substantial increase in expertise capacity, training, and financial and other resources. Other authors (e.g., Read, 2008) have noted this, and it likely partially explains why so little progress, relative to the level of need, has been made in assessing and thereby reducing bycatch worldwide over the last few decades.

Many areas throughout the world have never been surveyed to estimate marine mammal abundance (Kaschner et al., 2012). The marine mammal and wildlife biology research communities have conducted workshops to train scientists and managers in important techniques, such as line-transect analysis[10] and mark-recapture analysis[11]. The number and scope of marine mammal surveys and abundance estimates from South America have increased rapidly over the last several decades, in part due to education efforts of the Latin American Society for Aquatic Mammals (SOLAMAC). However, conducting marine mammal surveys to estimate abundance for all areas where problematic fisheries exist still represents an enormous task. The level of training will likely need to be expanded substantially.

There is also a clear need for capacity building of expertise in bycatch mitigation techniques. Workshops and reports by various organizations (e.g., FAO, 2018, 2021; Leaper and Calderan, 2018) are helping to disseminate information, but it will likely also take practical workshops or demonstrations of specific techniques, such as gear modification, for these methods to become more widely used in a timely manner.

Marine mammal bycatch in small-scale fisheries (often also termed 'artisanal' fisheries) is increasingly being recognized as an important conservation issue, but the products from such fisheries are not usually exported to the United States, so the US MMPA Import Provisions would not have direct relevance to many such fisheries. Small-scale fisheries are typically defined as fishing conducted from small, sometimes sail-powered, vessels with little advanced technology, as opposed to large-scale company-owned commercial fishing conducted from large modern vessels, though the exact definition is not such a simple dichotomy (Smith and Basurto, 2019). Nonetheless, small-scale fisheries have been estimated to represent as much as half of total global fisheries production and employ more than 99% of the world's fishermen, but bycatch is likely more a function of fishing gear type and target species, rather than whether a fishery is small- or large-scale (Jones et al., 2018). Bycatch risk from small-scale gillnet fisheries has been predicted to predominantly occur in tropical and sub-tropical regions dominated by low- and middle-income nations (Temple et al., 2021). Managing marine mammal bycatch in small-scale fisheries in developing nations is particularly difficult, given the dispersed nature of such fisheries and lack of resources, and because there is often little or no infrastructure in place for management. However, the level of marine mammal bycatch in small-scale fisheries, especially gillnets, can be high (Palacios and Gerrodette, 1996; Read, 2008; Brownell et al., 2019). Several important efforts are being made to build capacity in developing nations. One example is *The Consortium for the Conservation of the Atlantic Humpback Dolphin*, which was initiated in response to growing concerns regarding the declining conservation status of this species. Part of its focus has been to create the incentives, resources, and capacity needed for local stakeholders to engage in research and monitoring, such as marine mammal stranding response training for 'ecoguards' in Gabon, and a guide to identify marine mammals of the Atlantic coast of Africa[12]. More general efforts include *The Global Marine Animal Stranding Toolkit*[13] designed to provide training for responding to marine mammal strandings, including evaluation of human interactions such as bycatch. Another is a practical guide for the safe handling and release of bycaught small cetaceans from fishing gear produced by the Convention on the Conservation of Migratory Species of Wild Animals (CMS) and the World Wide Fund for Nature (Hamer and Minton, 2020). The International Whaling Commission (IWC) has established an *Expert Advisory Panel on Entanglement Response*[14] for large whales, which is developing best practices and conducting workshops around the world to train people to remove fishing gear from living whales. The IWC has also started a *Bycatch Mitigation Initiative*[15], whose initial focus is on small-scale/artisanal fisheries in coastal areas of developing countries; this initiative is expected to provide technical advice and capacity development/training. Many more such efforts will be needed to address bycatch in small-scale fisheries throughout the world.

Conclusion and Hopes for a Brighter Future for Marine Mammal Populations

The MMPA Import Provisions do not apply to all fisheries in the world that have bycatch of marine mammals; it

---

[10] https://workshops.distancesampling.org/

[11] http://www.phidot.org/forum/viewforum.php?f=8

[12] https://www.sousateuszii.org/resources/

[13] https://www.gmast.org/

[14] https://iwc.int/entanglement-response-network

[15] https://iwc.int/bycatch

applies only to fisheries that export to the United States. However, this represents a substantial increase in the number of fisheries that might be managed to limit marine mammal bycatch. To estimate bycatch and abundance of marine mammals for the more than 1,800 fisheries on NOAA's List of Foreign Fisheries would be a very large undertaking. To continue to export seafood to the United States, any given fishery would need to have its bycatch of marine mammals assessed as part of a regulatory program (or implement alternative measures to reduce bycatch) to apply for and receive a comparability finding. Additionally, by implementing a regulatory program to comply with the MMPA Import Provisions, countries may increase capacity to assess and manage marine mammal bycatch in all their fisheries (i.e., non-export fisheries).

It is not clear where funding and expertise to accomplish all this work will come from, although the US National Marine Fisheries Service has indicated a willingness to provide technical assistance contingent on available funding and resources[16]. For that reason, we have tried to provide guidance to ensure that the most important situations are given priority, similar to the framework for reducing marine mammal bycatch in the United States.

Williams et al. (2016) noted that the MMPA Import Provisions could have significant conservation benefits for marine mammal populations, but only if it is accompanied by substantial investments to boost scientific and compliance capacity. Otherwise, if these investments are not made, it could have little effect other than inflicting economic hardship on fishing communities, many of which are already poor and struggling. Similarly, after assessing the risk of marine mammal bycatch in small-scale fisheries in Southeast Asia, Johnson et al. (2017) concluded that export countries will have trouble achieving and documenting compliance with the rule within the 5-year exemption period.

Although we have highlighted the important issue of building capacity for conducting assessments of marine mammal bycatch throughout the world, even areas with considerable capacity have not necessarily achieved meaningful bycatch reduction. For example, several authors recently argue that the European Union (EU) has failed to adequately assess and mitigate bycatch of small cetaceans (Bearzi and Reeves, 2021; Carlén et al., 2021; Dolman et al., 2021; Rogan et al., 2021); they suggest this is due to diffuse management authority, a lack of political will, and the fact that the EU has no overarching quantitative conservation objectives. Rogan et al. (2021) suggest the EU establish a comprehensive plan that would include quantitative management objectives, generate estimates of abundance and bycatch mortality, and establish biological reference points to guide management actions; such a plan would incorporate much of the scheme we suggest here (**Figure 1**) and would be consistent with best practice guidelines for reference point-based management (Curtis et al., 2015).

Even when a specific management scheme is in place, such as in the United States, it can still be difficult to achieve substantial reduction of bycatch. US Take Reduction Plans have successfully reduced bycatch to below PBR for most stocks (McDonald et al., 2016), but there is at least one notable exception. Despite implementing multiple mitigation measures over two decades, bycatch mortality and serious injury of North Atlantic right whales continues to exceed the stock's PBR, and the population is declining (Kraus et al., 2016; Pace et al., 2017). The right whale case has been particularly complex, involving multiple fisheries and gear types, additional human-caused mortality from vessel strikes, and recent shifts in the whales' distribution to areas with unmitigated threats attributed to climate change, highlighting the challenges of bycatch reduction even for high-capacity regulatory programs.

The Food and Agriculture Organization of the UN (FAO), CMS, and other groups, including some non-governmental organizations, have been advocating for marine mammal bycatch reduction and supporting development of mitigation measures. These efforts have had some success in identifying key issues, and reducing bycatch in some areas. Our hope is that by providing a summary of best practices for an entire framework for managing marine mammal bycatch, this paper will contribute to the spread of effective management efforts to address the leading source of human-caused mortality of marine mammals.

AUTHOR CONTRIBUTIONS

All the authors contributed to the concept and overall outline of the manuscript, helped identify what specific content should be in each section, contributed to the manuscript revision, and read and approved the submitted version. KL and PW led the writing of the manuscript. MCS wrote sections of the manuscript. KL produced the **Supplementary Material S2, S3**. PW produced the **Supplementary Table 1**. GMS produced the **Supplementary Table 2**. TF and MCS created the figures. KL and PW created the **Boxes 1**, **2**.

FUNDING

Support for this project was provided by the Lenfest Ocean Program (Contract ID: #31008).

ACKNOWLEDGMENTS

We thank the journal reviewers for their helpful comments on the manuscript. We also thank Nancy Young, Robyn Angliss, and Jim Lee for helpful comments on an earlier version of the manuscript.

SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2021.757330/full#supplementary-material

---

[16] https://media.fisheries.noaa.gov/dam-migration/mmpa_import_rule_compliance_guide_april_2019_eng_508.pdf

REFERENCES

Anderson, R. C., Herrera, M., Ilangakoon, A. D., Koya, K. M., Moazzam, M., Mustika, P. L., et al. (2020). Cetacean bycatch in Indian Ocean tuna gillnet fisheries. *Endang. Sp. Res.* 41, 39–53. doi: 10.3354/esr01008

ASCOBANS. (2000). *Resolution No. 3 Incidental Take of Small Cetaceans, Annex 9c, 3rd Session of the Meeting of Parties, Bristol, United Kingdom, 26–28 July 2000*. Avaialable Online at: http://www.service-board.de/ascobans_neu/files/2000-6.pdf

Ashe, E., Williams, R., Morton, A., and Hammond, P. S. (2021). Disentangling natural and anthropogenic forms of mortality and serious injury in a poorly studied pelagic dolphin. *Front. Mar. Sci.* 8:606876. doi: 10.3389/fmars.2021.606876

Baird, R. W., and Webster, D. L. (2020). Using dolphins to catch tuna: Assessment of associations between pantropical spotted dolphins and yellowfin tuna hook and line fisheries in Hawai'i. *Fish. Res.* 230:105652. doi: 10.1016/j.fishres.2020.105652

Baird, R. W., Anderson, D. B., Kratofil, M. A., and Webster, D. L. (2021). Bringing the right fishermen to the table: Indices of overlap between endangered false killer whales and nearshore fisheries in Hawai'i. *Biol. Cons.* 255:108975. doi: 10.1016/j.biocon.2021.108975

Barbraud, C., Delord, K., Marteau, C., and Weimerskirch, H. (2009). Estimates of population size of white-chinned petrels and grey petrels at Kerguelen Islands and sensitivity to fisheries. *Anim. Conserv.* 12, 258–265. doi: 10.1111/j.1469-1795.2009.00248.x

Bearzi, G., and Reeves, R. R. (2021). Shifting baselines of cetacean conservation in Europe. *ICES J. Mar. Sci.* 2021:fsab137. doi: 10.1093/icesjms/fsab137

Berninsone, L. G., Bordino, P., Gnecco, M., Foutel, M., Mackay, A. I., and Werner, T. B. (2020). Switching gillnets to longlines: an alternative to mitigate the bycatch of franciscana dolphins (*Pontoporia blainvillei*) in Argentina. *Front. Mar. Sci.* 7:699. doi: 10.3389/fmars.2020.00699

Berrow, S., Cosgrove, R., Leeney, R. H., O'Brien, J., McGrath, D., Dalgard, J., et al. (2008). Effect of acoustic deterrents on the behaviour of common dolphins (*Delphinus delphis*). *J. Cet. Res. Manage.* 10, 227–234.

Bigelow, K. A., Kerstetter, D. W., Dancho, M. G., and Marchetti, J. A. (2012). Catch rates with variable strength circle hooks in the Hawaii-based tuna longline fishery. *Bull. Mar. Sci.* 88, 425–447. doi: 10.5343/bms.2011.1052

Brandon, J. R., Punt, A. E., Moreno, P., and Reeves, R. R. (2017). Towards a tier system approach for calculating limits on human-caused mortality of marine mammals. *ICES J. Mar. Sci.* 74, 877–887. doi: 10.1093/icesjms/fsw202

Braulik, G. T., Kasuga, M., Wittich, A., Kiszka, J., MacCaulay, J., Gillespie, D., et al. (2018). Cetacean rapid assessment: An approach to fill knowledge gaps and target conservation across large data deficient areas. *Aquat. Conserv. Mar. Freshw. Ecosyst.* 28, 216–230. doi: 10.1002/aqc.2833

Brownell Jr., R., Reeves, R. R., Read, A. J., Smith, B. D., Thomas, P. O., Ralls, K., et al. (2019). Bycatch in gillnet fisheries threatens Critically Endangered small cetaceans and other aquatic megafauna. *Endang. Spec. Res.* 40, 285–296. doi: 10.3354/esr00994

Buckland, S. T., Anderson, D. R., Burnham, K. P., Laake, J. L., Borchers, D. L., and Thomas, L. (2001). *Introduction to distance sampling*. Oxford: Oxford University Press.

Bunnefeld, N., Hoshino, E., and Milner-Gulland, E. J. (2011). Management strategy evaluation: a powerful tool for conservation? *Trends Ecol. Evol.* 26, 441–447. doi: 10.1016/j.tree.2011.05.003

Campbell, R., Holley, D., Christianopolous, D., and Gales, N. G. (2008). Mitigation of incidental mortality of Australian sea lions in the west coast rock lobster fishery. *Endang. Species. Res.* 5, 345–358. doi: 10.3354/esr00155

Carlén, I., Nunny, L., and Simmonds, M. P. (2021). Out of sight, out of mind: how conservation is failing european porpoises. *Front. Mar. Sci.* 8:617478. doi: 10.3389/fmars.2021.617478

Carretta, J. V., and Barlow, J. (2011). Long-term effectiveness, failure rates, and "dinner bell" properties of acoustic pingers in a gillnet fishery. *Mar. Tech. Soc. J.* 45, 7–19. doi: 10.4031/MTSJ.45.5.3

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

Chao, A. (2001). An overview of closed capture-recapture models. *J. Ag. Biol. Environ. Stat.* 6, 158–178. doi: 10.1198/108571101750524670

Collins, T., Braulik, G. T., and Perrin, W. (2017). *Sousa teuszii* (errata version published in 2018). *IUCN Red List Threat. Sp.* 2017:e.T20425A123792572. doi: 10.2305/IUCN.UK.2017-3.RLTS.T20425A50372734.en

Curtis, K. A., Moore, J. E., Boyd, C., Dillingham, P. W., Lewison, R. L., Taylor, B. et al. (2015). Managing catch of marine megafauna: Guidelines for setting limit reference points. *Mar. Policy* 61, 249–263. doi: 10.1016/j.marpol.2015.07.002

D'Agrosa, C., Lennert-Cody, C., and Vidal, O. (2000). Vaquita bycatch in Mexico's artisanal gillnet fishery: Driving a small population to extinction. *Conserv. Biol.* 14, 1100–1119. doi: 10.1046/j.1523-1739.2000.98191.x

Dawson, S. M., Northridge, S., Waples, D., and Read, A. J. (2013). To ping or not to ping: the Use of active acoustic devices in mitigating interactions between small cetaceans and gillnet fisheries. *Endang. Spec. Res.* 19, 201–221.

de Boer, M. N., Saulino, J. T., Leopold, M. F., Reijnders, P. J. F., and Simmonds, M. P. (2012). Interactions between short-beaked common dolphin (*Delphinus delphis*) and the winter pelagic pair-trawl fishery off Southwest England (UK). *Int. J. Biodiv. Con.* 4, 481–499. doi: 10.5897/ijbc12.016

de Boer, M. N., Saulino, J. T., Van Waerebeek, K., and Aarts, G. (2016). Under pressure: cetaceans and fisheries co-occurrence off the coasts of Ghana and Côte d'Ivoire (Gulf of Guinea). *Front. Mar. Sci.* 3:178. doi: 10.3389/fmars.2016.00178

de Quiros, Y. B., Hartwick, M., Rotstein, D. S., Garner, M. M., Bogomolni, A., Greer, W., et al. (2017). Discrimination between bycatch and other causes of cetacean and pinniped stranding. *Dis. Aquat. Organ.* 127, 83–95. doi: 10.3354/dao03189

Dillingham, P. W., and Fletcher, D. (2008). Estimating the ability of birds to sustain additional human-caused mortalities using a simple decision rule and allometric relationships. *Biol. Cons.* 141, 1783–1792. doi: 10.1016/j.biocon.2008.04.022

Dolman, S. J., Evans, P. G. H., Ritter, F., Simmonds, M. P., and Swabe, J. (2021). Implications of new technical measures regulation for cetacean bycatch in European waters. *Mar. Policy* 124:104320. doi: 10.1016/j.marpol.2020.104320

Emery, T. J., Noriega, R., Williams, A. J., and Larcombe, J. (2019). Changes in logbook reporting by commercial fishers following the implementation of electronic monitoring in Australian Commonwealth fisheries. *Mar. Policy* 104, 135–145. doi: 10.1016/j.marpol.2019.01.018

FAO. (2018). *Report of the Expert Workshop on Means and Methods for Reducing Marine Mammal Mortality in Fishing and Aquaculture Operations. Rome, 20–23 March 2018. FAO Fisheries and Aquaculture Report No.1231*. Rome: FAO.

FAO. (2021). *Fishing operations. Guidelines to prevent and reduce bycatch of marine mammals in capture fisheries. FAO Technical Guidelines for Responsible Fisheries No.1*. Rome: FAO, doi: 10.4060/cb2887en

Franco-Trecu, V., Szephegyi, M. N., Doño, F., Forselledo, R., Reyes, F., Passadore, C., et al. (2019). Marine mammal bycatch by the industrial bottom trawl fishery at the Río de la Plata Estuary and the adjacent Atlantic Ocean. *Lat. Am. J. Aquat. Res.* 47, 89–101. doi: 10.3856/vol47-issue1-fulltext-10

Gales, N. (1995). *Hooker's sea lion (Phocarctos hookeri) recovery plan. Department of Conservation Threatened Species Recovery Plan Series No. 17.* Available Online at: https://www.doc.govt.nz/globalassets/documents/science-and-technical/tsrp17.pdf

Goldsworthy, S. D., and Page, B. (2007). A risk-assessment approach to evaluating the significance of seal bycatch in two Australian fisheries. *Biol. Conserv.* 139, 269–285. doi: 10.1016/j.biocon.2007.07.010

Hamer, D. J., Childerhouse, S. J., and Gales, N. J. (2012). Odontocete bycatch and depredation in longline fisheries: A review of available literature and of potential solutions. *Mar. Mam. Sci.* 28, E345–E374. doi: 10.1111/j.1748-7692.2011.00544.x

Hamer, D., and Minton, G. (2020). *Guidelines for the Safe and Humane Handling and Release of Bycaught Small Cetaceans from Fishing Gear. CMS Technical Series No. 43.* Bonn: UNEP/CMS Secretariat, 50.

Hamilton, S., and Baker, G. B. (2019). Technical mitigation to reduce marine mammal bycatch and entanglement in commercial fishing gear: lessons learnt and future directions. *Rev. Fish Biol. Fish.* 29, 223–247. doi: 10.1007/s11160-019-09550-6

Hammond, P. S., Francis, T. B., Heinemann, D., Long, K. J., Moore, J. E., Punt, A. E., et al. (2021). Estimating the abundance of marine mammal populations. *Front. Mar. Sci.* 8:735770. doi: 10.3389/fmars.2021.735770

Hammond, P. S., Mizroch, S. A., and Donovan, G. P. (1990). *Individual Recognition of Cetaceans: Use of Photo-Identification and Other Techniques to Estimate*

Population Parameters. Rep. Int. Whal. Commn (Special Issue 12). Cambridge: International Whaling Commission, 440.

Hamner, R. M., Wade, P. R., Oremus, M., Stanley, M., Brown, P., Constantine, R., et al. (2014). Critically low abundance and limits to human-related mortality for the Maui's dolphin. *Endang. Spec. Res.* 26, 87–92. doi: 10.3354/esr00621

Harcourt, R. G. (2001). Advances in New Zealand mammalogy 1990-2000: Pinnipeds. *J. R. Soc. N. Z.* 31, 135–160. doi: 10.1080/03014223.2001.9517644

Hines, E., Ponnampalam, L. S., Junchompoo, C., Peter, C., Vu, L., Huynh, T., et al. (2020). Getting to the bottom of bycatch: a GIS-based toolbox to assess the risk of marine mammal bycatch. *Endang. Spec. Res.* 42, 37–57. doi: 10.3354/esr01037

ICES. (2020). *EU request on emergency measures to prevent bycatch of common dolphin (Delphinus delphis) and Baltic Proper harbour porpoise (Phocoena phocoena) in the Northeast Atlantic. In Report of the ICES Advisory Committee, 2020. ICES Advice 2020, sr.2020.04*. Available Online at: https://www.ices.dk/sites/pub/Publication%20Reports/Advice/2020/Special_Requests/eu.2020.04.pdf

Johnson, A. F., Caillat, M., Verutes, G. M., Peter, C., Junchompoo, C., Long, V., et al. (2017). Poor fisheries struggle with U.S. import rule. *Science* 355, 1031–1032. doi: 10.1126/science.aam9153

Johnson, A., Salvador, G., Kenney, J., Robbins, J., Kraus, S., Landry, S., et al. (2005). Fishing gear involved in entanglements of right and humpback whales. *Mar. Mamm. Sci.* 21, 635–645. doi: 10.1111/j.1748-7692.2005.tb01256.x

Jones, B. L., Unsworth, R. K. F., Udagedara, S., and Cullen-Unsworth, L. C. (2018). Conservation concerns of small-scale fisheries: by-catch impacts of a shrimp and finfish fishery in a Sri Lankan lagoon. *Front. Mar. Sci.* 5:52. doi: 10.3389/fmars.2018.00052

Kaschner, K., Quick, N. J., Jewell, R., Williams, R., and Harris, C. M. (2012). Global coverage of cetacean line-transect surveys: Status quo, data gaps and future challenges. *PLoS One* 7:e44075. doi: 10.1371/journal.pone.0044075

Knowlton, A. R., Hamilton, P. K., Marx, M. K., Pettis, H. M., and Kraus, S. D. (2012). Monitoring North Atlantic right whale *Eubalaena glacialis* entanglement rates: a 30 yr retrospective. *Mar. Ecol. Prog. Ser.* 466, 293–302. doi: 10.3354/meps09923

Kraus, S. D., Brown, M. W., Caswell, H., Clark, C. W., Fujiwara, M., Hamilton, P. K., et al. (2005). North Atlantic right whales in crisis. *Science* 309, 561–562. doi: 10.1126/science.1111200

Kraus, S. D., Kenney, R. D., Mayo, C. A., McLellan, W. A., Moore, M. J., and Nowacek, D. P. (2016). Recent scientific publications cast doubt on North Atlantic right whale future. *Front. Mar. Sci.* 3:137. doi: 10.3389/fmars.2016.00137

Le Boeuf, B. J., Kenyon, K. W., and Villa-Ramirez, B. (1986). The Caribbean monk seal is extinct. *Mar. Mamm. Sci.* 2, 70–72. doi: 10.1111/j.1748-7692.1986.tb00028.x

Leaper, R., and Calderan, S. (2018). *Review of methods used to reduce risks of cetacean bycatch and entanglements CMS Technical Series No. 38*. Bonn, Germany: UNEP/CMS Secretariat.

Lowry, L. (2017). Zalophus japonicus (amended version of 2015 assessment). *IUCN Red List Threat. Sp.* 2017:e.T41667A113089431. doi: 10.2305/IUCN.UK.2017-1.RLTS.T41667A113089431.en

Mangi, S. C., Smith, S., and Catchpole, T. L. (2016). Assessing the capability and willingness of skippers towards fishing industry-led data collection. *Ocean Coast. Manage.* 134, 11–19. doi: 10.1016/j.ocecoaman.2016.09.027

Martien, K. K., Lang, A. R., Taylor, B. L., Rosel, P. E., Simmons, S. E., Oleson, E. M., et al. (2019). *The DIP Delineation Handbook: A Guide to Using Multiple Lines of Evidence to Delineate Demographically Independent Populations of Marine Mammals. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-SWFSC-622*. Available Online at: https://repository.library.noaa.gov/view/noaa/22660.

McDonald, S. L., Lewison, R. L., and Read, A. J. (2016). Evaluating the efficacy of environmental legislation: a case study from the US marine mammal take reduction planning process, *Glob. Ecol. Conserv.* 5, 1–11. doi: 10.1016/j.gecco.2015.11.009

McLellan, W. A., Arthur, L. H., Mallette, S. D., Thornton, S. W., McAlarney, R. J., Read, A. J., et al. (2015). Longline hook testing in the mouths of pelagic odontocetes. *ICES J. Mar. Sci.* 72, 1706–1713. doi: 10.1093/icesjms/fsu181

Mead, J. I., Spiess, A. E., and Sobolik, K. D. (2000). Skeleton of extinct North American sea mink (*Mustela macrodon*). *Quat. Res.* 53, 247–262. doi: 10.1006/qres.1999.2109

Methot, R. D., Tromble, G. R., Lambert, D. M., and Greene, K. E. (2014). Implementing a science-based system for preventing overfishing and guiding sustainable fisheries in the United States. ICES J. *Mar. Sci.* 71, 183–194. doi: 10.1093/icesjms/fst119

Milner-Gulland, E. J., and Akçakaya, H. R. (2001). Sustainability indices for exploited populations. *Trends Ecol. Evol.* 16, 686–692. doi: 10.1016/s0169-5347(01)02278-9

Moore, J. E., Curtis, K. A., Lewison, R. L., Dillingham, P. W., Cope, J. M., Fordham, S. V., et al. (2013). Evaluating sustainability of fisheries bycatch mortality for marine megafauna: a review of conservation reference points for data-limited populations. *Environ. Conserv.* 40, 329–344. doi: 10.1017/S037689291300012X

Moore, J. E., Cox, T. M., Lewison, R. L., Read, A. J., Bjorkland, R., McDonald, S. L., et al. (2010). An interview-based approach for assessing marine mammal and sea turtle captures in artisanal fisheries. *Biol. Cons.* 143, 795–805. doi: 10.7717/peerj.3151

Moore, M. J., van der Hoop, J., Barco, S. G., Costidis, A. M., Gulland, F. M., Jepson, P. D., et al. (2013). Criteria and case definitions for serious injury and death of pinnipeds and cetaceans caused by anthropogenic trauma. *Dis. Aquat. Organ.* 103, 229–264. doi: 10.3354/dao02566

Mustika, P. L. K., Purnomo, F. S., and Northridge, S. (2014). *A pilot study to identify the extent of small cetacean bycatch in Indonesia using fisher interview and stranding data as proxies*. Denpasar: The International Whaling Commission, 61. http://www.cetasindonesia.org/uploads/5/6/0/1/56013937/mustika_et_al_2014_updated_cetacean_bycatch_report_for_iwc_with_id_abstract.pdf

Myers, H. J., Moore, M. J., Baumgartner, M. F., Brillant, S. W., Katona, S. K., Knowlton, A. R., et al. (2019). Ropeless fishing to prevent large whale entanglements: Ropeless Consortium report. *Mar. Pol.* 107:103587. doi: 10.1016/j.marpol.2019.103587

Nagtzaam, G. (2009). The International Whaling Commission and the elusive great white whale of preservationism. *William. Mary Environ. Law Pol. Rev.* 33:375. doi: 10.4337/9781849803489.00011

National Marine Fisheries Service. (2012). *NOAA Fisheries Policy Directive 02-238-01: Process for distinguishing serious from non-serious injury of marine mammals*. 42. Available Online at: https://media.fisheries.noaa.gov/dam-migration/02-238-01.pdf

Otis, D. L., Burnham, K. P., White, G. C., and Anderson, D. R. (1978). Statistical inference from capture data on closed animal populations. *Wildlife Monogr.* 62, 3–135. Available online at: https://www.jstor.org/stable/3830650

Pace, R. M. III, Corkeron, P. J., and Kraus, S. D. (2017). State-space mark-recapture estimates reveal a recent decline in abundance of North Atlantic right whales. *Ecol. Evol.* 7, 8730–8741. doi: 10.1002/ece3.3406

Page, B., Mckenzie, J., Mcintosh, R., Baylis, A., Morrissey, A., Calvert, N., et al. (2004). Entanglement of Australian sea lions and New Zealand fur seals in lost fishing gear and other marine debris before and after government and industry attempts to reduce the problem. *Mar. Pollut. Bull.* 49, 33–42. doi: 10.1016/j.marpolbul.2004.01.006

Palacios, D. M., and Gerrodette, T. (1996). *Potential Impact of Artisanal Gillnet Fisheries on Small Cetacean Populations in the Eastern Tropical Pacific. Administrative Report JL–96-11*. La Jolla, CA: NMFS, Southwest Fisheries Science Center.

Palka, D. L., Rossman, M. C., VanAtten, A. S., and Orphanides, C. D. (2008). Effect of pingers on harbour porpoise (*Phocoena phocoena*) bycatch in the U.S. Northeast gillnet fishery. *J. Cet. Res. Manage.* 10, 217–226.

Pardalou, A., and Tsikliras, A. C. (2018). Anecdotal information on dolphin-fisheries interactions based on empirical knowledge of fishers in the northeastern Mediterranean Sea. *Ethics Sci. Environ. Pol.* 18, 1–8. doi: 10.3354/esep00179

Parry, L., Barlow, J., and Peres, C. A. (2009). Hunting for sustainability in tropical secondary forests. *Conserv. Biol.* 23, 1270–1280. doi: 10.1111/j.1523-1739.2009.01224.x

Perrin, W. F. (1969). Using porpoise to catch tuna. *World Fish.* 18, 42–45.

Perrin, W. F., Donovan, G. P., and Barlow, J. (1994). *Gillnets and Cetaceans. Rep. Int. Whal. Comm. (Special Issue 15).* Cambridge: International Whaling Commission, 629.

Pierce, G. J., Weir, C., Gutierrez, P., Verutes, G., Fontaine, M. C., Gonzalez, A. H., et al. (2020). "Is Iberian harbour porpoise (*Phocoena phocoena*) threatened by interactions with fisheries?," in *Paper Presented at the Scientific Committee of the International Whaling Commission SC/68B/SM04 Rev2*. Available online at: https://archive.iwc.int/pages/view.php?ref=17445&k=ac428ca5f5

Punt, A. E., Sepúlveda, M., Siple, M. C., Moore, J., Francis, T. B., Hammond, P. S., et al. (2021a). Assessing pinniped bycatch mortality with uncertainty in abundance and post-release mortality: a Case study from Chile. *Fish Res.* 235:105816. doi: 10.1016/j.fishres.2020.105816

Punt, A. E., Siple, M. C., Francis, T. B., Hammond, P. S., Heinemann, D., Long, K. J., et al. (2021b). Can we manage marine mammal bycatch effectively in low-data environments? *J. Appl. Ecol.* 58, 596–607. doi: 10.1111/1365-2664.13816

Punt, A. E., Siple, M., Sigurðsson, G. M., Víkingsson, G., Francis, T. B., Granquist, S. M., et al. (2020). Evaluating management strategies for marine mammal populations: an example for multiple species and multiple fishing sectors in Iceland. *Can. J. Fish. Aquat. Sci.* 77, 1316–1331. doi: 10.1139/cjfas-2019-0386

Read, A. J. (2008). The looming crisis: interactions between marine mammals and fisheries. *J. Mamm.* 89, 541–548. doi: 10.1644/07-MAMM-S-315R1.1

Read, A. J., Drinker, P., and Northridge, S. (2006). Bycatch of marine mammals in U.S. and global fisheries. *Conserv. Biol.* 20, 163–169. doi: 10.1111/j.1523-1739.2006.00338.x

Reeves, R. R., McClellan, K., and Werner, T. B. (2013). Marine mammal bycatch in gillnet and other entangling net fisheries, 1990 to 2011. *Endang. Spec. Res.* 20, 71–97. doi: 10.3354/esr00481

Rogan, E., Read, A. J., and Berggren, P. (2021). Empty promises: The European Union is failing to protect dolphins and porpoises from fisheries by-catch. *Fish. Fish.* 22, 865–869. doi: 10.1111/faf.12556

Rojas-Bracho, L., and Taylor, B. L. (2017). *Phocoena sinus*. *IUCN Red List Threat. Sp.* 2017:e.T17028A50370296. doi: 10.2305/IUCN.UK.2017-2.RLTS.T17028A50370296.en

Schwarz, C. J., and Seber, G. A. F. (1999). Estimating animal abundance: review III. *Stat. Sci.* 14, 427–456. doi: 10.1214/ss/1009212521

Siple, M. (2021). *mcsiple/mmrefpoints: First release (v0.0.1)*. Genève: Zenodo, doi: 10.5281/zenodo.4758402

Smith, B. D., Wang, D., Braulik, G. T., Reeves, R., Zhou, K., Barlow, J., et al. (2017). Lipotes vexillifer. *IUCN Red List Threat. Sp.* 2017:e.T12119A50362206. doi: 10.2305/IUCN.UK.2017-3.RLTS.T12119A50362206.en

Smith, H., and Basurto, X. (2019). Defining small-scale fisheries and examining the role of science in shaping perceptions of who and what counts: A systematic review. *Front. Mar. Sci.* 6:236. doi: 10.3389/fmars.2019.00236

Smith, T. (1979). *Report of the Status of Porpoise Stocks Workshop Administrative Report LJ–79–41.* La Jolla, CA: Southwest Fisheries Science Center.

Smith, T. (1983). Changes in size of three dolphin (*Stenella* spp.) populations in the eastern tropical Pacific. *Fish. Bull. U.S.* 81, 1–13.

Taylor, B. L., Rojas-Bracho, L., Moore, J., Jaramillo-Legorreta, A., Ver Hoef, J. M., Cardenas-Hinojosa, G., et al. (2017). Extinction is imminent for Mexico's endemic porpoise unless fishery bycatch is eliminated. *Conserv. Lett.* 10, 588–595. doi: 10.1111/conl.12331

Taylor, B. L., Wade, P. R., DeMaster, D. P., and Barlow, J. (2000). Incorporating uncertainty into management models for marine mammals. *Conserv. Biol.* 14, 1243–1252. doi: 10.1046/j.1523-1739.2000.99409.x

Temple, A. J., Westmerland, E., and Berggren, P. (2021). By-catch risk for toothed whales in global small-scale fisheries. *Fish Fish.* 1–5. doi: 10.1111/faf.12581

Turvey, S. T., and Risley, C. L. (2006). Modelling the extinction of Steller's sea cow. *Biol. Lett.* 2, 94–97. doi: 10.1098/rsbl.2005.0415

van der Hoop, J. M., Corkeron, P., and Moore, M. (2016). Entanglement is a costly life-history stage in large whales. *Ecol. Evol.* 7, 92–106. doi: 10.1002/ece3.2615

Verutes, G. M., Johnson, A. F., Caillat, M., Ponnampalam, L. S., Peter, C., Vu, L., et al. (2020). Using GIS and stakeholder involvement to innovate marine mammal bycatch risk assessment in data- limited fisheries. *PLoS One* 15:e0237835. doi: 10.1371/journal.pone.0237835

Wade, P. R. (1995). Revised estimates of dolphin kill in the eastern tropical Pacific, 1959-1972. *Fish. Bull. U.S.* 93, 345–354. https://spo.nmfs.noaa.gov/sites/default/files/pdf-content/1995/932/wade.pdf

Wade, P. R. (1998). Calculating limits to the allowable human-caused mortality of cetaceans and pinnipeds. *Mar. Mamm. Sci.* 14, 1–37. doi: 10.1111/j.1748-7692.1998.tb00688.x

Wade, P. R. (2018). "Stock Assessment," in *Encyclopedia of Marine Mammals*, 3rd Edn, eds B. Wursig, J. G. M. Thewissen, and K. M. Kovacs (San Diego: Academic Press), 1110–1115.

Walsh, W. A., Kleiber, P., and McCracken, M. (2002). Comparison of logbook reports of incidental blue shark catch rates by Hawaii-based longline vessels to fishery observer data by application of a generalized additive model. *Fish. Res.* 58, 79–94. doi: 10.1016/S0165-7836(01)00361-7

Weinbaum, K. Z., Brashares, J. S., Golden, C. D., and Getz, W. M. (2013). Searching for sustainability: are assessments of wildlife harvests behind the times? *Ecol. Lett.* 16, 99–111. doi: 10.1111/ele.12008

Welch, H., Brodie, S., Jacox, M. G., Bograd, S. J., and Hazen, E. L. (2020). Decision-support tools for dynamic management. *Conserv. Biol.* 34, 589–599. doi: 10.1111/cobi.13417

Wells, R. S., Allen, J. B., Hofmann, S., Bassos-Hull, K., Fauquier, D. A., Barros, N. B., et al. (2008). Consequences of injuries on survival and reproduction of common bottlenose dolphins (*Tursiops truncatus*) along the west coast of Florida. *Mar. Mamm. Sci.* 24, 774–794. doi: 10.1111/j.1748-7692.2008.00212.x

Werner, T. B., Northridge, S., Press, K. M., and Young, N. (2015). Mitigating bycatch and depredation of marine mammals in longline fisheries.ICES J. *Mar. Sci.* 72, 1576–1586. doi: 10.1093/icesjms/fsv092

Williams, R., Burgess, M. G., Ashe, E., Gaines, S. D., and Reeves, R. R. (2016). US seafood import restriction presents opportunity and risk. *Science* 354, 1372–1374. doi: 10.1126/science.aai8222

Williams, R., Gero, S., Bejder, L., Calambokidis, J., Kraus, S. D., Lusseau, D., et al. (2011). Underestimating the damage: interpreting cetacean carcass recoveries in the context of the Deepwater Horizon/BP incident. *Conserv. Lett.* 4, 228–233. doi: 10.1111/j.1755-263x.2011.00168.x

Williams, R., Grand, J., Hooker, S. K., Buckland, S. T., Reeves, R. R., Rojas-Bracho, L., et al. (2014). Prioritizing global marine mammal habitats using density maps in place of range maps. *Ecography* 37, 212–220. doi: 10.1111/j.1600-0587.2013.00479.x

Zydelis, R., Bellebaum, J., Österblom, H., Vetemaa, M., Schirmeister, B., Stipniece, A., et al. (2009). Bycatch in gillnet fisheries – An overlooked threat to waterbird populations. *Biol. Cons.* 142, 1269–1281. doi: 10.1016/j.biocon.2009.02.025

Author Disclaimer: The views expressed are those of the authors and do not necessarily reflect the views of the Lenfest Ocean Program or the Pew Charitable Trusts. The scientific results and conclusions, as well as any views or opinions expressed herein, are those of the authors and do not necessarily reflect the views of NOAA or the U.S. Department of Commerce. Export countries working with the US National Marine Fisheries Service should consider the guidance given in this paper in the context of the requirements set forth by the US MMPA import provisions in 50 CFR $ 216.24.

Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

frontiers in Marine Science

ORIGINAL RESEARCH
published: 01 December 2021
doi: 10.3389/fmars.2021.779066

Using Case Studies to Investigate Cetacean Bycatch/Interaction Under-Reporting in Countries With Reporting Legislation

***Charla Jean Basran*[1]* *and Guðjón Már Sigurðsson*[2]**

[1] Húsavík Research Centre, University of Iceland, Húsavík, Iceland, [2] Marine and Freshwater Research Institute, Hafnarfjörður, Iceland

Accurate reporting of cetacean bycatch in/interaction with fishing gear in fisher logbooks would be of immense scientific value; however, despite some countries having mandatory reporting laws, logbook reporting is widely considered unreliable and cetacean catches are thought to be under-reported. Despite this widespread notion of logbook unreliability, under-reporting has rarely been quantified. For this study, initially we compiled the first comprehensive legislation summary for countries which have cetacean bycatch/interaction reporting laws. We then used data provided by government and research agencies in three case study countries (New Zealand, United States, and Iceland) to test for differences between logbook and observer reported cetacean bycatch. Comparisons were made using paired t-tests and Wilcoxon tests with a set alpha of 0.05. Overall, cetacean bycatch recorded by observers was higher than that from fisher logbooks by an average of 774% in trawls, 7348% in nets, and 1725% in hook and line gears. When combining all years of data available, fisher logbook cetacean catch per unit efforts or average number of individuals caught were significantly less than those from observer data for all gear types that could be examined in all countries, except for lining in New Zealand. Overall, there was significant under-reporting in the case study countries despite differences in geographic location, cetacean species and density and EEZ size, suggesting these results would likely be similar in many countries with comparable, well-developed fishing industries. Under-reporting in logbooks, despite laws, was clearly quantified and it is known that fishers have little incentive to report and have concerns over negative repercussions to the industry over bycatch issues. If logbook reporting is to continue in some fisheries, clearer legislation, simplified reporting using new technology (such as smartphone apps) and combination with electronic monitoring cameras to verify compliance may improve reporting accuracy. The introduction of electronic monitoring, given its lower cost compared to observer programs and high accuracy, may be the most viable option to obtain reliable cetacean bycatch estimates, and could be considered to replace logbook reporting altogether.

Keywords: bycatch, under-reporting, entanglement, cetaceans, fisheries, observers, logbooks, CPUE

Edited by:
Guillermo Luna-Jorquera,
Universidad Católica del Norte, Chile

Reviewed by:
Finn Larsen,
Technical University of Denmark,
Denmark
Arne Bjørge,
Norwegian Institute of Marine
Research (IMR), Norway

****Correspondence:***
Charla Jean Basran
cjb2@hi.is

Specialty section:
This article was submitted to
Marine Megafauna,
a section of the journal
Frontiers in Marine Science

Received: *17 September 2021*
Accepted: *12 November 2021*
Published: *01 December 2021*

Citation:
Basran CJ and Sigurðsson GM
(2021) Using Case Studies
to Investigate Cetacean
Bycatch/Interaction Under-Reporting
in Countries With Reporting
Legislation.
Front. Mar. Sci. 8:779066.
doi: 10.3389/fmars.2021.779066

INTRODUCTION

Cetacean bycatch in or interaction with a wide array of fishing gear types is a global issue which is difficult to quantify and manage. Bycatch is defined as the capture of non-target species in fishing gear which died or were likely fatally injured (Hall, 1996). This is more often the case for small cetaceans that cannot free themselves from gear. Bycatch is considered one of the main causes of anthropogenic mortality in cetaceans and it was previously estimated that over 300,000 cetaceans are killed or seriously injured annually in fisheries world-wide (Read et al., 2006). Much of the fishing gear that has been implicated in such incidents, such as gillnets, is set in shallow waters with low visibility, or deep waters with low light, where it is very unlikely for cetaceans to see the gear in time to avoid it (Kastelein et al., 2001). Though odontocetes use echolocation and mysticetes use hearing and interpretation of sounds for orientation, the acoustic reflectivity of nets is relatively weak, meaning the animals may also have difficulty detecting them (Lien et al., 1990; Au and Jones, 1991; Mooney et al., 2007).

Due to the detectability issues for the cetaceans, bycatch and interaction with fishing gear poses the serious threat of extinction to several small, vulnerable cetacean populations, such as the vaquita (e.g., Taylor et al., 2017) and the North Atlantic right whale (e.g., Moore et al., 2021), by causing unsustainable mortality. Even when not fatal, entanglements can potentially have negative impacts on the individual, such as lowered reproductive success (Robbins and Mattila, 2001; Rolland et al., 2017), which can then impact the recruitment rate of the population.

It is imperative to understand the magnitude of cetacean bycatch and interaction issues in fisheries to implement sustainable fishing practices and conserve cetacean populations. Having fishers log all cetacean bycatch/interactions would be of immense scientific value, and therefore some countries have developed logbook reporting systems for their respective fisheries and have made this reporting mandatory by law; however, accurate and reliable reporting is rare, and few countries have systematically reported data (Read et al., 2006). Even in countries where reporting is mandatory, under-reporting of bycatch and interactions is still recognized as a serious issue (e.g., Cornish et al., 2004). This has led to the need for onboard observers to monitor and record bycatch/interactions, but this is a costly solution that is not viable for all fisheries (Reeves et al., 2013) and to-date it has proven difficult for many countries to be able to quantify bycatch in their fisheries and in turn make informed management decisions (Young and Iudicello, 2007). Understanding and managing cetacean bycatch has become a particularly important issue for governments worldwide since the United States of America (USA) began enforcing a rule in the Marine Mammal Protection Act stating that all imported fish products must come from fisheries that do not cause serious harm to marine mammal populations (NOAA, 2019). This rule came into effect on January 1, 2017 (Federal Register, 2016); however, an initial exemption period of 5 years was granted for countries to work on implementing proper marine mammal bycatch management (NOAA, 2019), which was later extended by 1 year (Federal Register, 2020), meaning proper management practices need to be adopted by fisheries exporting their products to the United States by 2023.

In this study, we reviewed which countries, of the 30 with the largest fisheries industries (FAO, 2018), have laws requiring fishers to report cetacean bycatch/interactions, and then further reviewed the details of the legislation of each of these countries. We then used data from four of these countries; New Zealand, Iceland, United States, and Norway, for case studies investigating the amount of cetacean bycatch/interaction that is reported in different gear categories in each country. We then compared the reported cetacean bycatch/interaction rates in fisher logbooks to reported rates calculated from observer programs for three of these countries. Finally, we investigated logbook reporting over time and recommended strategies to improve cetacean bycatch/interaction monitoring, which could be used to improve upon systems already in place or implemented in countries that have yet to tackle cetacean bycatch management in their fisheries.

Countries With Cetacean Bycatch/Interaction Reporting Legislation

Twelve countries out of 30 were identified as having legislation that included mandatory cetacean bycatch/interaction reporting. The laws differ between the countries in terms of what size vessels have to report and what details must be reported. Below is a summary of the cetacean bycatch reporting laws for each country individually. We acknowledge that this list may not be complete and will update it, once new information is available to us, at this website: https://heima.hafro.is/~gudjon/marinemammalbycatch.html.

United States

The United States has a federal Marine Mammal Protection Act (MMPA) that was established in 1972 (U.S. Fish and Wildlife Service International Affairs, 2020). This act prohibits any take of marine mammals. Starting in 1994, under code § 1387, it became mandatory for all vessels fishing in a "Category I or II" fishery to apply for a "marine mammal authorization certificate" from the National Oceanic and Atmospheric Administration (NOAA) fisheries department which allows the vessel to incidentally take marine mammals without being in violation of the MMPA, so long as they are abiding to other regulations (Legal Information Institute, 2020). Category I "designates fisheries with frequent deaths and serious injuries [to marine mammals] incidental to commercial fishing" and Category II "designates fisheries with occasional deaths and serious injuries [to marine mammals]" (NOAA Fisheries, 2020a). In addition, under this same code, it became mandatory for all fishing vessels to report any death or serious injury of a marine mammal during fishing activities within 48 h of the incident (Legal Information Institute, 2020). The reports must include the vessel identification, the information of the owner, the name of the fishery, and the information about the incident including the species of marine mammal, the type of injury, and the date, time, and location. In addition to the mandatory reporting, NOAA has five hotlines for the different regions of the country to report a marine mammal in distress, as well as a smartphone app (NOAA Fisheries, 2020b).

Canada

Canada's Fisheries Act, first established in 1985, later included Marine Mammal Regulations in 1993 (Government of Canada, 2020). These regulations govern the protection and harvest of marine mammals in Canadian waters. The regulations were further amended in 2018 to include Accidental Contact with Marine Mammals regulations. These regulations made it mandatory for all interactions between vehicles or fishing gear and marine mammals to be reported to the Minister of Fisheries or reported in a mandatory logbook, even if the animal did not appear to be injured or deceased (Government of Canada, 2018). The incidents should be reported no later than 48 h after the end of a fishing trip and must include the type of vehicle and/or type of fishing gear involved, the vessel name and owner, and specific information about the incident including species of marine mammal, date, time, location, and condition of the animal (Fisheries and Oceans Canada, 2020a). In addition to this mandatory reporting, Canada has eight hotlines set up in different regions of the country to report incidents of marine mammal entanglement where the animal is in need of professional assistance (Fisheries and Oceans Canada, 2020b).

Australia

Australia's Environment Protection and Biodiversity Conservation Act (EPBC) was implemented in 1999 (Australian Government Department of Agriculture Water and the Environment, 2020a). Under this act, all cetaceans are listed as protected species and the rules for all Commonwealth fisheries state "all interactions with EPBC Act–listed species, whether authorized or not, must be reported to the Department of the Environment and Energy" (Australian Government Department of Agriculture and Water Resources, 2018). All Australian Commonwealth fisheries are managed by the government's Australian Fisheries Management Authority (AFMA) which permits accredited fisheries to incidentally interact with protected species without it being a punishable offense under the EPBC, so long as these interactions are recorded in AFMA fishing logbook (AFMA, 2020). An interaction includes "any physical contact a person, boat or gear has with a protected species." In addition to incident reporting in logbooks, Australia also has nine agencies around the country which can be contacted in the event that a whale is witnessed entangled in fishing gear and is in need of professional assistance (Australian Government Department of Agriculture Water and the Environment, 2020b).

New Zealand

New Zealand implemented a Marine Mammal Protection Act, similar to the Act in United States, in 1978. Section 16 of this Act was created in 1996 and covers "Reporting of accidental death or injuries" which states that any person who incidentally kills or injures a marine mammal while fishing must both record the incident in the official logbook and submit a written report to the fishery officer within 48 h of returning to port (Parliamentary Counsel Office-New Zealand and Legislation, 1978). The reports must include the location, species or animal description, and the circumstances of the event. Despite the Act covering deaths or injuries, the fisher logbook data collected by Fisheries New Zealand also contains reports of cetaceans caught alive and uninjured (Fisheries New Zealand, pers. comm. 13.05.2020). In addition to this mandatory reporting, the New Zealand Department of Conservation has a hotline to report marine mammals entangled in fishing gear and in distress (Department of Conservation, n.d.).

Sweden

Harbor porpoise (*Phocoena phocoena*) is listed as a protected species in Sweden (Naturvårdsverket, 2016). Any porpoises which are found dead or incidentally killed in Sweden belong to the state and must be reported as stated in Article 33 of the Swedish Hunting Ordinance (1987:905) (Sveriges Riksdag, 2020). The regulations are specifically in place for harbor porpoises only and do not include any other cetacean species that could be caught in Swedish waters. According to the Swedish Agency for Marine and Water Management (SwAM), it is not mandatory for fishers to report harbor porpoise bycatch in fishing logbooks (Havs-och vattenmyndighetens, 2018), but they are legally required to report this bycatch to the police or directly to the Swedish Museum of Natural History which is commissioned by SwAM to collect such reports (Naturhistoriska riksmuseet–Peter Mortensen, 2020). The reports must include the location (including coordinates), date, length of the animal, estimated weight of the animal (if possible), and optionally the depth of the fishing gear and the type of fishing gear (Naturhistoriska riksmuseet-Katarina Loso, 2020). Upon approval by the Swedish Museum of Natural History, compensation of 1000 Swedish krona (ca.100USD) is paid to anyone who collects and freezes a deceased harbor porpoise for their research (Naturhistoriska riksmuseet–Katarina Loso, 2019).

Finland

In Finland, Section 62 of the Finnish Fishing Act, first established in 1982, covers reporting of bycatch (Finlex, 2015). The legislation simply states that any bycatch of harbor porpoises must be immediately reported to the Finnish National Resources Centre. No other cetacean species are covered by this legislation, and it is not specified how the reports should be submitted. It is possible for bycatch to be recorded in logbooks and then the logbook information reported to the National Resources Centre, or it is possible to make an online report through the National Resource Centre webpage (Olli Loisa, pers. comm., 24.06.2020). The online report must include the name of the reporter or vessel, the date, time and location of the incident, the gender and age class of the animal (if known), and whether the animal was alive or deceased (Luonnonvarakeskus, 2020).

Norway

Norway has "Regulations on position reporting and electronic reporting for Norwegian fishing and catching vessels" which state under § 10 and § 12 that vessels with a length of 15 m or more fishing in Norwegian waters must electronically report all catch, including bycatch of marine mammals, to the Norwegian

Directorate of Fisheries (Nærings-og fiskeridepartementet, 2009). This mandatory reporting began in 2011 (Fiskeridirektoratet, pers. comm., 15.04.2021). The information required in the reports includes date, time, position, fishing zone, species, gear damage, number of animals, and weight. Most reports also include the gear type (though it is only mandatory to report when using a trawl) and mesh size (though it is only mandatory to report when using a trawl, Danish seine, or seine) (Nærings-og fiskeridepartementet, 2009). For recreational fishers and vessels below 15 m in length, they have the option to voluntarily use an app ("fritidsfiskeappen") to report bycatch to the Directorate of Fisheries, but this is currently not mandatory by law (Fiskeridirektoratet, pers. comm., 09.07.2020).

Iceland

Iceland established the Fisheries Management Act in 1990. Under Article 17 of this Act, it is stated that all catch must be recorded in special logbooks which are provided by and submitted to the Directorate of Fisheries (FAOLEX Database, 2006). This must include information about all cetaceans. This logbook reporting system became electronic in 2009 and under Article 3 of the "Regulation on registration and electronic submission of catch information" states that all marine mammal bycatch must be reported in the electronic logbook, including date, ship identification, fishing gear type, location, species and number of animals (Atvinnuvega-og nýsköpunarráðuneyti, 2020).

France

Mandatory reporting of cetacean bycatch is relatively new in France. In 2018, the country passed the "Decree of September 6, 2018 amending the Order of July 1, 2011 setting the list of marine mammals protected on national territory and the terms of their protection" which states that all marine mammal bycatch must be reported in fishing logbooks (electronic for vessels 12 m and larger and paper for smaller vessels) for the purpose of scientific research (Ministère de l'Agriculture et de l'Alimentation, 2018). This came into effect 1 January 2019. The logbook reports must include the date, species, number of animals, estimated weight of each animal, and if the animal was discarded in the sea (Tachoires et al., 2018).

South Korea

The Korean Ministry of Oceans and Fisheries has a "Notice on Conservation and Management of Whale Resources." This Notice was created in accordance with the Fisheries Act and Fisheries Resource Management Act for the "preservation and management of cetacean resources" in Korean waters [Ministry of Oceans and Fisheries Korea (Fisheries Resource Policy Division), 2018]. This notice states that any capture of a cetacean must be reported to the maritime police chief, regardless of if the animal was alive or died during the incident. If the bycatch is reported to the maritime police, the fishers can then legally sell the meat (Mills et al., 1997). Due to this, it is possible that "incidental" take in Korean waters may at times be intentional capture, which is then reported as bycatch in order to profit from the sales (Baker et al., 2006; Lukoschek et al., 2009). All Korean cetacean bycatch data is first confirmed by an inspector (South Korea Ministry of Oceans and Fisheries, pers. comm. 08.05.2020) and is then reported to the International Whaling Commission (IWC). The available data in the progress reports include year, location, species, number of animals, life status of the animals, and fishing gear involved (IWC, 2020).

Chile

In September 2012, Article 7 of the Chilean General Law for Fisheries and Aquaculture was amended to include law no. 20.625 "Law on Discards and Bycatch" (Subsecretaría de Pesca y Acuicultura, 2020). Under this law, fishers are required to report bycatch of all marine mammals in vessel logbooks, which are electronic for commercial fishing vessels over 15 m and paper for artisanal vessels. The logbooks are collected by the National Fisheries and Aquaculture Service. The law states that all marine mammals are released when possible, and all reports of interaction incidents must include details about the vessel, location, date, number of animals caught, species, and life status (dead or alive) (Subsecretaría de Pesca y Acuicultura, pers. comm., 10.11.2020).

Japan

Since 2001, Article 91, Paragraph 2 of Japan's Ministerial Ordinance on Fisheries Permits and Controls (in Ministry of Agriculture, Forestry and Fisheries Ordinance No. 5 of 1963) has included mandatory reporting of "baleen whale, etc." bycatch in fixed fishing nets (Institute for Cetacean Research, 2011; Fisheries Agency Research Management Department, 2020). The law covers seven species of baleen whale and three species of toothed whale designated by the IWC. A bycatch report should be submitted for all incidents, including releasing the animal alive and must include the date and location, species (including a photograph), type of set net fishery and permit number, and length, gender, evidence of lactation, and measurements of fetus (where applicable) (Institute for Cetacean Research, 2011). It is also required to take a DNA sample and send it for testing to the Institute of Cetacean Research if the animal will be used. Once these actions are completed, it is permitted to sell the whale meat or use it for personal consumption. The Japanese Fisheries Resources Conservation Law also includes an additional three species of baleen whale and one species of toothed whale that must be reported, for which possession and sale are prohibited (Institute for Cetacean Research, 2011).

MATERIALS AND METHODS

The search for countries which have cetacean bycatch/interaction reporting laws was based on the top 30 countries with the largest fishing industries (FAO, 2018). An internet search was used to determine the fisheries governing body in each country and search their fisheries legislature for the keywords "mammal," "bycatch," "reporting," and "log." For countries where the relevant legislature could not be found or was not clear due to language barriers, the governing body was contacted directly through email to ask for further information.

The governments and relative ministries in each of the countries with cetacean bycatch reporting legislation were contacted directly through email about this study to inquire about available data. For those countries where it was possible, data was requested from 2009 to 2019. Raw fisher-reported logbook data including year, species, number of animals, and gear type was provided directly from Fisheries New Zealand, National Oceanic and Atmospheric Administration (NOAA, United States), Fiskistofa (Directorate of Fisheries, Iceland) and Hafrannsóknastofnun (Marine and Freshwater Research Institute, Iceland), and Fiskeridirektoratet (Directorate of Fisheries, Norway). Fishing effort data per year was provided by Fisheries New Zealand, Fiskistofa and Hafrannsóknastofnun (Iceland), and Havforskningsinstituttet (Norway). For the United States, effort data was provided by Pacific Fisheries Information Network, Western Pacific Fisheries Information Network, Alaska Fisheries Information Network, Gulf States Marine Fisheries Commission, and Atlantic States Marine Fisheries Commission, which included data for each state and territory (including Puerto Rico and the US Virgin Islands), except for Alabama, where permission was not granted to release this data. Fisher-reported cetacean bycatch/interactions per unit effort ("catch") (CPUEs) per gear type were calculated based on kgs of catch (New Zealand, Norway, United States), number of trips (Iceland lumpsucker gillnets), and number of net-nights (number of nets × soak time) (Iceland cod and other gillnets). CPUE was also calculated for individual species per gear type category when data were sufficient. All fisher logbook CPUEs were calculated using all reports where gear type was specified, regardless of the reported life-status of the animal.

Cetacean bycatch data from observer programs in each of the case study countries were provided directly from the relevant government or research office (New Zealand, Iceland) or gathered from the NOAA official stock assessment reports (United States[1]) similar to work conducted by Read et al. (2006) on earlier data. Minimum estimates of annual number of cetaceans caught were calculated from the stock assessment reports only considering data coming directly from observer programs and excluding supplementary data that is available in some reports, such as from strandings. Norwegian "reference fleet" data, which is used to estimate cetacean bycatch in Norwegian waters, were not available for comparison for this study.

To compare the two methods of quantifying bycatch (logbook vs. observer), pairwise t-test or Wilcoxon test comparisons between fisher logbook CPUE and observer CPUE were conducted for each gear category and species, where data were sufficient, for New Zealand and Iceland. Since the United States observer bycatch/interaction estimates were available in the form of estimated average number of individuals caught per year, based on 5-year time blocks, this data was compared to the average number of cetaceans caught per year in the fisher logbook data, based on the same 5-year time blocks, also using pairwise t-tests or Wilcoxon tests. Where data were sufficient, fisher and observer data were also split into "early time period" and "late time period" categories and compared in the same manner separately. Additionally, t-tests were used to compare "early time period" vs. "late time period" fisher logbook CPUE for each gear category in order to investigate logbook reporting over time in each of the case study countries. All tests were performed using a set alpha of 0.05 in the statistical software R (R Core Team, 2019).

RESULTS

Reported cetacean bycatch/interactions by observers were on average 774% higher than fisher reported bycatch in trawls, 7348% higher in nets and 1725% higher in hook and lines. When broken down by individual countries, the average annual estimated cetacean CPUE based on observer data in New Zealand was 52% higher in trawl, 779% higher in passive netting, and 754% higher in lining compared to the CPUEs based on fisher logbook data (**Table 1**). In Iceland, the estimated CPUE based on inspector (observer) data was 329% higher in the lumpfish gillnet fishery and 26920% higher in the cod and others (cod+) gillnet fishery compared to CPUEs based on fisher logbook data. For the United States, the mean annual number of individuals estimated as bycatch/seriously injured based on observer data pooled into 5-year time blocks was on average 2696% higher in hook and lines, 1365% in nets, and 1495% in trawl when compared to the mean annual number of individuals in the same 5-year times blocks reported in fisher logbooks (**Table 1**). For Norway, CPUEs for the 15m+ vessel fishing fleet were calculated based on fisher logbooks for the first time, but there were no available data to compare this to the 15m+ vessel reference fleet. However, for seines ($n = 3$), trap ($n = 1$), and trawl ($n = 13$) gear categories, there were reports in the fisher logbooks that were not detected by the reference fleet (Norway Marine Research Institute, pers comm. 09.03.2021). Further results from statistical comparisons between fisher logbook data and observer data for each case study country are detailed below.

New Zealand

There were nine cetacean species reported as bycatch in the New Zealand exclusive economic zone between 2009 and 2019 in fisher logbooks (**Table 2**). There were also reports of unspecified dolphin/toothed whales, baleen whales, and beaked whales. New Zealand logbooks specify between the categories of "alive and uninjured," "alive and injured," and "deceased." When combining all reports from all years the percent of individuals reported in each category were 2.1, 17.5, and 80.4% respectively. Reports came from six different gears (trawl, passive netting, lining, other lining, potting, and seine). There were seven cetacean species reported in the observer data, six of which were also reported in the fisher data and one of which was not [Risso's dolphin (*Grampus griseus*)]. The observer data did not include bycatch records of minke whale (*Balaenoptera bonaerensis*), humpback whale (*Megaptera novaeangliae*) or fin whale (*Balaenoptera physalus*), which were reported in the fisher logbook data, though it did have records of unspecified baleen whales which may account for these species. The observer data consistently covered trawls (25.9–56.1% coverage per year), passive netting (0–10.2% coverage per year), and lining (1.8–11%

---

[1] https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-stock-assessment-reports-species-stock#cetaceans

coverage per year), and rarely covered the other gears (other lining, potting, and seine). Despite observer coverage in lining, trawl and passive netting, as well as a low coverage in "other lining" and potting, there was no observed cetacean bycatch in any gear type in 2019.

Cetacean catch per unit effort (CPUE) was calculated for trawl, passive netting and lining gear types, as well as for three dolphin species for which the most data were available (common dolphins (*Delphinus delphis*) in trawl gear, Hector's dolphins (*Cephalorhynchus hectori*) in passive netting gear, and dusky dolphins (*Lagenorhynchus obscurus*) in passive netting gear). The trawl category had the highest total number of individuals reported in fisher logbook data ($n = 202$). The fisher CPUE for trawl fisheries was significantly less than the observer CPUE for all years combined (2009–2019) ($p = 0.028$, $t = -2.16$) (**Table 3**). Data were also sufficient to split trawl data into an early time period (2009–2014) and a late time period (2015–2019). The fisher logbook trawl CPUE was significantly lower than the observer trawl CPUE for the earlier time period ($p = 0.004$, $t = -4.16$); however, there was no difference between the CPUEs for the later time period ($p = 0.68$, $t = 0.50$). When comparing the fisher logbook trawl CPUE for the early time period to the late time period, there was no significant difference between them ($p = 0.65$, $W = 18$) (**Table 4**). Comparison of the CPUEs for passive netting yielded similar results, with fisher logbook CPUE being significantly lower than observer CPUE for all years combined (2009–2019, excluding 2011 when there was no observer coverage) ($p = 0.0071$, $t = -3.03$), fisher logbook CPUE being significantly lower than observer CPUE in the earlier time period (2009–2014, excluding 2011) ($p = 0.003$, $t = -5.34$), and there being no significant difference between the CPUEs in the later time period (2015–2019) ($p = 0.22$, $t = -0.86$) (**Table 3**). When comparing the fisher logbook CPUE for the earlier time period to the later time period, there was no significant difference between them ($p = 0.72$, $t = -0.38$) (**Table 4**).

The CPUEs for lining were not significantly different for all years combined (2009–2019, excluding 2010 and 2013 when there was no observer coverage) ($p = 0.43$, $V = 15$) (**Table 3**). It was not possible to compare the CPUEs for the early time period due to little observer data; however, CPUEs could be compared for the late time period (2015–2019). For the late time period, fisher logbook CPUE was significantly lower than observer CPUE ($p = 0.021$, $t = -2.95$). When comparing the fisher logbook CPUE for the early time period to the late time period, there was no significant difference between them ($p = 0.18$, $t = -1.47$) (**Table 4**).

The fisher logbook CPUE for common dolphins in trawl gear was significantly lower than the observer CPUE for all years combined (2009–2019) ($p = 0.011$, $t = -2.70$) (**Table 5**). The fisher logbook CPUE was also significantly lower in the early time period (2009–2014) ($p = 0.002$, $t = -5.05$), but was not significantly lower in the later time period (2015–2019) ($p = 0.49$, $t = -0.032$). The fisher logbook CPUE for Hector's dolphins caught in passive net gear was also significantly lower than the observer CPUE for all years combined (2009–2019, excluding 2011 when there was no observer coverage) ($p = 0.018$, $t = -2.46$) and significantly lower in the earlier time period (2009–2014, excluding 2011) ($p = 0.012$, $t = 3.59$) (**Table 5**).

TABLE 1 | Average annual number of individual cetaceans reported as bycatch per year, average annual catch per unit effort (CPUE) based on 100 kg fish caught (New Zealand, Norway, United States), number of trips (Iceland lumpfish gillnet) or net-nights (Iceland cod+ gillnet), coefficient of variance (CV), and 95% confident intervals (CIs) calculated based on fisher logbook data and observer data CPUEs for each country and each gear type for which the average annual number of cetaceans caught was at least 1 animal.

Country	Gear-type	Group	Average number individuals reported per year	Average CPUE per year	Percent increase (%)	CV	CIs
New Zealand	Passive Netting	Fisher	5.1	1.04E-04		0.67	6.4E-05 – 1.5E-04
		Observer	1.7	9.14E-04	+779	0.87	4.2E-04 – 1.4E-03
	Trawl	Fisher	18.4	5.41E-06		0.54	3.7E-06 – 7.1E-06
		Observer	11.7	8.22E-06	+52	0.77	4.5E-06 – 1.2E-05
	Lining	Fisher	2.1	1.37E-05		1.00	4.6E-06 – 1.8E-05
		Observer	1.1	1.17E-04	+754	1.31	1.7E-05 – 2.2E-04
Iceland	Lumpfish gillnet	Fisher	107.8	2.96E-02		0.92	7.8E-03 – 5.2E-02
		Observer	7.7	1.27E-01	+329	0.51	7.5E-02 – 1.8E-01
	Cod+ gillnet	Fisher	54.0	2.92E-04		0.26	2.2E-04 – 3.7E-04
		Observer	30.3	7.89E-02	+26920	0.22	6.2E-02 – 9.6E-02
Norway	Gillnet	Fisher	1.1	2.87E-06		1.22	5.8E-07 – 5.2E-06
	Trawl	Fisher	1.4	1.74E-07		1.28	2.9E-08 – 3.2E-07
United States*	Trawl	Fisher	23.0	2.60E-06		0.31	16.8–29.3
		Observer	368.2	X	+1495	0.49	209.5–526.6
	Net	Fisher	26.7	2.78E-06		0.25	20.9–32.5
		Observer	391.5	X	+1365	0.54	205.3 – 577.9
	Hook and line	Fisher	6.1	6.76E-06		0.09	5.6 – 6.6
		Observer	170.0	X	+2696	0.47	99.3 – 240.8

**United States CV and CIs are based on average number of individuals reported based on 5-year time blocks, not CPUE, due to the observer data being available as raised number of individuals caught per the same time blocks.*

TABLE 2 | Cetacean species included in cetacean bycatch/interaction reports from each country, with indication of if they were reported in both fisher logbook and observer data (F + O), fisher logbook data only (F), or observer data only (O).

Country	Common name	Scientific name	Reported in
New Zealand	Bottlenose dolphin	*Tursiops truncatus*	F + O
	Common dolphin	*Delphinus delphis*	F + O
	Dusky dolphin	*Lagenorhynchus obscurus*	F + O
	Hector's dolphin	*Cephalorhynchus hectori*	F + O
	Killer whale	*Orcinus orca*	F + O
	Pilot whale	*Globicephala* spp.	F + O
	Fin whale	*Balaenoptera physalus*	F
	Humpback whale	*Megaptera novaeangliae*	F
	Minke whale	*Balaenoptera acutorostrata*	F
	Risso's dolphin	*Grampus griseus*	O
Iceland	Harbor porpoise	*Phocoena phocoena*	F + O
	White-beaked dolphin	*Lagenorhynchus albirostris*	F + O
	Common dolphin	*Delphinus delphis*	F
	Risso's dolphin	*Grampus griseus*	F
	Cuvier's beaked whale	*Ziphius cavirostris*	F
	Northern bottlenose whale	*Hyperoodon ampullatus*	F
	Fin whale	*Balaenoptera physalus*	F
	Humpback whale	*Megaptera novaeangliae*	F
Norway	Harbor porpoise	*Phocoena phocoena*	F + O
	Minke whale	*Balaenoptera acutorostrata*	F + O
	Bottlenose dolphin	*Tursiops truncatus*	F
	Beluga whale	*Delphinapterus leucas*	F
	Killer whale	*Orcinus orca*	F
	Blue whale	*Balaenoptera musculus*	F
United States	Harbor porpoise	*Phocoena phocoena*	F + O
	Atlantic white-sided dolphin	*Lagenorhynchus acutus*	F + O
	Bottlenose dolphin	*Tursiops truncatus*	F + O
	Common dolphin	*Delphinus delphis*	F + O
	Dall's porpoise	*Phocoenoides dalli*	F + O
	Northern right whale dolphin	*Lissodelphis borealis*	F + O
	Pacific white sided dolphin	*Lagenorhynchus obliquidens*	F + O
	Risso's dolphin	*Grampus griseus*	F + O
	Rough-toothed dolphin	*Steno bredanensis*	F + O
	Beluga whale	*Delphinapterus leucas*	F + O
	Dwarf sperm whale	*Kogia sima*	F + O
	False killer whale	*Pseudorca crassidens*	F + O
	Killer whale	*Orcinus orca*	F + O
	Pilot whale	*Globicephala spp.*	F + O
	Sperm whale	*Physeter macrocephalus*	F + O
	Gray whale	*Eschrichtius robustus*	F + O
	Humpback whale	*Megaptera novaeangliae*	F + O
	Fin whale	*Balaenoptera physalus*	F + O
	Minke whale	*Balaenoptera acutorostrata*	F + O
	Spotted dolphin	*Stenella* spp.	F
	Bryde's whale	*Balaenoptera brydei*	F
	Pygmy sperm whale	*Kogia breviceps*	O

TABLE 3 | *p*-value results of paired *t*-test/Wilcoxon tests determining if fisher reported CPUE was significantly lower than observer reported CPUE (New Zealand, Iceland) and if fisher reported annual average number of individuals caught based on 5-year time blocks significantly differed from observer reported annual average number of individuals caught based on the same 5-year time blocks (United States) for each gear category where enough data were available.

Country	Gear category	Time period	Years	*p*-value
New Zealand	Trawl	All	2009–2019	**0.028**
		Early	2009–2014	**0.004**
		Late	2015–2019	0.68
	Passive netting	All	2009–2019 (ex. 2011)	**0.007**
		Early	2009–2014 (ex. 2001)	**0.003**
		Late	2015–2019	0.22
	Lining	All	2009–2019 (ex. 2010, 2013)	0.43
		Early	X	X
		Late	2015–2019	**0.021**
Iceland	Lumpfish gillnet	All	2014–2019	**0.003**
		Early	2014–2016	**0.006**
		Late	2017–2019	0.1
	Cod+ gillnet	All	2016–2019	**0.001**
		Early	X	X
		Late	X	X
United States	Trawl	All time blocks	*	**0.006**
	Net	All time blocks	*	**0.009**
	Hook and line	All time blocks	*	**0.005**

Significant p-values are in bold.

**United States data was based on average number of individuals caught for the 5-year time blocks 2009–2013, 2010–2014, 2011–2015, 2012–2016, and 2013–2017.*

There was not enough observer reported bycatch of Hector's dolphins to compare the late time period. Oppositely, there was no significant difference in the CPUEs for dusky dolphins caught in passive netting for all years combined (2009–2019, excluding 2011 when there was no observer coverage) ($p = 0.28$, $V = 10$) and no significant difference between the CPUEs for the early time period (2009–2014, excluding 2011) ($p = 0.10$, $t = -1.54$) (**Table 5**). There was not enough observer reported bycatch of dusky dolphins in passive net gear to compare the late time period.

Iceland

There were eight cetacean species reported as bycatch in Icelandic fisher logbooks in the years where reporting could be considered complete (2014–2019) (**Table 2**). There were also reports of an unspecified dolphin and unspecified medium cetacean. Reports came from three different fishing gear categories (trawl, passive netting, and hook and line). There were 984 individual cetaceans reported as bycatch in all gear combined between 2009 and 2019; however, 647 of these individuals could be included in this study from the lumpfish gillnet fishery (2014–2019) and 216 of these individuals could be included in this study from the "cod

TABLE 4 | Average CPUE and *p*-value results of *t*-tests or Wilcoxon tests comparing fisher logbook CPUE over time for all gear types with sufficient data.

Country	Gear-type	Group	Years	Average CPUE	*p*-value
New Zealand	Passive Netting	Early	2009–2014	9.3487E-05	
		Late	2015–2019	0.000111538	0.72
	Trawl	Early	2009–2014	6.11667E-06	
		Late	2015–2019	0.000004568	0.65
	Lining	Early	2009–2014	6.92333E-06	
		Late	2015–2019	1.63536E-05	0.18
Iceland	Lumpfish gillnet	Early	2014–2016	0.011307951	
		Late	2017–2019	0.047988214	0.14
Norway	Gillnet	Early	2011–2015	9.71898E-07	
		Late	2016–2019	5.23959E-06	0.08
	Trawl	Early	2011–2015	9.75392E-08	
		Late	2016–2019	2.70052E-07	0.9
United States	Trawl	Early	2009–2014	3.1251E-06	
		Late	2015–2019	1.47519E-06	0.11
	Net	Early	2009–2014	2.53633E-06	
		Late	2015–2019	3.06044E-06	0.59
	Hook and line	Early	2009–2014	6.41707E-06	
		Late	2015–2019	9.2772E-06	0.17

and others” (cod+) gillnet fishery (2016–2019). There were only two cetacean species reported in the inspector (observer) data [harbor porpoise (*Phocoena phocoena*) and white-beaked dolphin (*Lagenorhynchus albirostris*)]. Cetaceans were only observed by inspectors in two gillnet categories: lumpfish gillnetting (0.74–2.82% coverage per year), and cod and other (cod+) gillnetting (0.15–0.25% coverage per year). There is also some inspector coverage on bottom trawls and long-lines (ICES, 2020); however, there has never been cetacean bycatch reported (Iceland Marine and Freshwater Research Institute, unpub. data). Cetacean catch per unit effort was calculated separately for each of the two gillnet fisheries that had inspector coverage (lumpfish and cod+). Inspectors started reporting marine mammal bycatch in the lumpfish fishery in 2014 and the cod+ fishery in 2016, therefore only data from these years onward could be used in CPUE comparisons. The fisher logbook CPUE for the lumpfish gillnet fishery was significantly less than the inspector reported CPUE for all years combined (2014–2019) ($p = 0.003$, $t = -4.67$) (**Table 3**). Data was also sufficient to split the lumpfish gillnet data into an early time period (2014–2016) and a late time period (2017–2019). The fisher logbook CPUE was significantly lower than the inspector CPUE for the early time period ($p = 0.006$, $t = -9.27$); however, there was no significant difference between the CPUEs for the late time period ($p = 0.10$, $t = -1.94$). When comparing the fisher logbook CPUE for the early time period to the late time period, there was no significant difference between them ($p = 0.14$, $t = -2.17$) (**Table 4**).

The fisher logbook CPUE was also significantly lower than the inspector reported CPUE for the cod+ gillnet fishery for all years combined (2016–2019) ($p = 0.001$, $t = -9.29$) (**Table 3**). There were not enough years of data to compare an early and late time period of the fisher and observer CPUEs or to compare the fisher logbook CPUE over time.

TABLE 5 | *p*-value results of paired *t*-test/Wilcoxon tests determining if fisher reported CPUE is significantly lower than observer reported CPUE (New Zealand) and if fisher reported annual average number of individuals caught based on 5-year time blocks is significantly lower than observer reported annual average number of individuals caught based on the same 5-year time blocks (United States) for the most commonly reported species in specific gear categories.

Country	Gear category	Species	Time period	Years	*p*-value
New Zealand	Trawl	Common dolphin (*Delphinus delphis*)	All	2009–2019	**0.011**
			Early	2009–2014	**0.002**
			Late	2015–2019	0.49
	Passive netting	Hector’s dolphin (*Cephalorhynchus hectori*)	All	2009–2019 (ex. 2011)	**0.018**
			Early	2009–2014 (ex. 2011)	**0.012**
			Late	X	X
	Passive netting	Dusky dolphin (*Lagenorhynchus obscurus*)	All	2009–2019 (ex. 2011)	0.28
			Early	2009–2014 (ex. 2011)	0.1
			Late	X	X
United States	Net	Bottlenose dolphin (*Tursiops truncatus*)	All (ex. 2012–2016 mean)	*	**0.04**
		Common dolphin (*Delphinus delphis*)	All (ex. 2012–2016 mean)	*	**0.0004**
		Harbor porpoise (*Phocoena phocoena*)	All (ex. 2011–2015 mean)	*	**0.008**
		Humpback whale (*Megaptera novaeangliae*)	All	*	0.00002**
		Pilot whale (*Globicephala* spp.)	All (ex. 2011–2015 mean)	*	0.32
	Trawl	Atlantic white-sided dolphin (*Lagenorhynchus acutus*)	All	*	**0.02**
		Bottlenose dolphin (*Tursiops truncatus*)	All (ex. 2011–2015 and 2012–2016 mean)	*	**0.002**
		Common dolphin (*Delphinus delphis*)	All (ex. 2012–2016 mean)	*	**0.00007**
	Hook & Line	False killer whale (*Pseudorca crassidens*)	All	*	**0.009**
		Pilot whale (*Globicephala* spp.)	All (ex. 2011–2015 mean)	*	**0.0002**

Significant p-values are in bold.

**United States data was based on average number of individuals caught for the 5-year time blocks 2009–2013, 2010–2014, 2011–2015, 2012–2016, and 2013–2017.*

***p-value showing mean annual humpback whale bycatch/interaction was significantly higher in the fisher logbook data compared to the observer data.*

Norway

There were six cetacean species reported by Norwegian fishing vessels 15 m or greater (15m+) between 2011 and 2019 (**Table 2**). The reports came from five different gear types (Danish seine, purse seine, trawl, trap, and gillnet). There were only two species reported as bycatch by the 15m+ reference fleet (harbor porpoise and minke whale) and, additionally, reports simply labeled as "dolphin." The reference fleet reports did not include orca (*Orcinus orca*), beluga (*Delphinapterus leucas*), blue whale (*Balaenoptera musculus*), or bottlenose dolphin (*Tursiops truncatus*) which were included in the fisher logbook data. Reports from the reference fleet were from two different gear categories (hook and line and gillnet). There was no available data on the effort of the 15m+ reference fleet, therefore it was not possible to statistically compare the fisher logbook CPUEs to the reference fleet CPUEs for these gear categories. However, it could be noted that there was cetacean bycatch reported in the fisher logbook data in the seine, trawl, and trap gear categories that were not detected by the reference fleet, though the fleet is covering these gear categories (Norway Marine Research Institute, pers. comm.). Fisher logbook gillnet and trawl CPUEs could be compared between an early time period (2011–2015) and a late time period (2016–2019) for each gear separately. Both the gillnet and trawl CPUEs did not differ significantly between the two time periods ($p = 0.08$, $W = 3$; $p = 0.90$, $W = 9$ respectively) (**Table 4**).

United States

There were 21 different cetacean species reported as bycatch/seriously injured in fisher logbooks in the United States between 2009 and 2017, matching the dates for which observer reports were available (**Table 2**). There were also reports of unidentified baleen whales, small cetaceans (porpoise or dolphin), toothed whales and beaked whales. The logbook reports came from four broad gear categories: trawl, pot and trap, hook and line, and net. The net category included all set nets and seine gear. There were 21 species reported as bycatch between 2009 and 2017 in observer data reported in the NOAA Marine Mammal Stock Assessment Reports[2]. However, in the fisher logbook data, pilot whales were not split into the two known species (*Globicephala melas* and *Globicephala macrorhynchus*) so these species were also combined in the observer data (taking the total down to 20). Fisher logbook data included spinner dolphins (*Stenella longirostris*) and Bryde's whales (*Balaenoptera edeni*) which were not in the observer reports. The observer reports included pygmy sperm whales (*Kogia breviceps*) which were not included in the fisher reports; however, the unidentified toothed whales in the fisher reports may account for this species.

United States observer bycatch data is reported as the mean annual number of individuals of each species caught based on the most recent 5 years of data. For comparison, the same means were calculated using the fisher logbook data for the same five 5-year time blocks (2009–2013, 2010–2014, 2011–2015, 2012–2016, and 2013–2017) for net, trawl, and hook and line gears. The mean number of cetaceans reported as bycatch annually in the fisher logbook data was significantly lower than the observer estimates for the net, trawl and hook and line gear types ($p = 0.009$, $t = -3.9$; $p = 0.006$, $t = -4.4$; $p = 0.005$, $t = -4.6$ respectively) (**Table 3**).

CPUEs were calculated for the fisher logbook data and used to compare reporting over time. Data was sufficient to compare an early time period (2009–2014) with a late time period (2015–2019) for the net, trawl, and hook and line categories. There was no significant difference between the two time periods for any of the categories ($p = 0.59$, $t = -0.55$; $p = 0.11$, $t = 1.93$; $p = 0.17$, $W = 7$ respectively) (**Table 4**).

Comparisons were also made between the mean annual number of individuals of the most commonly reported species in the fisher logbook data (species reported in five or more years between 2009 and 2017), excluding means from time blocks where the observer report was not available. These were bottlenose dolphin (excluding 2012–2016 mean), common dolphin (excluding 2012–2016 mean), harbor porpoise (excluding 2011–2015 mean), humpback whale, and pilot whale spp. (excluding 2011–2015 mean) in the net category; Atlantic white-sided dolphin (*Lagenorhynchus acutus*), bottlenose dolphin (excluding 2011–2015 and 2012–2016 means) and common dolphin (excluding 2012–2016 mean) in the trawl category; and false killer whales (*Pseudorca crassidens*) and pilot whale spp. (excluding 2011–2015 mean) in the hook and line category.

In the net category, harbor porpoise, common dolphin, and bottlenose dolphin mean annual numbers of individuals reported as bycatch/serious injury were significantly lower in the fisher logbook data when compared to the estimated annual means from the observer data ($p = 0.008$, $t = -4.9$; $p = 0.0004$; $t = -14.0$; $p = 0.04$, $t = -2.6$ respectively) (**Table 5**). There was no significant difference between the mean annual number of pilot whale spp. calculated from fisher logbook data and the mean annual number calculated from observer data ($p = 0.32$, $t = 1.17$). The mean annual number of humpback whales calculated from fisher logbook data was significantly greater than the mean annual number estimated from observer data ($p = 0.00002$, $t = 20.7$). In the trawl category, the mean annual numbers of individuals reported as bycatch/serious injury in the fisher logbook data were significantly less than the estimated annual means from the observer data for all three species (Atlantic white-sided dolphin: $p = 0.02$, $t = -2.9$; bottlenose dolphin: $p = 0.002$, $t = -16.4$; common dolphin: $p = 0.00007$, $t = -24.6$) (**Table 5**). In the hook and line category, the mean annual numbers of individuals reported as bycatch/serious injury in the fisher logbook data were significantly less than the estimated annual means from the observer data for both species (false killer whale: $p = 0.009$, $t = -7.3$; pilot whale spp.: $p = 0.0002$, $t = -17.8$) (**Table 5**).

DISCUSSION

Out of the 30 countries with the largest fishing industries in the world (FAO, 2018), 12 were determined to have some form of cetacean bycatch/interaction reporting legislation. The legislation

---

[2] https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-stock-assessment-reports-species-stock

ranged from very minimal (only harbor porpoises covered by law in Sweden and Finland and very little reporting) to well established (clear legislation and logbook reporting including all species e.g., in United States and New Zealand). Of these 12, ten of the countries were identified as also having some level of onboard observer coverage which they use to calculate cetacean bycatch/interaction CPUE, which varies by percent coverage and fisheries monitored (e.g., Hanrahan and Pelrine, 1997; Okuda et al., 2017; Muñoz et al., 2018; ICES, 2020; Ministry for Primary Industries New Zealand, 2020; National Marine Fisheries Service, 2020; Australian Fisheries Management Authority, 2021). Finland does not have any form of observer program (ICES, 2020) and no information could be found for South Korea, other than that all landed bycatch is verified by an inspector (South Korea Ministry of Oceans and Fisheries, pers. comm., 08.05.2020).

This study provides a broad, quantitative overview of the global issue of under-reporting of cetacean bycatch/interactions in fisher logbooks compared to observer programs using case studies of entire countries. Likely due to the difficult nature of accessing and/or compiling datasets for different countries, under-reporting of cetacean bycatch/interactions has not been investigated in this manner previously. When looking generally at the differences between fisher logbook and observer data from New Zealand, United States, and Iceland, the average bycatch of cetaceans per year (dead or alive) was underestimated by anywhere between 52 and 26920% for the three major gear types which could be examined (trawl, net, and hook and line). The differences were significant when combining data for all available years for all gear types that could be examined in each country, except for the lining category in New Zealand, despite an over 750% increase in CPUE calculated from observer data. Low observer coverage and zeros in the data may have affected the results in this case, particularly since fisher logbook CPUE was significantly lower than observer CPUE in the late time period comparison. Under-reporting of harbor porpoise bycatch in the gillnet fishery in Iceland was quantified previously through a questionnaire, which determined logbook data underestimated bycatch by a ratio of approximately 1:26 (Ólafsdóttir, 2010). Quantified cetacean under-reporting in logbooks has not been published for fisheries in the other case study countries. In addition, CPUE of cetaceans in Norway's 15m+ fishing fleet, though small, was quantified using logbook data for the first time. This determined that there is some amount of cetacean bycatch in Norway going unaccounted for that is likely underestimated given the trend in the other countries with well-developed fishing industries.

For cases where it was possible to split data into both early and late time periods for separate comparisons (trawl and passive netting in New Zealand, lumpfish gillnet in Iceland), in all cases the fisher logbook CPUEs were significantly lower than the observer CPUEs for the early time period but there were no significant differences between them in the late time period. We considered if this could be an indication that fisher reporting was increasing over time in these fisheries; however, when we compared the fisher logbook CPUEs from the early time period to the late time period there was no significant differences and therefore no evidence for this. The most likely explanation is that small sample sizes and zeros in the data from late time periods affected the results. For example, observer CPUE in 2019 in New Zealand was zero for all gear types and Fisheries New Zealand could not provide an explanation for this. The logbook data from fishers could be a very valuable tool for gaining insight into cetacean bycatch and interaction with fishing gear; however, given the overall stark differences in reporting in logbooks versus estimations of bycatch calculated from observer data, there is vast room for improvement in logbook reporting if it is to still be considered a useful practice. The fact that the three countries for which comparisons could be made between cetacean bycatch/interaction data reported in fisher logbooks and bycatch estimates calculated from observer data differ in geographic location, cetacean species and size of exclusive economic zone [New Zealand: 1.2 million square nautical miles (Sea Around US, 2016), US: 3.4 million square nautical miles (NOAA, 2011), Iceland: 0.22 million square nautical miles (FAO, 2011)], overall the analysis of each country showed significant under-reporting, suggesting the case is likely similar in many other countries with well-developed commercial fishing industries, particularly using trawl, net, and hook and line gears. There was, however, variation in the CPUEs per gear type between each of the case study countries. This variation may be due to differences in number of cetacean species and overall density of cetaceans, the vulnerability of different cetacean species to bycatch and/or differences in observer coverage. Due in part to the different metrics of effort used to calculate CPUEs in each country, and the need to use average number of animals caught in the United States case study, the case study countries could not be directly compared to each other in the study to investigate the differences further.

One of the most likely causes of under-reporting of cetacean bycatch, even in mandatory logbooks, is the concerns fishers have of punishment or negative consequences to the fishing industry. For example, in Atlantic Canada, one-quarter of target participants for interviews discussing long-line bycatch refused the interview based on concerns of consequences and general distrust of the researchers (Carruthers and Neis, 2011). Similarly, in Iceland, nearly half of questionnaire respondents refused to answer a question about why they would not report cetacean bycatch or responded they were concerned about the potential negative consequences (Basran and Rasmussen, 2021). Particularly when endangered or critically endangered species are involved in bycatch incidents (such as the North Atlantic right whale), changes to or closures of fisheries can occur due to the serious implications to the stock if even one individual is removed (e.g., Merrick et al., 2001). Significant under-reporting in logbooks was also demonstrated for several different individual species in separate gear types in both New Zealand and the United States. These concerns over consequences, paired with the fact that filling out bycatch reports is extra work, usually with no reward, gives fishers very little incentive to report. Though, in the aforementioned 12 countries, not reporting cetacean bycatch is a punishable offense by law, violations are virtually impossible to track without independent observers and inspections. Differing legislature and different interpretations of the laws by fishers may also influence under-reporting of cetacean bycatch/interactions. For example, in Iceland, the wording of the law lacks clarity on the matter of what life status of the animals need to be reported. Results from a questionnaire targeting Icelandic fishers revealed that those reporting cetaceans in their logbooks are only doing so if the animal is dead (Basran and Rasmussen, 2021), though fisheries scientists believe all catches, even if released alive, should be reported (Iceland Marine and Freshwater Research Institute. pers. comm. 03.12.2020). A similar issue arises in Norway, where the logbooks are designed to report landed catch, including bycatch of cetaceans, but it is not required for fishers to land cetacean bycatch (Norway Directorate of Fisheries, pers. comm. 14.07.2020). This suggests that even though the law states fishers should report all cetacean bycatch, not only will they not report injured animals, but they are also unlikely to report all deceased animals if they did not land them.

Though bycatch/interaction under-reporting is likely an issue for virtually all cetacean species, it has been noted that it is particularly an issue for large whale bycatch/interactions given their rare and difficult-to-observer nature (IWC, 2011). Most of the reports from all four case study countries involved dolphins and porpoises. Given that these species are the most likely to drown when they are caught in fishing gear, it can be suspected that fishers may be the most inclined to report these events. Both small whale and, particularly, large whale interactions with gear are less likely to be witnessed given that these species may be able to break away from entangling gear, meaning many incidents will go unreported (Robbins and Mattila, 2001; IWC, 2011). This is particularly true for pot/trap and gillnet gears which are left in the water, unattended, for longer periods of time and are well known for entangling whales (Johnson et al., 2005). For example, in Iceland, a study based on scarring estimated that a minimum of 25% of humpback whales have been entangled previously (Basran et al., 2019) and additionally 15% of questionnaire respondents witnessed humpback whales interact with their fishing gear (Basran and Rasmussen, 2021). Furthermore, there have been reports of humpback whale deaths due to entanglement in interviews with fishing vessel captains (Basran and Rasmussen, 2021), and based on examination of stranded animals (Víkingsson et al., 2004, 2005; Víkingsson, 2011). Despite this evidence, there was only one humpback whale reported as bycatch in the Icelandic fisher logbooks between 2009 and 2019 (Iceland Marine Research Institute, unpub. data). Similarly in the Gulf of Maine, United States, it was estimated based on scarring that a minimum of 50% of humpback whales (Robbins and Mattila, 2004; Robbins, 2009) and 83% of North Atlantic right whales have been entangled previously (Knowlton et al., 2012); however, there were only 32 humpback whales (averaging 2.9 individuals reported per year) and no North Atlantic right whales reported in the logbooks (2009–2019) despite the law being clear about reporting all interactions leading to injury or death, and the injury category including a sub-code "released trailing gear." Though many of the incidents may have gone unwitnessed, it is likely a number of them were witnessed but unreported, as demonstrated by the questionnaires and interviews previously conducted in Iceland.

Due to the unreliability of logbook reporting, observer programs are needed to estimate cetacean bycatch/interaction with fishing gear more accurately; however, there are barriers to observer programs being widely used in all fishing nations. Firstly, observer programs can be very costly, with the latest report from the United States stating it costed 79.5 million USD for observer coverage in 54 fisheries (National Marine Fisheries Service, 2020) and therefore it is unlikely that they will be implemented in all fisheries globally that are high-risk of catching cetaceans. Additionally, sufficient observer effort must be used in order to produce accurate estimates. A computer simulation, assuming unbiased observer programs and requiring 90% of the simulated observer samples to estimate bycatch within 10% of the actual number, estimated that coverage must be at least 20% to accurately estimate bycatch of common species and 50% to accurately estimate bycatch of rarer species (Babcock et al., 2003). The observer coverage in New Zealand was an average of 44% per year for trawl gear, which should be sufficient for accurate estimates; however, averaged 5% for both passive netting and lining gears, suggesting the observer estimates could be under-representing the total catch of cetaceans, particularly for cases involving large whales or rare species (Read et al., 2006), given the low coverage. This could be of particular concern for the endangered Hector's dolphin which was most reported in passive netting gear. Observer coverage in Iceland only averaged 2% for the lumpfish gillnet fishery and 0.2% for the cod+ gillnet fishery meaning that the observer coverage is likely not enough to accurately estimate bycatch of cetaceans. Given the vast expanse, complicated management, and diversity of the United States fishing industries, estimating the overall observer coverage for each gear category was not possible in this study. Individual fisheries have anywhere from zero observer coverage (such as several gillnet and seine fisheries that are classified as low incidence of cetacean mortality ("Category III"[3]) to 100% observer coverage [such as the shallow-set longline fishery in Hawaii (NOAA, 2018)]. In Norway, a "reference fleet" is used for bycatch estimation, where vessel operators are paid a small fee for accurate reporting (Clegg and Williams, 2020). In addition, the Norwegian Directorate of Fisheries maritime service has onboard inspectors which assess bycatch among other things (Fiskeridirektoratet, 2017). Cetacean bycatch has not been investigated or publicly reported on for the Norway large vessel/high seas reference fleet (for which it is mandatory for all vessels to report cetacean bycatch), and neither these data, nor data from maritime service inspectors, were available for this study. There were 16 vessels in the large vessel reference fleet in 2019 using bottom gillnet, longline, demersal seine, purse seine, bottom trawl and shrimp trawl gear types (Clegg and Williams, 2020).

It is important to consider that differences in the training and duties of observers may affect the resulting cetacean bycatch estimates. In New Zealand, the United States, and Iceland, the main observer programs collect data on fishing activity,

---

[3] https://www.fisheries.noaa.gov/national/marine-mammal-protection/list-fisheries-summary-tables#table-1-category-iii

fish catch (including biological measurements and sample collection), fish discards, and marine mammal, seabird, and reptile bycatch (Fiskistofa, 2010; Christensen-Dalsgaard et al., 2019; Ministry for Primary Industries New Zealand, 2020; NOAA Fisheries, 2021b). Additionally, New Zealand's Department of Conservation and Alaska's Marine Mammal Observer Program collect dedicated protected species or marine mammal data through observers (Department of Conservation New Zealand Government, 2020a,b; NOAA Fisheries, 2021a). It has been previously determined that observers that have other duties in addition to recording marine mammal bycatch, under-report bycatch (therefore the bycatch estimates are negatively biased) (Bravington and Bisack, 1996). Randomization of observer coverage is also important to consider when estimating bycatch. In the case of Iceland, it is known that the observer coverage is often not random but based on vessels who have had unusual data or low compliance with fishing laws (Christensen-Dalsgaard et al., 2019). Additionally, using landed catch as the metric of effort, as is done in the United States, relies on accurate reporting of the catch, and if landed catch is under-estimated, marine mammal bycatch estimates could be positively biased (Bravington and Bisack, 1996).

RECOMMENDATIONS AND CONCLUSION

Though, in theory, bycatch reporting in mandatory fisher logbooks could be a cost-effective, scientifically valuable way to monitor cetacean bycatch, results from this study showed significant under-reporting and support that logbooks are not reliable. In all four case study countries, fisher logbooks did contain more cetacean species than observer data, which may be an indicator that fishers have more information, particularly about rare events, than observer programs can detect without very high coverage. However, fishers may have difficulty accurately identifying species (e.g., Stoller, 2020) and there is currently no widely used system in place, such as electronic monitoring, to verify the reports in fisher logbook data. Given this, it should be carefully considered if the use of such logbooks should be continued or implemented in the future. If logbook use is to continue, it is recommended that clarification of reporting laws would be a first step to improving logbook reporting. Secondly, countries could consider introducing simple reporting of cetacean bycatch/interactions using a mobile phone app where fishers could pre-fill their vessel and fishing gear information and, in the case of cetacean bycatch/interaction, open the app and take a picture or video of the event as supplementary material for the report. A reporting app is likely to be more successful on larger commercial vessels with several crew members, of which one could have time to record the incident, opposed to small vessels with one or few crew members. Though this does not address fishers' concerns over repercussion for reporting, it may increase reporting from those who find their current system to be a hassle.

Another way to estimate cetacean bycatch is by using a "reference fleet," where the vessels are compensated for reporting. This is the strategy used in Norway, and fisheries scientists there believe their reference fleet program yields them accurate bycatch data (Norway Marine Research Institute, pers. comm., 11.12.2019). However, biases in data from a select set of vessels should be considered before choosing to use a reference fleet opposed to other monitoring methods.

We recommended that the most viable option to collect cetacean bycatch data is to equip more fishing vessels with electronic monitoring cameras, as has already been tested in the United States and New Zealand, as well as other countries with bycatch reporting legislation (Australia, Canada, and Sweden), which could lower the cost and improve the coverage of bycatch observation compared to observer programs (Course et al., 2020; van Helmond et al., 2020), as well as improve the accuracy of bycatch estimation compared to fisher logbooks. In Denmark, it was determined that, compared to their observer programs, the use of electronic monitoring was 6.7 times less expensive while likely providing more accurate results than those from general fisheries observer programs where the observers have several jobs onboard (Kindt-Larsen et al., 2012). The cameras monitoring harbor porpoise bycatch were shown to be effective and reliable in trials in gillnet fisheries, where it was noted that they were able to capture bycatch that fell out of the net before it made it onboard (Kindt-Larsen et al., 2012). If the use of mandatory logbooks is to continue in certain fisheries, the use of monitoring cameras may improve reporting through the "observer effect" (e.g., Burns and Kerr, 2008; Porter, 2010; van Helmond et al., 2020), where the accuracy of fisher-reported data greatly increases when there is a way to verify it. Therefore, even if the cameras were to malfunction or not have the ability to identify all bycatch, or if only some of the footage is reviewed as verification, the fisher reports should be a more reliable source of information (Course et al., 2020). This has been shown in Australia, where logbook reporting of marine mammal bycatch significantly increased in the gillnet hook and trap sector of a scalefish and shark fishery after implementation of camera monitoring (Emery et al., 2019). Offering some compensation to fishers for turning in accurate reports could be considered in conjunction with camera monitoring to provide incentive to fishers to support the program. If an annual check of cetacean bycatch from the video footage closely matched the logbooks, then the vessel could be compensated in some way, such as a rebate on mandatory fishing license and operation fees or an additional share of the fishing quota. This could be a way to build a better relationship between fishers and scientists and gather accurate bycatch information.

Under-reporting of cetacean bycatch/interaction in fisher logbooks, despite reporting legislation, was clearly quantified for separate gear types for the first time in the case studies presented here, by comparing these data to data from established observer programs. This issue is a detriment to fisheries and cetacean population management. Given the high costs of observer programs and the suggestion that some fishers/fisheries express concerns for having an observer on board for health, safety and financial reasons (e.g., Hulac, 2020; Moore, 2020 – National Fisherman – 25.02.2020), electronic monitoring could be a viable option to eliminate the need for many observers onboard vessels and provide an intermediate solution, balancing the views of government, scientists and fishers, in addition to gathering more accurate data in an unbiased manner. Electronic monitoring can be used in conjunction with logbook reporting as a way to improve and verify reports or could be considered to replace logbook reporting altogether.

DATA AVAILABILITY STATEMENT

The data analyzed in this study is subject to the following licenses/restrictions: The authors of this article were granted data from government agencies and research institutes upon request, with the stipulation not to share this data with any third-parties. The authors can provide information on how to request this same data to interested parties. Requests to access these datasets should be directed to CJB.

AUTHOR CONTRIBUTIONS

CJB and GMS compiled the data and wrote the manuscript. CJB analyzed the data. Both authors contributed to the article and approved the submitted version.

ACKNOWLEDGMENTS

The authors would like to thank all the government ministries and agencies globally who provided information and data that was imperative to this work.

REFERENCES

AFMA (2020). *Protected Species Interaction Reports*. Available online at: https://www.afma.gov.au/sustainability-environment/protected-species-management/protected-species-interaction-reports (accessed September 7, 2020)

Atvinnuvega-og nýsköpunarráðuneyti (2020). *Reglugerð um Skráningu og Rafræn Skil Aflaupplýsinga*. Reykjavík: Atvinnuvega-og nýsköpunarráðuneyti.

Au, W. W. L., and Jones, L. (1991). Acoustic reflectivity of nets: implications concerning incidental take of dolphins. *Mar. Mamm. Sci.* 7, 258–273. doi: 10.1111/j.1748-7692.1991.tb00101.x

Australian Fisheries Management Authority (2021). *Observer Program*. Available online at: *https://www.afma.gov.au/monitoring-enforcement/observer-program*. (Accessed September 4, 2021).

Australian Government Department of Agriculture Water and the Environment (2020a). *About the EPBC Act*. Available online at: https://www.environment.gov.au/epbc/about (accessed September 7, 2020)

Australian Government Department of Agriculture Water and the Environment (2020b). *Entanglements*. Available online at: https://www.environment.gov.au/marine/marine-species/cetaceans/entanglements A(accessed September 7, 2020)

Australian Government Department of Agriculture and Water Resources (2018). *Commonwealth Fisheries Bycatch Policy*. Available online at: https://www.agriculture.gov.au/sites/default/files/sitecollectiondocuments/fisheries/environment/bycatch/bycatch.pdf A(accessed September 7, 2020)

Babcock, E. A., Pikitch, E. K., and Hudson, C. G. (2003). *How much Observer Coverage is Enough to Adequately Estimate Bycatch*. Miami, FL: Pew Institute for Ocean Science.

Basran, C. J., and Rasmussen, M. H. (2021). Fishers and whales in Iceland: whale interactions with fishing gear from the fishers' perspective, with focus on humpback whales (*Megaptera novaeangliae*). *J. Cetacean Res. Manag.* (in press).

Baker, C. S., Lukoschek, V., Lavery, S., Dalebout, M. L., Yong-un, M., Endo, T., et al. (2006). Incomplete reporting of whale, dolphin and porpoise "bycatch" revealed by molecular monitoring of Korean markets. *Anim. Conserv.* 9, 474–482. doi: 10.1111/j.1469-1795.2006.00062.x

Basran, C. J., Bertulli, C. G., Cecchetti, A., Rasmussen, M. H., Whittaker, M., and Robbins, J. (2019). First estimates of entanglement rate of humpback whales *Megaptera novaeangliae* observed in coastal Icelandic waters. *Endanger. Species Res.* 38, 67–77. doi: 10.3354/ESR00936

Bravington, M. V., and Bisack, K. D. (1996). Estimates of harbour porpoise bycatch in the gulf of maine sink gillnet fishery, 1990-1993. *Rep. Int. Whaling Comm.* 46, 567–574SC47SM24.

Burns, R. J., and Kerr, G. N. (2008). Observer effect on fisher bycatch reports in the New Zealand ling (*Genypterus blacodes*) bottom longlining fishery, New Zealand. New Zea. *J. Mar. Fresh.* 42, 23–32. doi: 10.1080/00288330809509933

Carruthers, E. H., and Neis, B. (2011). Bycatch mitigation in context: using qualitative interview data to improve assessment and mitigation in a data-rich fishery. *Biol. Conserv.* 144, 2289–2299. doi: 10.1016/j.biocon.2011.06.007

Christensen-Dalsgaard, S., Anker-Nilssen, T., Crawford, R., Bond, A., Sigurðsson, G. M., Glemarec, G., et al. (2019). What's the catch with lumpsuckers? a North Atlantic study of seabird bycatch in lumpsucker gillnet fisheries. *Biol. Conserv.* 240:108278. doi: 10.1016/j.biocon.2019.108278

Clegg, T., and Williams, T. (2020). *Monitoring Bycatches in Norwegian Fisheries – Species Registered by the Norwegian Reference Fleet 2015-2018*. Available online at: https://www.hi.no/en/hi/nettrapporter/rapport-fra-havforskningen-en-2020-8 A(accessed April 5, 2021)

Cornish, V., Benaka, L., and Fogarty, M. (2004). *Evaluating Bycatch: A National Approach to Standardized Bycatch Monitoring Programs*. Washington, DC: NOAA. NOAA Technical Memorandum NMFS-F/SPO-66.

Course, G. P., Pierre, J., and Howell, B. K. (2020). *What's in the Net? Using Camera Technology to Monitor, and Support Mitigation of, Wildlife Bycatch in Fisheries*. London: The Living Planter Centre.

Department of Conservation New Zealand Government (2020a). *How to Report Marine Mammal Sightings*. Available online at: https://www.doc.govt.nz/nature/native-animals/marine-mammals/marine-mammal-sightings/ A(accessed September 8, 2020)

Department of Conservation New Zealand Government (2020b). *CSP Fisheries Observers*. Available online at: https://www.doc.govt.nz/our-work/conservation-services-programme/about-csp/csp-fisheries-observers/ A(accessed September 5, 2021)

Emery, T. J., Noriega, R., Williams, A. J., and Larcombe, J. (2019). Changes in logbook reporting by commercial fishers following the implementation of electronic monitoring in Australian Commonwealth fisheries. *Mar. Policy* 104, 135–145. doi: 10.1016/j.marpol.2019.01.018

FAO (2011). *Iceland. Document: C1083*. Rome: FAO.

FAO (2018). *The State of World Fisheries and Aquaculture: Meeting the Sustainable Development Goals*. Rome: FAO.

FAOLEX Database (2006). *Iceland: Fisheries Management Act, 1990*. Available online at: http://www.fao.org/faolex/results/details/en/c/LEX-FAOC003455 A(accessed September 13, 2020)

Federal Register (2016). *Fish and Fish Product Import Provisions of the Marine Mammal Protection Act*. Available online at: https://www.federalregister.gov/documents/2016/08/15/2016-19158/fish-and-fish-product-import-provisions-of-the-marine-mammal-protection-act A(accessed September 13, 2020)

Federal Register (2020). *Modification of Deadlines Under the Fish and Fish Product Import Provisions of the Marine Mammal Protection Act*. Available online at: https://www.federalregister.gov/documents/2020/11/03/2020-24210/modification-of-deadlines-under-the-fish-and-fish-product-import-provisions-of-the-marine-mammal A(accessed October 08, 2021)

Finlex (2015). *Kalastuslaki*. Available online at: https://finlex.fi/fi/laki/ajantasa/2015/20150379#L6P62 A(accessed September 12, 2020).

Fisheries Agency Research Management Department (2020). *ひげ鯨の混獲状況について (令和2年)*. Available online at: https://www.jfa.maff.go.jp/j/whale/attach/pdf/bycatch-16.pdf A(accessed August 25, 2021)

Fisheries and Oceans Canada (2020a). *Schedule V Marine Mammal Interaction Form*. Available online at: https://www.dfo-mpo.gc.ca/species-especes/documents/mammals-mammiferes/report-rapport/Fish-Harvester-Form-Eng.pdf A(accessed September 7, 2020)

Fisheries and Oceans Canada (2020b). *Report a Marine Mammal or Sea Turtle Incident or Sighting: I See an Injured, Stranded, Entangled or Dead Marine Mammal or Sea Turtle*. Available online at: https://www.dfo-mpo.gc.ca/species-especes/mammals-mammiferes/report-rapport/page01-eng.html A(accessed September 7, 2020)

Fiskeridirektoratet (2017). *Hva er Fiskeridirektoratets Sjøtjeneste?*. Available online at: https://www.fiskeridir.no/Yrkesfiske/Regelverk-og-reguleringer/Stengte-fiskefelt/gammel-Nord-for-62-N/Hva-er-Fiskeridirektoratets-sjoetjeneste A(accessed April 13, 2021)

Fiskistofa (2010). *Erindisbréf fyrir Eftirlitsmenn Fiskistofu*. Available online at: https://www.fiskistofa.is/umfiskistofu/hlutverk/erindisbref/ A(accessed September 5, 2021)

Government of Canada (2018). *Regulations Amending the Marine Mammals Regulations: SOR/2018-126*. Available online at: http://www.gazette.gc.ca/rp-pr/p2/2018/2018-07-11/html/sor-dors126-eng.html A(accessed September 7, 2020)

Government of Canada (2020). *Marine Mammal Regulations SOR/93-56 Fisheries Act*. Available online at: https://laws-lois.justice.gc.ca/eng/regulations/sor-93-56/fulltext.html A(accessed September 7, 2020).

Hall, M. A. (1996). On bycatches. *Rev. Fish Biol. Fisher.* 6, 319–352. doi: 10.1007/BF00122585

Hanrahan, J., and Pelrine, J. V. (1997). *At-sea Observer Program Operations Manual*. Available online at: https://waves-vagues.dfo-mpo.gc.ca/Library/276830.pdf A(accessed April 8, 2021).

Havs-och vattenmyndighetens (2018). *Havs- och Vattenmyndighetens Föreskrifter (HVMFS 2018:11) om Befälhavares Skyldigheter att Rapportera och Anmäla Yrkesmässigt Fiske i Havet*. Available online at: https://www.havochvatten.se/download/18.ffe12416a5562728043580/1556535951364/HVMFS%202018-11-keu%2020190501.pdf (accessed September 12, 2020)

Hulac, B. J. (2020). *Fishing Fleets Say NOAA Observers are too Risky Amid Pandemic. CQ Roll Call*. Available online at: https://www.rollcall.com/2020/04/10/fishing-fleets-say-noaa-observers-are-too-risky-amid-pandemic/ A(accessed April 8, 2021)

ICES (2020). Working group on bycatch of protected species (WGBYC). *ICES Sci. Rep.* 81:209.

Institute for Cetacean Research (2011). 定置網に混獲されたひげ鯨等の取り扱いの手引き 고래자원의 보존과 관리에 관한 고시. Available online at: https://www.icrwhale.org/pdf/higekujira.pdf A(accessed August 25, 2021)

IWC (2011). Report of the scientific committee. *J. Cetacean Res. Manag.* 12:451.

IWC (2020). *Progress Reports for Korea, Republic of*. Available online at: https://portal.iwc.int/progressreportspublic/report A(accessed May 11, 2020).

Johnson, A., Salvador, G., Kenney, J., Robbins, J., Kraus, S., Landry, S., et al. (2005). Fishing gear involved in entanglements of right and humpback whales. *Mar. Mamm. Sci.* 21, 635–645. doi: 10.1111/cobi.12590

Kastelein, R. A., de Haan, D., Vaughan, N., Staal, C., and Schooneman, N. M. (2001). The influence of three acoustic alarms on the behaviour of harbour porpoises (*Phocoena phocoena*) in a floating pen. *Mar. Environ. Res.* 52, 351–371. doi: 10.1016/s0141-1136(01)00090-3

Kindt-Larsen, L., Dalskov, J., Stage, B., and Larsen, F. (2012). Observing incidental harbour porpoise Phocoena phocoena bycatch by remote electronic monitoring. *Endang. Spec. Res.* 19, 75–86. doi: 10.3354/esr00455

Knowlton, A. R., Hamilton, P. K., Marx, M. K., Pettis, H. M., and Kraus, S. D. (2012). Monitoring North Atlantic right whale *Eubalaena glacialis* entanglement rates: a 30 yr retrospective. *Mar. Ecol. Prog. Series* 446, 293–302. doi: 10.3354/meps09923

Legal Information Institute (2020). *1 U.S. Code §1387. Taking of Marine Mammals Incidental to Commercial Fishing Operations*. Available online at: https://www.law.cornell.edu/uscode/text/16/1387 A(accessed September 7, 2020)

Lien, J., Todd, S., and Guigne, J. (1990). "Inferences about perception in large cetaceans, especially humpback whales, from incidental catches in fixed fishing gear, enhancement of nets by "alarm" devices, and the acoustics of fishing gear," in *Sensory Abilities of Cetaceans*, eds J. Thomas and R. Kastelein (New York, NY: Plenum Press), 347–362. doi: 10.1007/978-1-4899-0858-2_23

Lukoschek, V., Funahashi, N., Lavery, S., Dalebout, M. L., Cipriano, F., and Baker, C. S. (2009). The rise of commercial "bycatch whaling" in Japan and Korea. *Anim. Conserv.* 12, 398–399. doi: 10.1111/j.1469-1795.2009.00313.x

Luonnonvarakeskus (2020). *Sivusaalisilmoitus*. Helsinki: Luonnonvarakeskus.

Merrick, R. L., Clapham, P. J., Cole, T. V. N., Gerrior, P., and Pace, R. M. III (2001). *Identification of Seasonal Area Management Zones for North Atlantic Right Whale Conservation*. Woods Hole, MA: National Marine Fisheries Service. Northeast Fisheries Science Centre Reference Document 01-14.

Mills, J. A., Ishirhara, A., Sakaguchi, I., Kang, S., Parry-Jones, R., and Phipps, M. (1997). *Whale Meat Trade in East Asia: a Review of the Markets in 1997*. Cambridge: TRAFFIC International.

Ministère de l'Agriculture et de l'Alimentation (2018). *Arrêté du 6 Septembre 2018 Portant Modification de l'arrêté du 1er juillet 2011 Fixant la Liste des Mammifères Marins Protégés sur le Territoire National et les Modalités de leur Protection*. Available online at: https://www.legifrance.gouv.fr/download/pdf?id=5Tw5TynhQbPmAr5ORS57k4hwvMXPOPNZ5JP4BnMJBZ4= (accessed September 13, 2020)

Ministry for Primary Industries New Zealand (2020). *Fisheries Observers*. Available online at: https://www.mpi.govt.nz/fishing-aquaculture/commercial-fishing/operating-as-a-commercial-fisher/fisheries-observers/ A(accessed April 8, 2021)

Ministry of Oceans and Fisheries Korea (Fisheries Resource Policy Division) (2018). 고래자원의 보존과 관리에 관한 고시. 해양수산부고시 제 *2018-98호, 2018. 8. 27.,* 일부개정.

Mooney, T. A., Au, W. W. L., Nachtigall, P. E., and Trippel, E. A. (2007). Acoustic and stiffness properties of gillnets as they relate to small cetacean bycatch. *ICES J. Mar. Sci.* 64, 1324–1332. doi: 10.1093/icesjms/fsm135

Moore, K. (2020). *New Jersey-based Herring Fleet Sues Over at-sea Monitoring Rule. National Fishermen*. Available online at: https://www.nationalfisherman.com/northeast/new-jersey-based-herring-fleet-sues-over-at-sea-monitoring-rule A(accessed April 8, 2020)

Moore, M. J., Rowles, T. K., Fauquier, D. A., Baker, J. D., Biedron, I., Durbas, J. W., et al. (2021). Assessing North Atlantic right whale health: threats, and development of tools critical for conservation of the species. *Dis. Aquat. Organ.* 143, 205–226. doi: 10.3354/dao03578

Muñoz, R. V., Medina, L. O., Ahumada, B. S., Pizarro, A. G., Vargas, S. H., Araya, R. O., et al. (2018). *INFORME FINAL Convenio de Desempeño 2017. Programa de Observadores Científicos 2017-2018. Programa de Investigación del Descarte y Captura de Pesca Incidental en Pesquerías Pelágicas. Programa de Monitoreo y Evaluación de los Planes de Redución del Descarte y de la Pesca Incidental 2017-2018*. Available online at: https://www.ifop.cl/wp-content/contenidos/uploads/RepositorioIfop/InformeFinal/P-581129.pdf (accessed April 9, 2021)

National Marine Fisheries Service (2020). *National Observer Program FY 2018 Annual Report*. Available online at:NOAATechnical MemorandumNMFSFSPO206 https://spo.nmfs.noaa.gov/sites/default/files/TMSPO206.pdf A(accessed April 7, 2021). NOAA Technical Memorandum NMFS-F/SPO-206

Naturhistoriska riksmuseet-Katarina Loso (2020). *Död Tumlare - fynd Eller Bifångst*. Available online at: https://www.nrm.se/forskningochsamlingar/miljoforskningochovervakning/rapporteradjur/tumlare/dodtumlarefyndellerbifangst.9003620.html (accessed September 12, 2020)

Naturhistoriska riksmuseet–Katarina Loso (2019). *Ersättning*. Available online at: https://www.nrm.se/forskningochsamlingar/miljoforskningochovervakning/rapporteradjur/tumlare/ersattningvidinsamling.9004932.html A(accessed September 12, 2020)

Naturhistoriska riksmuseet–Peter Mortensen (2020). *Om Statens Vilt*. Available online at: https://www.nrm.se/forskningochsamlingar/zoologi/statensvilt/omstatensvilt.9005981.html A(accessed September 12, 2020)

Naturvårdsverket (2016). *Fridlysta Fåglar, Däggdjur, Kräldjur, Groddjur Och Ryggradslösa Djur*. Available online at: http://www.swedishepa.se/upload/var-natur/djur-och-vaxter/fridlyst/fridlysta-blomvaxter/artlista-alla-fridlysta-djurarter.pdf A(accessed September 12, 2020)

NOAA (2011). *Map of the U.S. Exclusive Economic Zone*. Washington, DC: NOAA.

NOAA (2018). *U.S. Pacific Marine Mammal Stock Assessment: 2017 – Short-finned Pilot Whale (Globicephala macrorhynchus): Hawaii Stock*. Washington, DC: NOAA.

NOAA (2019). *NOAA Fisheries Establishes International Marine Mammal Bycatch Criteria for U.S. Imports*. Washington, DC: NOAA.

NOAA Fisheries (2020a). *Marine Mammal Authorization Program*. Available online at: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-authorization-program (accessed September 7, 2020)

NOAA Fisheries (2020b). *Report a Stranded or Injured Marine Animal*. Available online at: https://www.fisheries.noaa.gov/report (accessed September 7, 2020)

NOAA Fisheries (2021a). *Alaska Marine Mammal Observer Program*. Available online at: https://www.fisheries.noaa.gov/alaska/fisheries-observers/alaska-marine-mammal-observer-program (accessed September 5, 2021)

NOAA Fisheries (2021b). *North Pacific Observer Duties*. Available online at: https://www.fisheries.noaa.gov/alaska/fisheries-observers/north-pacific-observer-duties (accessed September 5, 2021)

Nærings-og fiskeridepartementet (2009). *Forskrift om posisjonsrapportering og elektronisk rapportering for norske fiske- og fangstfartøy*. Available online at: https://lovdata.no/dokument/SF/forskrift/2009-12-21-1743/KAPITTEL_3#%C2%A712 (accessed September 13, 2020).

Okuda, T., Nishida, T., and Ichii, T. (2017). "On-board scientific observer program of Japanese trawl vessels," in *Proceedings of the 2nd Meeting of the Southern Indian Ocean Fisheries Agreement (SIOFA) Scientific Committee, 13-17 March 2017*, (Saint Denis).

Ólafsdóttir, D. (2010). *Report on Monitoring of Marine Mammal Bycatch in Icelandic Fisheries, Statistics for 2009 and Review of Previous Information. NAMMCO Scientific Publication NAMMCO/SC17/16*. Reykjavík: Marine Research Institute.

Parliamentary Counsel Office-New Zealand and Legislation (1978). *Marine Mammal Protection Act 1978: 16 Reporting of Accidental Death or Injury*. Available online at: http://www.legislation.govt.nz/act/public/1978/0080/latest/whole.html#DLM25350 (accessed September 8, 2020)

Porter, R. D. (2010). Fisheries observers as enforcement assets: lessons from the North Pacific. *Mar. Policy* 34, 583–589. doi: 10.1016/j.marpol.2009.11.005

R Core Team (2019). *R: A Language and Environment for Statistical Computing*. Vienna: R Foundation for Statistical Computing.

Read, A. J., Drinker, P., and Northridge, S. (2006). Bycatch of marine mammals in U.S. and global fisheries. *Conserv. Biol.* 20, 163–169. doi: 10.1111/j.1523-1739.2006.00338.x

Reeves, R. R., McClellan, K., and Werner, T. B. (2013). Marine mammal bycatch in gillnet and other entangling net fisheries, 1990 to 2011. *Endanger. Species Res.* 20, 71–97. doi: 10.3354/esr00481

Robbins, J. (2009). *Scar-Based Inference into Gulf of Maine Humpback Whale Entanglement: 2003-2006*. Maryland, MD: Center National Marine Fisheries Service. Report to the Northeast Fisheries Science Center National Marine Fisheries Service, EA133F04SE0998.

Robbins, J., and Mattila, D. (2004). *Estimating Humpback Whale (Megaptera novaeangliae) Entanglement Rates on the Basis of Scar Evidence*. Maryland, MD: Center National Marine Fisheries Service. Report to the Northeast Fisheries Science Center National Marine Fisheries Service, 43EANF030121.

Robbins, J., and Mattila, D. K. (2001). "Monitoring entanglements of humpback whales (Megaptera novaeangliae) in the Gulf of maine on the basis of caudal peduncle scarring," in *Report to the 53rd Scientific Committee Meeting of the International Whaling Commission*, (Hammersmith).

Rolland, R. M., McLellan, W. A., Moore, M. J., Harms, C. A., Burgess, E. A., and Hunt, K. E. (2017). Fecal glucocorticoids and anthropogenic injury and mortality in North Atlantic right whales *Eubalaena glacialis*. *Endanger. Species Res.* 34, 417–429. doi: 10.3354/esr00866

Sea Around US (2016). *Catches by Taxon in the waters of New Zealand*. Available online at: http://www.seaaroundus.org/data/#/eez/554?chart=catch-chart&dimension=taxon&measure=tonnage&limit=10 A(accessed April 13, 2021)

Stoller, A. (2020). *Assessing the Feasibility of Citizen Science to Monitor Change and Cetacean Distribution in Iceland*. thesis, Edinburgh: University of Edinburghmastersthesis.

Subsecretaría de Pesca y Acuicultura (2020). *Fija el Texto Refundido, Coordinado y Sistematizado de la Ley N° 18.892, de 1989 y sus Modificaciones, Ley General de Pesca y Acuicultura*. Available online at: https://www.subpesca.cl/portal/615/articles-88020_documento.pdf (accessed November 24, 2020)

Sveriges Riksdag (2020). *Jaktförordning (1987:905)*. Available online at: https://www.riksdagen.se/sv/dokument-lagar/dokument/svensk-forfattningssamling/jaktforordning-1987905_sfs-1987-905 (accessed September 12, 2020)

Tachoires, S., Guichard, B., Peltier, H., Ducloy, P., Benatre, M., Naviner, M., et al. (2018). *Aide à la Declaration des Captures Accidentelles et à la Reconnaissance de Mammifères Marins*. Available online at: https://agriculture.gouv.fr/telecharger/93538?token=d213c191badb8f10020e6f6c745f871c A(accessed September 13, 2020).

Taylor, B. L., Rojas-Bracho, L., Moore, J., Jaramillo-Legorreta, A., Ver Hoef, J. M., Cardenas-Hinojosa, G., et al. (2017). Extinction is imminent for Mexico's endemic porpoise unless fishery bycatch is eliminated. *Conserv. Lett.* 10, 588–595. doi: 10.1111/conl.12331

U.S. Fish and Wildlife Service International Affairs (2020). *Marine Mammal Protection Act*. Available online at: https://www.fws.gov/international/laws-treaties-agreements/us-conservation-laws/marine-mammal-protection-act.html#:~{}:text=The%20Marine%20Mammal%20Protection%20Act,enacted%20on%20October%2021%2C%201972.&text=The%20MMPA%20prohibits%2C%20with%20certain,mammal%20products%20into%20the%20U.S A(accessed September 7, 2020)

van Helmond, A. T. M., Mortensen, L. O., Plet-Hanse, K. S., Ulrich, C., Needle, C. L., Oesterwind, D., et al. (2020). Electronic monitoring in fisheries: lessons from global experiences and future opportunities. *Fish Fish.* 21, 162–189. doi: 10.1111/faf.12425

Víkingsson, G. A. (2011). *). Iceland. Progress Report on Cetacean Research, May 2010 to April 2011, with Statistical Data for the Calendar Year 2010*. Reykjavik: Marine Research Institute.

Víkingsson, G. A., Ólafsdóttir, D., and Gunnlaugsson, Th. (2004). *Iceland. Progress Report on Cetacean Research, April 2003 to April 2004, with Statistical data for the Calendar Year 2003*. Reykjavik: Marine Research Institute.

Víkingsson, G. A., Ólafsdóttir, D., and Gunnlaugsson, Th. (2005). *Iceland. Progress Report on Cetacean Research, April 2004 to May 2005, with Statistical data for the Calendar Year 2004*. Reykjavik: Marine Research Institute.

Young, N. M., and Iudicello, S. (2007). *Worldwide Bycatch of Cetaceans*. Washington, DC: U.S. Department of Commerce. NOAA Technical Memo. NMFS-OPR-36.

Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

frontiers
in Marine Science

REVIEW
published: 10 December 2021
doi: 10.3389/fmars.2021.752356

Estimating Bycatch Mortality for Marine Mammals: Concepts and Best Practices

Jeffrey E. Moore[1]*, Dennis Heinemann[2], Tessa B. Francis[3,4], Philip S. Hammond[5], Kristy J. Long[6], André E. Punt[3], Randall R. Reeves[7], Maritza Sepúlveda[8], Guðjón Már Sigurðsson[9], Margaret C. Siple[10], Gísli A. Víkingsson[9], Paul R. Wade[11], Rob Williams[12] and Alexandre N. Zerbini[11,13,14]

[1] Marine Mammal and Turtle Division, SWFSC, National Marine Fisheries Service, La Jolla, CA, United States, [2] U. S. Marine Mammal Commission, Bethesda, MD, United States, [3] School of Aquatic and Fishery Sciences, University of Washington, Seattle, WA, United States, [4] Puget Sound Institute, University of Washington Tacoma, Tacoma, WA, United States, [5] Sea Mammal Research Unit, Scottish Oceans Institute, University of St Andrews, St Andrews, United Kingdom, [6] Office of Protected Resources, NOAA, National Marine Fisheries Service, Silver Spring, MD, United States, [7] Okapi Wildlife Associates, Hudson, QC, Canada, [8] Facultad de Ciencias, Universidad de Valparaíso, Valparaíso, Chile, [9] Pelagic Division, Marine and Freshwater Research Institute, Hafnarfjörður, Iceland, [10] Resource Assessment and Conservation Engineering Division, NOAA, National Marine Fisheries Service, Seattle, WA, United States, [11] Marine Mammal Laboratory, AFSC, NOAA Alaska Fisheries Science Center (AFSC), Seattle, WA, United States, [12] Oceans Initiative, Seattle, WA, United States, [13] Cooperative Institute for Climate, Ocean, and Ecosystem Studies (CICOES), University of Washington, Seattle, WA, United States, [14] Marine Ecology and Telemetry Research, Seabeck, WA, United States

Edited by:
*Alastair Martin Mitri Baylis,
South Atlantic Environmental
Research Institute, Falkland Islands*

Reviewed by:
*Simon David Goldsworthy,
South Australian Research
and Development Institute, Australia
Justin Cooke,
CEMS, Germany*

***Correspondence:**
*Jeffrey E. Moore
jeff.e.moore@noaa.gov*

Specialty section:
*This article was submitted to
Marine Megafauna,
a section of the journal
Frontiers in Marine Science*

***Received:** 02 August 2021*
***Accepted:** 09 November 2021*
***Published:** 10 December 2021*

Citation:
Moore JE, Heinemann D, Francis TB, Hammond PS, Long KJ, Punt AE, Reeves RR, Sepúlveda M, Sigurðsson GM, Siple MC, Víkingsson GA, Wade PR, Williams R and Zerbini AN (2021) Estimating Bycatch Mortality for Marine Mammals: Concepts and Best Practices. Front. Mar. Sci. 8:752356. doi: 10.3389/fmars.2021.752356

Fisheries bycatch is the greatest current source of human-caused deaths of marine mammals worldwide, with severe impacts on the health and viability of many populations. Recent regulations enacted in the United States under the Fish and Fish Product Import Provisions of its Marine Mammal Protection Act require nations with fisheries exporting fish and fish products to the United States (hereafter, "export fisheries") to have or establish marine mammal protection standards that are comparable in effectiveness to the standards for United States commercial fisheries. In many cases, this will require estimating marine mammal bycatch in those fisheries. Bycatch estimation is conceptually straightforward but can be difficult in practice, especially if resources (funding) are limiting or for fisheries consisting of many, small vessels with geographically-dispersed landing sites. This paper describes best practices for estimating bycatch mortality, which is an important ingredient of bycatch assessment and mitigation. We discuss a general bycatch estimator and how to obtain its requisite bycatch-rate and fisheries-effort data. Scientific observer programs provide the most robust bycatch estimates and consequently are discussed at length, including characteristics such as study design, data collection, statistical analysis, and common sources of estimation bias. We also discuss alternative approaches and data types, such as those based on self-reporting and electronic vessel-monitoring systems. This guide is intended to be useful to managers and scientists in countries having or establishing programs aimed at managing marine mammal bycatch, especially those conducting first-time assessments of fisheries impacts on marine mammal populations.

Keywords: Potential Biological Removal (PBR), seafood import provisions, stock assessment, marine mammal, estimation, bycatch, Marine Mammal Protection Act

INTRODUCTION

Fisheries bycatch is the greatest current source of human-caused deaths of marine mammals worldwide (Lewison et al., 2004; Read et al., 2005; Avila et al., 2018). Bycatch occurs when species not targeted by fishers are incidentally and unintentionally hooked, entangled or entrapped by fishing gear (Hall et al., 2000). Most species of marine mammals—cetaceans, pinnipeds, sirenians and sea otters—are affected by bycatch (Reeves et al., 2013), with hundreds of thousands or perhaps millions of individuals killed annually (Read et al., 2006). Most bycatch occurs in gillnet fisheries (Read, 2008; Reeves et al., 2013), but there is notable bycatch in other types of gear as well, including but not limited to longlines, set nets, stow nets, seines, trawls, and pot or trap gear. Before the 1990s—prior to the enactment of key amendments to the United States Marine Mammal Protection Act (MMPA)—hundreds of thousands of dolphins were killed each year in Eastern Tropical Pacific tuna purse-seine fisheries alone (Hall, 1998). Fisheries-related mortality has been the dominant factor, or at least a major contributing factor, in causing population decline or preventing population recovery (e.g., from historical whaling and sealing impacts) of many marine mammal species. Examples of species highly affected by bycatch include the North Atlantic right whale (*Eubalaena glacialis*), Hector's dolphin (*Cephalorhynchus hectori*), New Zealand sea lion (*Phocarctos hookeri*), the franciscana (*Pontoporia blainvillei*), the nearly-extinct vaquita (*Phocoena sinus*), and the extinct baiji (*Lipotes vexillifer*) (Wang et al., 2006; Slooten, 2007; Turvey et al., 2007; Chilvers, 2008; Secchi, 2010; Rolland et al., 2016; Taylor et al., 2017; Jaramillo-Legoretta et al., 2019; also see Brownell et al., 2019).

In 2016, the United States enacted regulations under the MMPA aimed at reducing marine mammal bycatch in international fisheries[1]. The regulations, stemming from the Fish and Fish Product Import Provisions of the MMPA (hereafter, "MMPA Import Provisions")[2], require fisheries exporting fish and fish products to the United States (hereafter, "export fisheries") to have or establish marine mammal protection standards that are comparable in effectiveness to the standards for United States commercial fisheries. To continue exporting their products to the United States, nations must apply for and receive a "comparability finding" from the United States National Oceanic and Atmospheric Administration[3]. To achieve a comparability finding, the harvesting nation's program regulating an export fishery must: (1) prohibit the intentional killing or serious injury[4] of marine mammals in the fishery, and (2) conduct marine mammal stock (population) assessments that establish bycatch limits for those marine mammal populations interacting with export fisheries, estimating marine mammal bycatch in those fisheries, and taking measures if necessary to reduce total bycatch below the bycatch limits. Alternatively, harvesting nations may adopt other approaches, such as the implementation of bycatch mitigation measures, that are comparable in effectiveness to United States standards for export fisheries [Food and Agriculture Organization of the United Nations [FAO], 2021]. These comparability requirements are conceptually straightforward, but can be difficult to achieve in practice, especially for the most economically challenged countries (Williams et al., 2016). In the United States, the Guidelines for Assessing Marine Mammal Stocks [GAMMS; National Marine Fisheries Service [NMFS], 2016] provide guidance on the key assessment elements: estimating stock abundance, estimating bycatch mortality and serious injury, and comparing the latter to conservation reference points derived from the former. For example, in the United States, bycatch is compared to the conservation reference point call the Potential Biological Removal (PBR) level[5], which is calculated from an estimate of the minimum population size and other parameters. PBR is defined conceptually in the MMPA and operationalized from a management strategy evaluation study by Wade (1998). More generally, comprehensive reviews of protected species reference point estimation and assessment frameworks have been conducted by Lonergan (2011), Moore et al. (2013), Curtis et al. (2015), and Moore and Curtis (2016).

The objective of this paper is to describe best practices for estimating bycatch mortality, which is a key ingredient for population or stock assessment, whereby the mortality estimates are compared to a conservation or limit reference point. Reference point estimation is tied to estimating population size, a topic thoroughly reviewed by Hammond et al. (2021), this issue. An in-depth description of the broader assessment framework is

---

[1] United States Federal Register: 81 FR 54389.

[2] United States Federal Code: 16 U.S.C. 1371(a)(2).

[3] https://media.fisheries.noaa.gov/dam-migration/mmpa_import_rule_compliance_guide_april_2019_eng_508.pdf

[4] The term "serious injury" is defined in United States regulations as "any injury that will likely result in mortality" (50 CFR 229.2), and by policy directive (https://media.fisheries.noaa.gov/dam-migration/02-238.pdf) "likely" is defined as "more likely than not" (i.e., greater than a probability of 0.5).

[5] The MMPA defines PBR as the "maximum number of animals, not including natural mortalities, that may be removed from a marine mammal stock [population] while allowing that stock to reach or maintain its optimum sustainable population." PBR is calculated as the product of the minimum population estimate of the stock (N_{min}), one-half the maximum, theoretical or estimated net productivity rate of the stock at a small population (R_{max}), and a recovery factor (F_r) that ranges from 0.1 to 1.0.

found in Wade et al. (2021), this issue. Readers should examine these papers to understand the broader management context within which bycatch estimation takes place under the MMPA Import Provisions and how estimating marine mammal bycatch and population size relate to each other.

There are important precursors to designing a program to estimate bycatch in a fishery (discussed more thoroughly by Wade et al., 2021). The first is making use of exploratory data (which may need to be collected anew) to characterize the fishery (number of vessels, vessel types, gears used, when and where fished, target species, etc.) and identify marine mammal populations that might interact with it. Our use of the term "fishery" is consistent with its usage under the United States List of Fisheries and List of Foreign Fisheries. That is, a fishery is characterized by a collection of fishers using similar methods (e.g., vessel and gear types), fishing for certain target species, operating in a certain place and time. Examples include the United States drift gillnet fishery for swordfish and thresher shark off the United States West Coast, or the Mexican demersal longline fishery for deepwater snappers in the Gulf of Mexico. Our use of marine mammal "population" is consistent with definitions provided by the GAMMS [National Marine Fisheries Service [NMFS], 2016], i.e., a group of interbreeding individuals that is more or less demographically independent from other groups. The United States marine mammal Stock Assessment Reports provide numerous examples of defined population "stocks" (e.g., Carretta et al., 2021). A marine mammal population may occur entirely within the geographic range of a fishery or the fishery and marine mammal population may only slightly overlap in space or time.

In the absence of data, inferences about the likelihood of bycatch occurrence can be made through exploring the spatial overlap of marine mammal populations and fishing gears known to catch or entangle marine mammals. If bycatch is known or expected to occur, and if negligible impacts to the population cannot be ruled out, then this points to the need to mitigate or undertake a formal bycatch estimation program, which is the focus of the remainder of this paper. Estimation should be prioritized for high-risk gears and fisheries that interact with marine mammal populations at particular risk (see risk categories in Box 1 in Wade et al., 2021, *this issue*). In the context of complying with MMPA Import Provisions, priority should be given to those fisheries categorized as "Export Fisheries" on NOAA's List of Foreign Fisheries[6]. Once obtained, bycatch estimates can then be compared to conservation reference points that depend on the population's size and growth rate to assess the likely or potential impacts of the fishery on the population's viability and whether mitigation actions are needed.

We proceed by discussing a general bycatch estimator and how to obtain the bycatch-rate and fisheries-effort data needed to apply the estimator. Scientific observer programs provide the most robust source of information for estimation and should be used when possible. Observer programs are therefore discussed at length, including program-design considerations, data collection, statistical analysis, and common sources of estimation bias. In addition to their value for directly estimating bycatch, scientific observer programs can also be used to assess and improve compliance of required mitigation measures, estimating the efficacy of such measures (e.g., comparing bycatch rate estimates before and after mitigation, or in sectors of the fisheries with vs. without mitigation), and improve the quality of information provided by fishermen (Cox et al., 2007; Porter, 2010; Snyder and Erbaugh, 2020). Because scientific observer programs tend to be expensive and logistically difficult to implement, we also discuss alternative approaches and data types for making bycatch inferences, and the caveats associated with these. This document is intended to be useful to managers and scientists in harvesting nations maintaining or establishing regulatory programs aimed at reducing marine mammal bycatch, including for the purposes of achieving a comparability finding under the MMPA Import Provisions.

BYCATCH MORTALITY ESTIMATOR

Bycatch mortality is the total number of animals that die (or are expected to die) in a fishery from interacting with fishing gear. Bycatch mortality is typically estimated annually for each gear-specific fishery affecting a defined population. Summing across all fisheries interacting with the population provides a total annual estimate for the population. A general point estimator of bycatch mortality for population *i* is:

$$\mu_{it} = N_{it} E_t c_{it} m_{it}, \tag{1}$$

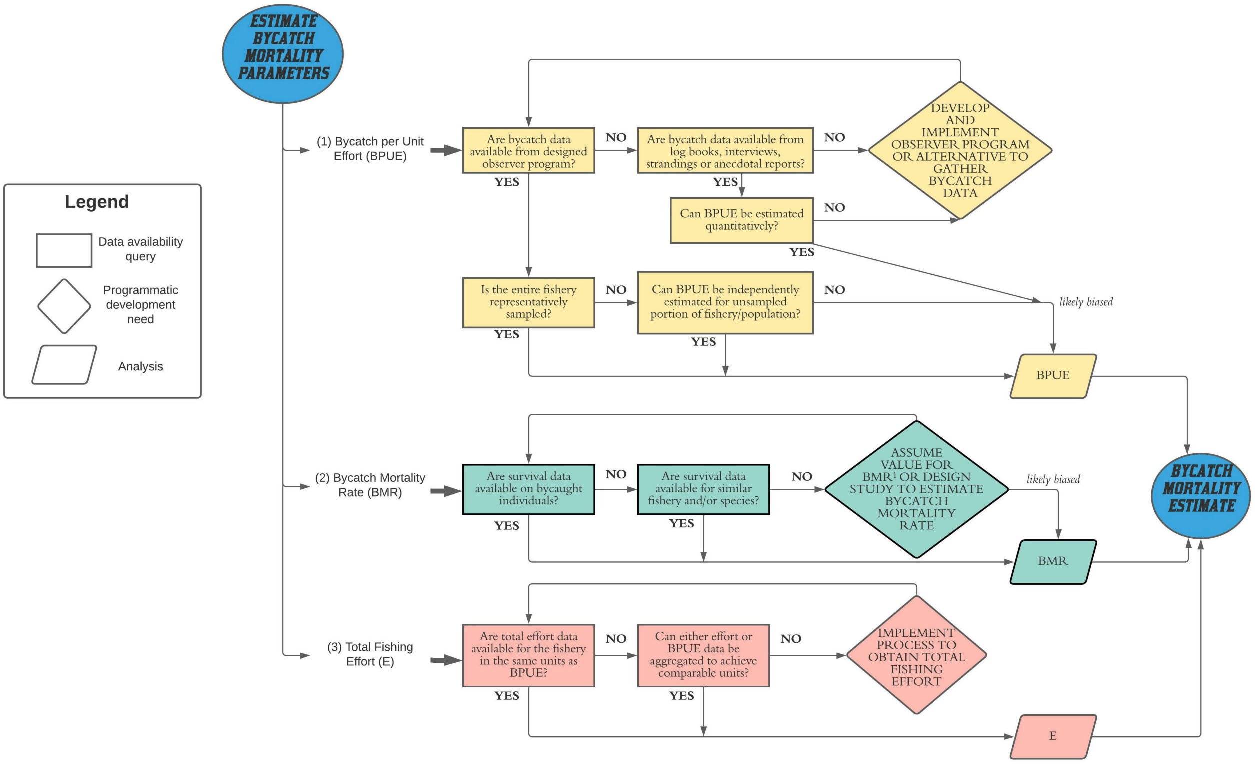
where the expected bycatch mortality in year t, μ_{it}, is the product of animal abundance in the population (N_{it}), total fishing effort (E_t), a scaling parameter referred to as catchability (c_{it}) (which has the unit: bycatch N^{-1} effort^{-1}, and can be thought of as the likelihood that a single animal in the population would be caught by a single unit of fishing effort), and the fraction of bycaught animals that are dead or expected to die (m_{it}; the bycatch mortality rate, BMR), noting that for some types of gear, animals may be released or escape alive after being fatally injured. N_{it} and c_{it} are correlated and in practice will often be difficult to estimate separately. For example, c_{it} will lower if N_{it} is defined as the entire population (including potentially large numbers that never overlap with the fishery), whereas c_{it} will be higher if N_{it} refers to just those animals in the area of the fishery, which may be difficult to estimate. Although there may be cases where c_{it} is explicitly estimated (e.g., from a concurrent time series of abundance and bycatch data; Moore and Curtis, 2016), more typically the product N_{it} c_{it} is estimated as a single parameter referred to as "bycatch per unit effort" (BPUE), or b, where

$$\mu_{it} = b_{it} E_t m_{it}. \tag{2}$$

The sections below give details on how these terms may be estimated, and **Figure 1** illustrates the associated decisions that must be made.

---

[6] https://www.fisheries.noaa.gov/foreign/international-affairs/list-foreign-fisheries



1. Use a precautionary value, typically 1.0.

FIGURE 1 | Decision tree for estimating the components of bycatch mortality: bycatch per unit effort (BPUE), bycatch mortality rate (BMR), and total fishery effort (E). For example, one sex or age class, or a portion of the fishing grounds, missed or incompletely covered by the sampling design.

ESTIMATING *b*, BYCATCH-PER-UNIT-OF-EFFORT FROM FISHERIES OBSERVER DATA

The most accurate way to estimate BPUE is with data from scientific fisheries observer programs, whereby a representative sample of fishing effort is directly observed by independent observers aboard fishing vessels, and the number of marine mammals bycaught (and killed or injured) is recorded. A simple point estimate for BPUE can be calculated as bycatch observed divided by effort observed. For example, if researchers observed 100 gillnet sets and two dolphins were captured, the BPUE would be two dolphins per 100 sets, or 0.02 dolphins per set. If instead of observing and recording sets, the researchers observed 10 complete fishing trips (which might last many days and include multiple sets and retrievals of one or multiple gear types) and counted six dolphins captured, then BPUE would be 0.6 dolphins per trip.

The fraction of a fishing fleet's effort that is observed is referred to as the "observer coverage." BPUE is more precisely estimated for populations in which animals are caught in greater numbers (because either N or c is higher), and in fisheries with higher observer coverage. Small populations for which bycatch is an infrequent or rare event pose particular bycatch estimation challenges (Martin et al., 2015; Gray and Kennelly, 2018; Wakefield et al., 2018) and require fairly high observer coverage levels to avoid severe biases due to small sample size. Curtis and Carretta (2020) developed the observer coverage calculator *ObsCovgTools* in R (R Core Team, 2019) that calculates coverage levels required to meet user-defined bycatch estimation objectives. Objectives include estimating bycatch to a desired precision level, estimating the probability of observing bycatch when it exists in a fishery, and providing an upper confidence limit for bycatch, even if no bycatch is observed. Outputs are conditioned on inputs, such as total effort in the fishery and expected BPUE and sampling variance, which can be obtained from a pilot study or borrowed from a similar study, or based on expert opinion. Under the MMPA, performance tests of the PBR control rule used for setting conservation reference points are based on bycatch estimation coefficients of variation (CVs) of 0.3 or better (Wade, 1998), so we suggest this as a reasonable default input for the target precision.

In addition to having adequate levels of observer coverage, statistically valid BPUE estimates demand the use of well-trained observers and an appropriate survey design. Designing an observer training program and prescribing field protocols (datasheets, etc.) are beyond the scope of this paper, but numerous resources address these topics in detail and should be consulted when designing an observer program [e.g., Pacific Islands Regional Office Observer Program [PIRO-OP], 2017; Northeast Fisheries Science Center [NEFSC], 2019]. As for survey design, the goal is to obtain observer data from a representative sample of the fishery with respect to the suite of attributes that characterize fishing effort, such as the geographic distribution of effort, temporal distribution on diurnal and seasonal timescales, vessel and gear characteristics, and types of effort (e.g., sets, hooks, etc.). For example, a gillnet fishery might have the following hypothetical characteristics: 30% of sets in July, 60% of sets in August, and 10% of sets in September; 30% of sets over the continental shelf and 70% of sets offshore; 50% of sets using long nets deployed from large vessels, and 50% of sets using shorter nets deployed from smaller vessels. Ideally, to avoid a biased estimate the observer dataset would have effort in roughly the same proportions; i.e., the observed component of the fishery would be a microcosm of the whole fishery. If this is not possible, stratification of the sampling and effort can potentially reduce problems associated with non-proportional sampling, as long as the important strata are identified and adequately sampled (see below under discussion of biases). Sampling does not need to be exactly proportional to the effort in each stratum, indeed strata sample sizes can be adjusted to the variance in BPUE in each stratum to increase the estimate's precision without introducing significant bias.

The most statistically valid estimates typically are achieved by stratified random sampling, whereby the fishing effort is subdivided into relatively homogenous subgroups with respect to a particular variable (e.g., by area or season) (e.g., Liggins et al., 1997; Cotter and Pilling, 2007; Benoît and Allard, 2009). Precision is improved especially by sampling more intensively in strata where variance of the bycatch is higher (if this is known), for example if bycatch rates in 1 month tend to be more variable than bycatch rates in other months. This would be the recommended approach if there were sufficient knowledge of all the fishing vessels and their schedule of fishing deployments, and if all vessels could accommodate observers [e.g., bunk and deck space for the observer(s)]. In this scenario, one would randomly select a certain percentage of gear deployments ahead of time and place observers on the vessels expecting to make those deployments. But this is rarely practical in fisheries because of, among other things, uncertainties about who is fishing when and where, and the unwillingness (if observer program participation is voluntary) or inability to accommodate observers (see section "Sources of Bias in Bycatch Estimation"). Whatever the circumstances, the observer program must be diligent about obtaining a sample that accurately and precisely represents the fishery as well as possible (Benoît and Allard, 2009; Benoît et al., 2012; Mangi et al., 2015; Fernandes et al., 2021). It would also be beneficial to representatively sample in relation to spatial-temporal variation in animal density, although this information will often not be available. In situations where the effort is well characterized (e.g., how much fishing effort is occurring when, where, and how), but the observed fishing effort is extremely non-representative (e.g., zero or very small sample sizes in one or more strata), statistical approaches can be used in some cases to eliminate bias in bycatch estimates. Statistical approaches are discussed further below in the "Biases in Bycatch Estimation . . ." section.

ESTIMATING *m*, BYCATCH MORTALITY RATE

Observers typically document bycaught marine mammals as "dead" or "released/escaped alive," often with an assessment of the type of gear interaction, observation of any gear remaining on the animal, and characterization of any injuries. Animals that escape or are released alive might be uninjured or, if injured, could die later or recover and survive. Thus, an unbiased estimate of bycatch mortality, the bycatch mortality rate (BMR), requires an estimate of the proportion of bycaught individuals that die, whether immediately or eventually (i.e., post-release mortality). In the United States, following a bycatch event and based on data collected at the time of detection and observation, the bycaught individual is categorized as "dead," "seriously injured," or "not seriously injured." Those categories are based on guidelines developed through scientific analyses of data on injury severity and outcome, where "seriously injured" was defined as an animal having a greater than 50 percent chance of dying after release, and "not seriously injured" as the animal having a less than 50 percent chance (National Marine Fisheries Service [NMFS], 2012a,b).

Ideally, to determine the post-release mortality rate, bycaught individuals would be tagged prior to release and monitored afterward. Although this approach has been used for marine fish and sharks (e.g., Davis, 2002; Cadigan and Brattey, 2006; Campana et al., 2009; Carruthers et al., 2009; Patterson et al., 2014) and marine turtles (e.g., Álvarez de Quevedo et al., 2013; Stacy et al., 2016; Maxwell et al., 2018; Parga et al., 2020), it has not been employed with marine mammals. Punt et al. (2021) used a modeling approach to estimate post-release mortality rate of two pinniped species bycaught in Chilean purse seine and trawl fisheries. In practice, most bycatch mortality rate estimates are based on small data sets, categorical assignments (e.g., Andersen et al., 2008; Olaya-Ponzone et al., 2020), or expert assessments. The United States injury guidelines are based on either analyses of scarring data or subsequent observations documenting the condition, health, and fate of known individuals following the detection of injuries due to interactions with fishing gear (see case studies in Andersen et al., 2008). Only a small number of published studies provide estimates of BMR (e.g., Wells et al., 2008; Cassoff et al., 2011; Dolman and Moore, 2017; Pettis et al., 2017; Olaya-Ponzone et al., 2020).

It may not always be necessary to estimate BMR. For example, small cetaceans and pinnipeds caught in gillnets and some trawl fisheries are typically found dead. Conservatively, in the absence of data specific to a study population and fishery, it is prudent to set BMR to 1.0 for marine mammals captured in gillnet fisheries, as suffocating or drowning in the nets is by far the most likely outcome. For bycatch in other types of gear (e.g., purse seine, longline, some trawl), approximate values for BMR might be taken from the literature for similar species and gears, but in fact few such estimates are available.

ESTIMATING E, FISHING EFFORT

General Principles for Estimating E

Scaling the observed-sample estimate of BPUE to an estimate of total bycatch in a fishery requires knowing the total amount of effort in the fishery (this is the sampling frame within which a subset of effort has been observed). Critically, the effort metric used for estimating BPUE and for characterizing the whole fishery must be the same. For example, if observers collect data for a random sample of fishing trips with an estimate of how many marine mammals on average are caught per trip, then the total number of trips made by the fleet must be quantified to properly extrapolate to the whole fishery. Similarly, if BPUE is quantified for a gillnet fishery by observing a random number of gillnet deployments (sets), then the total number of gillnet sets made by the fishery must also be known or estimated. In some observer programs, observers monitor all fishing activity over the course of a particular period (e.g., 24 h) and BPUE is measured as the number of bycatch events per effort-period (e.g., per-day); in this case, the number of effort-days (# boats x the # days each boat operates) would need to be known for the fleet. If the sampling frame is incomplete because the size and extent of the fleet has not been accurately determined, then the total bycatch mortality will be underestimated (i.e., negatively biased). Therefore, diligence is needed to identify all of the vessels operating in a particular fishery throughout its range.

Ideally, the units of fishing effort measured should be those most directly related to the amount of bycatch that occurs. For example, for a longline fishery, one might quantify the number of longline sets, or more coarsely, the number or total duration of longline fishing trips. However, the number of hooks on the line and their soak time (e.g., "hook-hours") more closely relates to the likelihood of an animal being bycaught. This distinction is relatively unimportant if large numbers of longline sets or trips (i.e., effort units) are randomly sampled. However, if the size of the observed sample is small or the sampling is biased, then bycatch mortality may be more accurately estimated by measuring the number of bycatch events per hook-hour and scaling this to hook-hours in the fleet (rather than quantifying bycatch per longline trip and the number of trips). Effort recorded in finer units can always be re-scaled into coarser units as needed, whereas data recorded in coarse units cannot be more finely resolved. Of course, there are trade-offs to how finely one measures effort. Obtaining fleet-wide information about the number of vessels and trips is easier and less costly than monitoring the number of hook-hours, for example. In addition, coarser units tend to be more statistically independent. For example, observations of bycatch-per-trip are more likely to be statistically independent than observations of bycatch-per-set, since set data will be correlated in time and space within the same trips. Observations at coarser scales thus tend to give more valid estimates of precision unless autocorrelations in hierarchical or nested datasets are properly taken into account.

Estimating Fishing Effort in Practice

Measures of fishing effort vary greatly, as do the methods for quantifying those measures. McCluskey and Lewison (2008) reviewed the types of effort measures available for different types of fishing fleets around the world, including artisanal or small-scale and industrial fleets (as well as recreational and IUU[7] fleets, not discussed further here). Though not well-defined (Tietze, 2016; Smith and Basurto, 2019), our use of "small-scale" refers to fleets that *tend toward* having lower capital or technological investment, being operated at the household/family level (rather than by companies), and having smaller vessel sizes. In the extreme, these fleets can consist of thousands of such vessels dispersed across vast geographic areas. Effort in small-scale fleets is usually poorly documented and rarely quantified, due to factors such as lack of awareness, funds and infrastructure, and institutional capacity. Interviews with a large, representative sample of small-scale fishers may be the most practical way to get useful estimates of effort (e.g., Gómez-Muñoz, 1990; Moore et al., 2010), and often measures of fishing-effort will necessarily be crude. For example, Lewison and Moore (2012), working with Nigerian colleagues, identified the number of fishing villages in each of three Nigerian states. For each state, they randomly sampled the villages, counted the number of fishing vessels on the beach in these villages and interviewed fishers there (stratified by boat or gear type) to obtain information about fishing methods, gears, seasonality, fish catch, and bycatch of marine mammals and sea turtles. The fishing effort metric was the average number of boats per village, multiplied by the number of fishing villages along the entire coast to estimate the number of boats per state. BPUE, also obtained from the interview data, was quantified in terms of animals caught per vessel per year. Rough total bycatch estimates were derived as catch per vessel (per year) multiplied by the number of active vessels in the state.

For industrial fisheries, a greater variety of methods for quantifying fishing effort data are generally available. In addition to interview approaches, industrial fleets are more amenable to implementing observer programs. Fleet-wide effort can be quantified through complete dockside monitoring when all vessels return to one central port or a few main ports, or using logbook data, whereby data are recorded on when, where, how and how much they fish (e.g., Roman et al., 2014). Collecting spatially and temporally explicit information about fishing effort (e.g., through logbook data) is extremely valuable, especially if total fishing effort (or BPUE) needs to be modeled rather than estimated using design-based approaches (McCluskey and Lewison, 2008). Inaccuracy is a potential pitfall of both interview-based and logbook data due to response bias (e.g., Cosgrove et al., 2016; Northridge et al., 2017; Luck et al., 2020).

---

[7] Illegal, unreported and unregulated.

The most accurate data on fishing effort are obtained from electronic logbooks that provide spatial and temporal fishing effort data, such as *via* a vessel monitoring system (VMS) in which data are uploaded *via* satellite on a regular schedule. A challenge to this approach is the resistance commonly shown by fishers to being monitored. Nevertheless, the availability of VMS and Automatic Identification Systems (AIS) data had led to recent advances in the ability to make inferences about fishing activity (effort levels and distribution) using computer science algorithms such as Global Fishing Watch[8] (e.g., Kroodsma et al., 2018).

In the absence of data to estimate effort directly, effort can sometimes be predicted or inferred from other characteristics of the fishery using models (e.g., McCluskey and Lewison, 2008; Greenstreet et al., 2009; Soykan et al., 2014; Johnson et al., 2017; Adibi et al., 2020), although their accuracy may be difficult to validate and may rely on unrealistic or unsupported assumptions or inaccurate information. For example, fish catch (landings) has been used as a proxy for fishing effort, either directly or through models, but landings data themselves are often inaccurate (e.g., Batista et al., 2015; Pauly and Zeller, 2016).

ESTIMATING μ, TOTAL MORTALITY, AND ITS UNCERTAINTY

Given (1) BPUE (b) and mortality (m) estimates obtained from an unbiased sample of observer data from a fishery and (2) an estimate of that fishery's total fishing effort (E) in comparable units, the simplest and most common estimator for total bycatch is a ratio estimator, whereby b^*m is multiplied by E. Equivalently, if e is the amount of effort observed, so that observer coverage $P = e/E$ is the proportion of the fleet observed, then the bycatch estimator can also be expressed as the bycatch mortality in the observed sample divided by P (Julian and Beeson, 1998; Carretta et al., 2004). For example, if 100 effort units out of 1,000 in the fleet are observed ($P = 0.1$), then estimated total bycatch mortality, $\mu = b \times m \times 1{,}000$ = observed bycatch mortality/0.1. Variance in this estimate is commonly calculated using resampling (e.g., bootstrapping) or delta methods (e.g., Zhou, 2002; Manly, 2011; Cruz et al., 2018). An advantage of bootstrapping is that it facilitates the accounting of variance on appropriate (independent) observational units. Often the independent sampling unit in an observer program is the fishing trip (e.g., it might be possible, given a rough schedule of fishing trips, to sample these randomly) whereas the multiple gear deployments observed within that trip are correlated (occurring in similar time and space and with similar methods). Treating each day or gear deployment as the observational unit would likely over-estimate the precision (underestimate the variance) of the estimates, whereas resampling fishing trips in the bootstrap analysis provides a valid variance estimate. Precision of the bycatch estimate is typically reported using coefficients of variation (CVs), along with other standard precision measures, such as 95% confidence intervals. As noted above, in the United States, performance tests of the PBR framework are based on the assumption that the CV for bycatch in an individual year 0.3 or less.

The above "design-based" methods for estimating bycatch and bycatch mortality assume that bycatch in the observed portion of the fishery can be extrapolated to the whole fishery, because the study is designed in a representative way. In many situations, bycatch is better estimated using a model-based approach, rather than simple ratio estimators. Examples include when the sample data are biased (not collected using a random or other representative sampling scheme), when multiple years of data have been collected and inferences about current bycatch levels can be informed by data from prior years, when multi-year datasets include years when no bycatch was observed (CVs cannot be calculated for these years using a simple ratio estimator), or when one desires to make probabilistic or predictive inferences about the likelihood of bycatch mortality exceeding a bycatch-limit reference point (e.g., in the current or a future year; Martin et al., 2015; Cruz et al., 2018; Carretta et al., 2019; Stock et al., 2019). Model-based approaches are discussed further in the next section.

BIASES IN BYCATCH ESTIMATION—SOURCES AND SOLUTIONS

Non-representative Sampling

Biased sampling (e.g., extreme over- or under-sampling the fishing fleet with respect to characteristics such as area, season, gear, or vessel type) should be avoided if possible, but if the total fishing effort is well characterized, then stratifying the sample of observer data can help address some biases. For example, if a fishery operates over a 3-month period, with most effort occurring in the second month, but most of the observed effort comes from the first month, then bycatch (and variance) can be estimated separately for each month (stratum) and the stratum estimates combined to obtain the total bycatch mortality. However, it is important in this scenario that sampling within each stratum is largely representative of the fishing occurring within the stratum. Ideally, stratification should be built into the study design, to ensure sufficient representativeness and to ensure the adequacy of within-stratum sample sizes. "*Post hoc* stratification" may not overcome severe design biases, such as when sample sizes are very small or absent within some strata, or when sampling biases exist across multiple attributes of the fishery (e.g., sampling in the third month under-represented an important fishing area).

Model-based approaches can be useful when sampling biases cannot be remedied by stratification, and in some other contexts. Model-based estimators use statistical relationships between potential explanatory variables (e.g., properties of a fishing

---

[8] globalfishingwatch.org

deployment in a certain time and place) and a response variable (e.g., bycatch mortality) to make predictions about bycatch mortality in the unobserved component of the fishery. If the sample data capture the range of variation in the important explanatory variables, then these relationships can be described (modeled) and used to predict bycatch throughout the fishery provided that the covariate values are known for all the fishing effort (e.g., from fishery logbooks). For example, Carretta et al. (2019) used a random-forest machine learning approach to estimate marine mammal, sea turtle and seabird bycatch in the California drift-gillnet fishery based on quantified relationships between observed bycatch and a suite of fishing-set characteristics (location, diurnal and seasonal time variables, bathymetry, oceanography, gear characteristics, etc.) (also see Stock et al., 2019 for a random forest example). Authier et al. (2021, this issue) showed how regularized multilevel regression with post-stratification could be used to estimate bycatch from non-representative sampling. Another common framework for estimating bycatch mortality using covariate data is generalized linear or generalized additive modeling (GLMs or GAMs), which can be implemented in a frequentist (Orphanides, 2009; Cruz et al., 2018; Stock et al., 2019) or Bayesian estimation framework (e.g., Martin et al., 2015; Moore and Curtis, 2016). Models can be particularly useful when multiple years of data exist, allowing information-rich years to inform bycatch estimates in more data-limited years, to resolve the problem of unestimable CVs in years when no bycatch is observed, and to evaluate longitudinal relationships to bycatch mortality such as a change in management actions (e.g., Carretta et al., 2019). Bayesian methods in particular are useful for obtaining probabilistic inferences, such as the probability that the bycatch rate has changed in response to a management action or that bycatch mortality exceeds a limit or other threshold (Martin et al., 2015; Moore and Curtis, 2016).

Taking a model-based approach may be the only option for obtaining valid estimates of bycatch if a sampling design is non-representative. Importantly, however, a model-based design cannot always provide unbiased estimation if the survey design is poor. In particular, if important covariates are not adequately sampled across their range of variation, or if many observations are not statistically independent, then the covariate relationships can be incorrect. As described earlier, there is no good substitute for a well-designed survey and fishery observer program.

Inaccurate Counts by Observers

Bycatch mortality estimates can be biased due to inaccurate counts of observed bycatch events (typically undercounts). Undercounts occur when observers are unable to record every bycatch event that occurs during a watch period. Observers may be engaged in other data collection tasks and not detect bycaught individuals, particularly those not brought on deck. The number of bycaught animals recorded by an observer can be less than the number that were actually bycaught because marine mammals caught on hooks can "drop-off," or those entangled in nets can "drop-out," at any time during the fishing or retrieval of the gear (e.g., Hamer et al., 2011). These problems can be exacerbated if the crew inadvertently or deliberately fail to inform the observer of the presence of bycaught individuals, or surreptitiously release or shake an animal out, or off, the gear. This source of bias can be minimized by assuring the cooperation of crews, although in practice it cannot be eliminated because it is very difficult to estimate the frequency of drop-offs and drop-outs that occur out of sight from the vessel.

Deployment Effects

"Deployment effects" refer to factors that make it logistically infeasible to carry out the planned sampling design, forcing non-representative sampling of the fishery (Benoît and Allard, 2009; Faunce and Barbeaux, 2011; Cahalan and Faunce, 2020; Fernandes et al., 2021). These factors (**Table 1**) include unequal ability to observe different vessel types in the fishery, non-participation in the observer program by fishers, inability to observe the fleet operating in certain locations and periods, sub-optimal allocation of observer effort due to incomplete knowledge of the fleet, and other logistical restraints (**Table 1**).

Observer Effects

"Observer effects" occur when fishers use different gear or fishing methods, target different fish species, fish in different areas or at different times, reduce effort per trip, or handle bycatch differently when observers are on board (Liggins et al., 1997; Benoît and Allard, 2009; Faunce and Barbeaux, 2011), presumably to reduce the chance of bycatch occurring (or being detected and reported). Observer effects result in observer data that are not representative of the entire fleet and may not accurately reflect the bycatch that occurred on the observed trips. Subsequently, bycatch mortality estimates are biased and likely more precise than is warranted (Cotter and Pilling, 2007). Observer effects and the resulting "observer biases" are difficult to confirm [National Marine Fisheries Service [NMFS], 2011]. In some studies, observer effects have been inferred based on catch statistics that differed significantly between observed and unobserved portions of a fleet (e.g., Wahlen and Smith, 1985; Walsh et al., 2002; Cotter and Pilling, 2007; Burns and Kerr, 2008; Faunce and Barbeaux, 2011; Kirkwood et al., 2020); some other studies failed to find such differences (e.g., Liggins et al., 1997). It is widely assumed that observer effects are common (e.g., Faunce and Barbeaux, 2011), especially when (i) captains and/or crew believe that observer data can be used against them (e.g., have enforcement consequences or lead to disadvantageous management changes) [National Marine Fisheries Service [NMFS], 2011], that an onboard observer requires additional effort on their part, or that having an observer on board constrains their behavior in some way (Cotter and Pilling, 2007), or (ii) fishers believe that greater profits can be made without observers on board (Furlong and Martin, 2000). One way to address this challenge is for managers to identify and provide incentives for fishers to cooperate, thereby helping to ensure the safety of observers and the integrity of their data. Potential incentives might include financial compensation, increased quota allocation, access to closed areas or seasons, permit fee relief, or access to restricted access fisheries.

If significant observer effects have been documented or are suspected, the problem can be handled in several ways, although

TABLE 1 | Various types of "deployment effects" that lead to biased bycatch estimates.

Factor	Problem	Bias occurs when	Design remedy
Vessel type	Observers are disproportionately placed on some vessel types over others; e.g., small vessels are less likely to be selected due to space constraints or safety concerns	Different vessel types (e.g., sizes) have significantly different bycatch rates	Using electronic monitoring to observe bycatch on vessels that cannot take observers, or placing observers on an alternate platform (e.g., another vessel)
Location and time	Observers are less likely to be placed on vessels operating out of certain locations (e.g., remote ports) or during certain times of the season	Vessels operating in different parts of the fishing grounds or at different times (e.g., seasons) have different bycatch rates, and observer coverage is not proportional to effort in those different areas or times	Detailed understanding of distribution of fishing effort and marine mammals, and the factors that affect their dynamics, to ensure representative observer coverage
Sub-optimal allocation	The optimal allocation of observers requires knowing the universe of trips, which is only possible at the end of the sampling period (e.g., fishing season), yet observers have to be allocated to trips while the season is underway	The real distribution of fishing effort differs significantly from the anticipated distribution upon which observer deployments were based; observed effort is not representative of the fishery	Adaptively modifying observer placements based on within-season monitoring of fishing effort
Logistical constraints	The ability to deploy observers deviates from the initial survey design, for example during periods of intense fishing effort	Some portions of the fishing effort are under-sampled by observers, and those portions have significantly different bycatch rates compared to the rest of the effort	Anticipating factors that could "overwhelm" an observer coverage design prior to deploying observers
Volunteer participation	Operators who volunteer to accommodate observers are more likely to comply with bycatch mitigation measures than operators who do not	The bycatch rates on vessels that volunteer are significantly less than those on vessels that do not volunteer	Requiring all vessels to carry observers, or independently estimating the bycatch rate in the unobserved portion of the fishery

the best approach is to address potential biases in the initial design of the observer program (Benoît and Allard, 2009). Theoretically, bycatch mortality estimates could be corrected if there is an estimate of the bias introduced by the observer effects, although such an estimate is rarely if ever available (Punt, 1999). Bias can be reduced by increasing observer coverage or deploying electronic monitoring devices on the unobserved portion of the fleet, although the latter may introduce its own sources of error (see "*Electronic vessel monitoring as an alternative to observer programs*" below) and may not be economically feasible. The bias should decrease to zero as observer coverage increases to 100%, although there is still the potential for bias due to unrepresentative sampling within trips or to fishers influencing the ability of observers to conduct their duties as required by the observer program (Benoît and Allard, 2009). Finally, a fishery can be stratified such that unbiased bycatch estimates are obtained from the observable vessels, thus confining the problems and bias to just a portion of the fishery, which can be subject to targeted monitoring to account for the under-representation (Furlong and Martin, 2000; Benoît and Allard, 2009).

Cryptic Bycatch Mortality

In general, "cryptic mortality" refers to human-caused mortality that is not, or cannot be, observed. Bycatch should be estimated across all fisheries for a given marine mammal population. However, it is relatively uncommon that all fisheries are observed, and IUU fisheries are, of course, unobserved. Cryptic deaths and injuries can (1) occur in observed fisheries when deaths and injuries are not detected by observers (e.g., drop-offs and drop-outs), (2) go undetected because some fisheries are not observed, or (3) result from "ghost-fishing" (Gilman et al., 2013). Several methods have been developed to estimate the magnitude of overall cryptic mortality (e.g., Williams et al., 2011; Peltier et al., 2012; Barbieri et al., 2013; Gilman et al., 2013; Prado et al., 2013; Wells et al., 2015; Carretta et al., 2016), from which it may be possible to estimate cryptic bycatch mortality. The most common approach estimates the recovery rate of carcasses as the ratio of the number of known deaths due to all causes (obtained, for example, from stranding data) to the estimated total number of deaths in the population (e.g., from a population model). The product of the inverse of the recovery rate and the number of known deaths due to fisheries interactions, excluding those documented by observers, provides an estimate of the undetected (i.e., cryptic) fisheries related mortality. This approach depends strongly on the assumption that the detection rate of deaths due to fisheries interactions is not different from the overall detection rate. Cryptic mortality from all sources, not just bycatch in fisheries, has been estimated to be one-half to two-thirds, and in extreme cases up to and exceeding 90%, of total mortality for marine mammal populations (see references above).

ALTERNATIVE APPROACHES FOR INFERRING BYCATCH IMPACTS

Designing and implementing a new observer program is challenging conceptually, logistically and financially [National Marine Fisheries Service [NMFS], 2011]. For any given level of observer coverage, logistics will often increase in complexity with the size of the fishing area and the fleet, and the number of observers deployed [National Marine Fisheries Service [NMFS], 2011]. Observers need to be well trained and competent in several skill areas, including species identification, collecting scientific measurements and samples, and data security, which can require substantial funding [National Marine Fisheries Service [NMFS], 2011]. Complete (100%) observer coverage, while undeniably providing the best bycatch mortality estimates, is the most expensive and difficult option. Gains in precision diminish as observer coverage approaches 100%. The cost per unit increase in precision increases as observer coverage increases [National Marine Fisheries Service [NMFS], 2011], forcing an "optimal" observer coverage level typically much less than 100%.

While observer programs are generally regarded as the most accurate approach for estimating bycatch, some less-than-ideal alternatives exist that, under favorable circumstances and if implemented well, can provide information to support cruder assessments of marine mammal bycatch in a fishery.

Self-Reporting (Logbooks or Interview Data) as an Alternative to Observer Programs

Vessel logbook data and data collected through "dock-side" interviews, in addition to providing information about fishing effort, can provide information about marine mammal bycatch. Bycatch data collected by these methods are generally incomplete and inaccurate, usually in the direction of under-reporting (e.g., Walsh et al., 2002; Emery et al., 2019; see Mangi et al., 2016 for discussion of the efficacy of self-reporting). It is widely assumed that logbook data are incomplete and inaccurate because fishers are not skilled at collecting fisheries data (e.g., Faunce, 2011; Faunce and Barbeaux, 2011; Sampson, 2011; Mangi et al., 2016), or that they withhold information they believe could have negative consequences for them. Gilman et al. (2019) suggested that fishers "may have an economic or regulatory disincentive to record accurate data." This problem may be less severe where there are strict legal requirements to report bycatch in logbooks, with surveillance, enforcement and punishments in place. Indeed, the use of electronic monitoring (see below) has been shown to improve the quality of logbook data (Emery et al., 2019). If logbook reporting can be assumed to be consistent throughout the fishery, then such data can be useful for extrapolating/estimating from more reliable data (observer program) that are limited in time and/or space.

Many interview-based assessments have been conducted to obtain semi-quantitative or qualitative information for characterizing fisheries in terms of describing gears and vessel-types, spatial or temporal patterns of fishing effort, and interactions with target and bycatch species, and for doing risk mapping, spatial planning and understanding socio-economic drivers of fisheries management issues (e.g., Moore et al., 2010; Liu et al., 2016; Whitty, 2016; Pilcher et al., 2017; Braulik et al., 2018). An advantage of using interview-based approaches to quantify bycatch mortality is the relatively low cost and relative logistical ease of talking with fishers compared to implementing an observer program. However, as is the case for logbook data, interview data are likely to provide biased inference if fishers are not forthcoming and honest (e.g., for fear of regulations that will limit their fishing opportunities), and interview responses are prone to memory error and interviewer effects. Conducting interviews is itself an art that requires skill and training (e.g., Moore et al., 2010; Lewison and Moore, 2012).

Nevertheless, there can be circumstances where self-reporting from logbooks or interviews provides useful information, at the very least providing information on minimum bycatch levels and on when and where at least some bycatch is occurring, in which gear types, and for which species (although accurate species identification can also be a problem with logbook and interview data, as fishers are unlikely to have been trained in species identification). Information from self-reporting can be useful for determining whether an observer program is needed, and, if it is, for guiding initial planning (e.g., prioritizing which fisheries or areas to observe first).

Electronic Vessel Monitoring as an Alternative to Observer Programs

An alternative to using fisheries observers is electronic monitoring using various technologies, such as GPS or AIS, video cameras, and gear sensors, that capture information on fishing location, catch, bycatch, and discards. Electronic monitoring systems can be used to monitor compliance with catch retention requirements or bycatch of protected species. Systems are now available that can monitor fishing activities on a vessel, and they are starting to supplement data collected by observers or to obtain data from previously unmonitored fisheries (Gilman et al., 2019). These systems integrate GPS units, hard disks, gear sensors and video cameras that provide a visual record of what was caught when and where, including bycatch (Mangi et al., 2015; van Helmond et al., 2020). Gear sensors can improve the efficiency of data collection and storage. For example, a reel sensor can determine when a longline is being retrieved and turn the system on only at those times. van Helmond et al. (2020) reviewed 100 pilot studies and 12 operational implementations, as of 2018, to monitor catch from around the world. As electronic monitoring systems are in the early stages of development and use, it is not yet clear how effective they will be at detecting and accurately recording data on marine mammal bycatch. Nonetheless, a number of systems deployed to monitor protected-species bycatch have reported marine mammal or seabird bycatch (McElderry et al., 2007; Evans and Molony, 2011; Kindt-Larsen et al., 2012; Bartholomew et al., 2018; Emery et al., 2019; Glemarec et al., 2020; van Helmond et al., 2020).

While these systems may collect data on numbers of species with relatively high precision, they cannot yet match observers in many tasks (e.g., species identification, measuring and weighing, sample collection) (Gilman et al., 2020). On the other hand, electronic monitoring systems can collect some data that observers cannot necessarily collect consistently (e.g., precise time and location of individual events, nature of handling and disposition of animals), and they can collect data on 100% of the effort during a fishing trip; observers miss some effort when

TABLE 2 | Strengths and weaknesses of electronic monitoring versus observers.

	Electronic monitoring	Human observing
Strengths	• Can be used when using human observers is impracticable or dangerous • Can provide high coverage levels with greater cost effectiveness • Some data types can be collected more accurately • Data can be reviewed and resampled • Relatively low cost after first year • Can have a lower overall cost • Can sample every gear set • Free from observer effects	• Superior data and ID accuracy • Biological samples can be collected • Can collect accurate concurrent environmental data and data on fishing gear and methods • Can assess the condition of bycaught individuals released alive
Weaknesses	• Privacy concerns by captains and crews • Image quality may be substandard if inexpensive equipment is used or if the cameras are not maintained and cleaned • Outlay costs can be high • Ongoing maintenance costs • Requires review and analysis systems, and training of operators/analysts, which can be costly	• High cost for high coverage levels • Potential for deployment and observer effects to bias data • Some vessels cannot take observers due to space or safety limitations • Coercion and corruption of observers • Observers cannot sample all of the time, and can only observe a single area or process at a time • Safety issues

they are off duty or ill, or weather prevents them from being on deck. Important costs associated with electronic monitoring systems include the often substantial time and funding needed to review and analyze the video streams, although advances in machine learning software hold promise for addressing this issue, and the need for video storage, which can be expensive (Margolis and Alger, 2020).

Several authors have identified strengths and weaknesses of electronic monitoring systems and compared the technology to traditional methods (e.g., Mangi et al., 2015; Suuronen and Gilman, 2020; **Table 2**). Because these systems can be "on" all the time, or started and stopped remotely or automatically based on sensor input, the fishers do not know when the system is collecting data. Further, the data are likely to be subsampled later, which also prevents fishers from knowing when they are being monitored. Therefore, the use of these systems could eliminate an observer effect, or discourage fishers from attempting to influence the data collected by an observer, when electronic monitoring is used to supplement observer data. Further, a sampling design applied to the recorded data could be completely representative and would not suffer from a deployment effect. Electronic monitoring can create a record that, for the duration that it is stored, can be revisited to verify information or resampled to address new questions, although most current applications retain raw data only for finite periods because of high data storage costs and infrastructure requirements.

Impediments to deploying and implementing electronic monitoring include resistance from fishers out of concern about the upfront cost, difficulties of installation, especially on small vessels, and privacy issues (McElderry et al., 2007; Mangi et al., 2015). Fishers may consider electronic monitoring an intrusion into their private workspace (Plet-Hansen et al., 2017) and may argue that camera surveillance reflects a governmental mistrust against them (Mangi et al., 2015). There are also concerns that some bycaught marine mammals may not be brought close enough to the vessel to be seen on camera, and regarding the capability of the video cameras to record sufficient detail to confirm the species identification of marine mammals in the water alongside the vessel and determine the extent of their injuries, particularly at night.

What Can Be Inferred Without Bycatch Monitoring Data?

There is no substitute for bycatch monitoring, but in the complete absence of a bycatch data collection system, there are indirect ways to infer whether bycatch is occurring and whether the impacts are likely to be trivial or worse. For example, beach-stranded and at-sea carcasses can provide information on interactions with fisheries and be used to help determine the need for an observer program. In the United States for example, stranding-network volunteers document human-caused injuries and deaths (e.g., as evidenced by vessel strikes, gunshot wounds, hooks, line, or net, or knife marks), and the data from strandings are used in marine mammal stock assessments [National Marine Fisheries Service [NMFS], 2016]. Often, carcasses bear clear evidence of a fishery interaction, although it is often not possible to link each case to a specific fishery or type of fishing activity. Stranding data can rarely be used to estimate bycatch mortality directly, but in some cases, models applied to stranding data have been used to infer estimates of the proportion of carcasses likely to strand ashore or minimum bycatch levels (e.g., Moore and Read, 2008; Williams et al., 2011; Carretta et al., 2016; Peltier et al., 2016, 2020). For pinnipeds, animals at rookeries can show direct evidence of entanglement; Page et al., 2004 used such data to calculate minimum entanglement mortality estimates. If a minimum estimate itself approaches or exceeds a bycatch-limit reference point (e.g., PBR) that may be sufficient to conclude that a management problem exists that needs to be addressed through an active effort to collect bycatch data more directly to inform mitigation.

It is widely understood that certain gear types represent a predictable threat to particular groups of marine mammals (Wade et al., 2021). For example, vertical buoy lines used to mark and retrieve fixed gear such as crab, lobster or fish traps have the potential to entangle large whales, and to result in their serious injury and death, but may not be a threat to smaller species. In contrast, gill nets are a serious threat to most marine mammals, including porpoises, dolphins and pinnipeds, as well as whales and sirenians. Similar to buoy lines, trawls, seines and longlines can be significant threats to particular marine mammals. Careful comparison by experts of the characteristics of an unobserved fishery with those of similar fisheries with known bycatch rates, combined with consideration of the extent of spatial-temporal overlap between the fishery and the distribution of marine mammal populations, can be used to make qualitative inferences about the likelihood of a population-level problem. Inferences of any kind can be made stronger by drawing upon multiple lines of information.

CONCLUDING REMARKS

This paper is especially intended for fisheries managers and researchers attempting to conduct first-time assessments of fisheries impacts on marine mammal populations. We have tried to break down the daunting challenge of estimating bycatch mortality, highlighting key central concepts, best practices, and typical impediments to obtaining good estimates. Bycatch estimates need to be compared to conservation reference points, which are derived for marine mammal populations mainly from estimates of population size. Population size and reference point estimation are not covered here, but we have provided references on these topics, and a more complete treatment of estimating abundance and reference points can be found in Hammond et al. (2021) and Wade et al. (2021), respectively, in this issue. Scientific observer programs are the only known way to obtain the data needed to estimate bycatch accurately. We therefore place considerable emphasis on this topic and hope the principles discussed in this paper will be useful for those developing fledgling observer programs. Importantly, the main principles—e.g., estimators and measurement units, survey design and statistical considerations, sources of bias—should be useful for the application of alternative bycatch estimation approaches (e.g., using logbooks, interviews) to the extent that these can be incorporated. Alternatives to observer programs have the key advantage of cost-effectiveness. If done well, they can provide useful information for the assessment process and in some cases may be sufficient for determining whether bycatch mitigation is required.

AUTHOR CONTRIBUTIONS

JM and DH designed and wrote the manuscript. TF helped design and produced the figures. All co-authors developed the concept, contributed to writing and editing, and approved its publication. All authors contributed to the article and approved the submitted version.

FUNDING

Support for this project was provided by the Lenfest Ocean Program (Contract ID: #31008).

ACKNOWLEDGMENTS

The authors would like to thank Jim Carretta for reviewing and improving the manuscript.

REFERENCES

Adibi, P., Pranovi, F., Raffaetà, A., Russo, E., Silvestri, C., Simeoni, M., et al. (2020). "Predicting fishing effort and catch using semantic trajectories and machine learning," in *Multiple-Aspect Analysis of Semantic Trajectories. MASTER 2019. Lecture Notes in Computer Science, vol 11889*, eds K. Tserpes, C. Renso, and S. Matwin (Cham: Springer), 83–99. doi: 10.1007/978-3-030-38081-6_7

Álvarez de Quevedo, I., San Félix, M., and Cardona, L. (2013). Mortality rates in by-caught loggerhead turtle *Caretta caretta* in the Mediterranean Sea and implications for the Atlantic populations. *Mar. Ecol. Prog. Ser.* 489, 225–234. doi: 10.3354/meps10411

Andersen, M. S., Forney, K. A., Cole, T. V. N., Eagle, T., Angliss, R., Long, K., et al. (2008). *Differentiating serious and non-serious injury of marine mammals: Report Of The Serious Injury Technical Workshop, 10-13 September 2007, Seattle, Washington*. Washington: NOAA, 94. https://repository.library.noaa.gov/view/noaa/4389.

Authier, M., Rouby, E., and MacLeod, K. (2021). Estimating cetacean bycatch from non-representative samples (I): a simulation study with regularized multilevel regression and post-stratification. *Front. Mar. Sci.* 8:719956. doi: 10.3389/fmars.2021.719956

Avila, I. C., Kaschner, K., and Dormann, C. F. (2018). Current global risks to marine mammals: taking stock of the threats. *Biol. Conserv.* 221, 44–58. doi: 10.1016/j.biocon.2018.02.021

Barbieri, M. M., Raverty, S., Hanson, M. B., Venn-Watson, S., Ford, J. K. B., and Gaydos, J. K. (2013). Spatial and temporal analysis of killer whale (*Orcinus orca*) strandings in the North Pacific Ocean and the benefits of a coordinated stranding response protocol. *Mar. Mamm. Sci.* 29, E448–E462. doi: 10.1111/mms.12044

Bartholomew, D. C., Mangel, J. C., Alfaro-Shigueto, J., Pingo, S., Jimenez, A., and Godley, B. J. (2018). Remote electronic monitoring as a potential alternative to on-board observers in small-scale fisheries. *Biol. Conserv.* 219, 35–45. doi: 10.1016/j.biocon.2018.01.003

Batista, M. I, Horta e Costa, B., Gonçalves, L., Henriques, M., Erzini, K., Caselle, J. E., et al. (2015). Assessment of catches, landings and fishing effort as useful tools for MPA management. *Fish. Res.* 172, 197–208. doi: 10.1016/j.fishres.2015.07.020

Benoît, H. P., and Allard, J. (2009). Can the data from at-sea observer surveys be used to make general inferences about catch composition and discards?. *Can. J. Fish. Aquat. Sci.* 66, 2025–2039. doi: 10.1139/F09-116

Benoît, H. P., Hurlburt, T., Chassé, J., and Jonsen, I. D. (2012). Estimating fishery-scale rates of discard mortality using conditional reasoning. *Fish. Res.* 125-126, 318–330. doi: 10.1016/j.fishres.2011.12.004

Braulik, G. T., Kasuga, M., Wittich, A., Kiszka, J. J., MacCaulay, J., Gillespie, D., et al. (2018). Cetacean rapid assessment: an approach to fill knowledge gaps and target conservation across large data deficient areas. *Aquat. Conserv.* 28, 216–230. doi: 10.1002/aqc.2833

Brownell, R. L. Jr., Reeves, R. R., Read, A. J., Smith, B. D., Thomas, P. O., Ralls, K., et al. (2019). Bycatch in gillnet fisheries threatens Critically Endangered small cetaceans and other aquatic megafauna. *Endanger. Species Res.* 40, 285–296. doi: 10.3354/esr00994

Burns, R. J., and Kerr, G. N. (2008). Observer effect on fisher bycatch reports in the New Zealand ling (*Genypterus blacodes*) bottom longlining fishery. *N. Z. J. Mar. Freshw. Res.* 42, 23–32. doi: 10.1080/00288330809509933

Cadigan, N. G., and Brattey, J. (2006). Reporting and shedding rate estimates from tag recovery experiments on Atlantic cod (*Gadus morhua*) in coastal Newfoundland. *Can. J. Fish. Aquat. Sci.* 63, 1944–1958. doi: 10.1139/f06-096

Cahalan, J., and Faunce, C. (2020). Development and implementation of a fully randomized sampling design for a fishery monitoring program. *Fish. Bull.* 118, 87–99. doi: 10.7755/FB.118.1.8

Campana, S. E., Joyce, W., and Manning, M. J. (2009). Bycatch and discard mortality in commercially caught blue sharks *Prionace glauca* assessed using archival satellite pop-up tags. *Mar. Ecol. Prog. Ser.* 387, 241–253. doi: 10.3354/meps08109

Carretta, J. V., Danil, K., Chivers, S. J., Weller, D. W., Janiger, D. S., Berman-Kowalewski, M., et al. (2016). Recovery rates of bottlenose dolphin (*Tursiops truncatus*) carcasses estimated from stranding and survival rate data. *Mar. Mamm. Sci.* 32, 349–362. doi: 10.1111/mms.12264

Carretta, J. V., Moore, J. E., and Forney, K. A. (2019). *Estimates of Marine Mammals, Sea Turtle, and Seabird Bycatch from the California Large-Mesh Drift Gillnet Fishery: 1990-2017*. Washington: NOAA, 76.

Carretta, J. V., Oleson, E. M., Forney, K. A., Muto, M. M., Weller, D. W., Lang, A. R., et al. (2021). *U.S. Pacific marine mammal stock assessments: 2020*. Washington: NOAA, 394.

Carretta, J. V., Price, T., Petersen, D., and Read, R. (2004). Estimates of marine mammal, sea turtle, and seabird mortality in the California Drift Gillnet Fishery for swordfish and thresher shark, 1996-2002. *Mar. Fish. Rev.* 66, 21–30.

Carruthers, E. H., Schneider, D. C., and Neilson, J. D. (2009). Estimating the odds of survival and identifying mitigation opportunities for common bycatch in pelagic longline fisheries. *Biol. Conserv.* 142, 2620–2630. doi: 10.1016/j.biocon.2009.06.010

Cassoff, R. M., Moore, K. M., McLellan, W. A., Barco, S. G., Rotstein, D. S., and Moore, M. J. (2011). Lethal entanglement in baleen whales. *Dis. Aquat. Organ.* 96, 175–185. doi: 10.3354/dao02385

Chilvers, B. L. (2008). New Zealand sea lions *Phocarctos hookeri* and squid trawl fisheries: bycatch problems and management options. *Endanger. Species Res.* 5, 193–204. doi: 10.3354/esr00086

R Core Team. (2019). *R: A Language and Environment for Statistical Computing*. Vienna: R Foundation for Statistical Computing.

Cosgrove, R., Gosch, M., Reid, D., Sheridan, M., Chopin, N., Jessopp, M., et al. (2016). Seal bycatch in gillnet and entangling net fisheries in Irish waters. *Fish. Res.* 183, 192–199. doi: 10.1016/j.fishres.2016.06.007

Cotter, A. J. R., and Pilling, G. M. (2007). Landings, logbooks and observer surveys: improving the protocols for sampling commercial fisheries. *Fish Fish.* 8, 123–152. doi: 10.1111/j.1467-2679.2007.00241.x

Cox, T. M., Lewison, R. L., Žydelis, R., Crowder, L. B., Safina, C., and Read, A. J. (2007). Comparing effectiveness of experimental and implemented bycatch reduction measures: the ideal and the real. *Conserv. Biol.* 21, 1155–1164. doi: 10.1111/j.1523-1739.2007.00772.x

Cruz, M. J., Machete, M., Menezes, G., Rogan, E., and Silva, M. A. (2018). Estimating common dolphin bycatch in the pole-and-line tuna fishery in the Azores. *PeerJ* 6:e4285. doi: 10.7717/peerj.4285

Curtis, K. A., and Carretta, J. V. (2020). ObsCovgTools: assessing observer coverage needed to document and estimate rare event bycatch. *Fish. Res.* 225:105493. doi: 10.1016/j.fishres.2020.105493

Curtis, K. A., Moore, J. E., Boyd, C., Dillingham, P. W., Lewison, R. L., Taylor, B. L., et al. (2015). Managing catch of marine megafauna: guidelines for setting limit reference points. *Mar. Policy* 61, 249–263. doi: 10.1016/j.marpol.2015.07.002

Davis, M. W. (2002). Key principles for understanding fish bycatch discard mortality. *Can. J. Fish. Aquat. Sci.* 59, 1834–1843. doi: 10.1139/f02-139

Dolman, S. J., and Moore, M. J. (2017). "Welfare implications of cetacean bycatch and entanglements," in *Marine Mammal Welfare*, ed. A. Butterworth (Cham: Springer), 41–65. doi: 10.1007/978-3-319-46994-2

Emery, T. J., Noriega, R., Williams, A. J., and Larcombe, J. (2019). Changes in logbook reporting by commercial fishers following the implementation of electronic monitoring in Australian Commonwealth fisheries. *Mar. Policy* 104, 135–145. doi: 10.1016/j.marpol.2019.01.018

Evans, R., and Molony, B. (2011). *Pilot Evaluation Of The Efficacy Of Electronic Monitoring On A Demersal Gillnet Vessel As An Alternative To Human Observers. Fisheries Research Report No. 221*. Western Australia: Department of Fisheries, 20.

Faunce, C. H. (2011). A comparison between industry and observer catch compositions within the Gulf of Alaska rockfish fishery. *ICES J. Mar. Sci.* 68, 1769–1777. doi: 10.1093/icesjms/fsr093

Faunce, C. H., and Barbeaux, S. J. (2011). The frequency and quantity of Alaskan groundfish catcher-vessel landings made with and without an observer. *ICES J. Mar. Sci.* 68, 1757–1763. doi: 10.1093/icesjms/fsr090

Fernandes, A. C., Oroszlányová, M., Silva, C., Azevedo, M., and Coelho, R. (2021). Investigating the representativeness of onboard sampling trips and estimation of discards based on clustering. *Fish. Res.* 234:105778. doi: 10.1016/j.fishres.2020.105778

Food and Agriculture Organization of the United Nations [FAO] (2021). *Fishing Operations. Guidelines To Prevent And Reduce Bycatch Of Marine Mammals In Capture Fisheries*. Rome: FAO, doi: 10.4060/cb2887en

Furlong, W. J., and Martin, P. M. (2000). "Observer deployment in the fishery and regulatory self-enforcement," in *Microbehavior and Macroresults: Proceedings of the 10th Biennial Conference of the International Institute of Fisheries Economics and Trade*, eds A. L. Shriver and R. S. Johnston (Corvallis: International Institute of Fisheries Economics and Trade), 1–8.

Gilman, E., Castejón, V. D. R., Loganimoce, E., and Chaloupka, M. (2020). Capability of a pilot fisheries electronic monitoring system to meet scientific and compliance monitoring objectives. *Mar. Policy* 113:103792.

Gilman, E., Legorburu, G., Fedoruk, A., Heberer, C., Zimring, M., and Barkai, A. (2019). Increasing the functionalities and accuracy of fisheries electronic monitoring systems. *Aquat. Conserv. Mar. Freshw. Ecosyst.* 29, 901–926. doi: 10.1002/aqc.3086

Gilman, E., Suuronen, P., Hall, M., and Kennelly, S. (2013). Causes and methods to estimate cryptic sources of fishing mortality. *J. Fish Biol.* 83, 766–803. doi: 10.1111/jfb.12148

Glemarec, G., Kindt-Larsen, L., Lundgaard, L. S., and Larsen, F. (2020). Assessing seabird bycatch in gillnet fisheries using electronic monitoring. *Biol. Conserv.* 243:108461. doi: 10.1016/j.biocon.2020.108461

Gómez-Muñoz, V. M. (1990). A model to estimate catches from a short fishery statistics survey. *Bull. Mar. Sci.* 46, 719–722.

Gray, C. A., and Kennelly, S. J. (2018). Bycatches of endangered, threatened and protected species in marine fisheries. *Rev. Fish Biol. Fish.* 28, 521–541. doi: 10.1007/s11160-018-9520-7

Greenstreet, S. P. R., Holland, G. J., Fraser, T. W. K., and Allen, V. J. (2009). Modelling demersal fishing effort based on landings and days absence from port, to generate indicators of "activity". *ICES J. Mar. Sci.* 66, 886–901. doi: 10.1093/icesjms/fsp068

Hall, M. A. (1998). An ecological view of the tuna–dolphin problem: impacts and trade-offs. *Rev. Fish Biol. Fish.* 8, 1–34. doi: 10.1023/A:1008854816580

Hall, M. A., Alverson, D. L., and Metuzals, K. I. (2000). By-catch: problems and solutions. *Mar. Pollut. Bull.* 41, 204–219. doi: 10.1016/S0025-326X(00)00111-9

Hamer, D. J., Ward, T. M., Shaughnessey, P. D., and Clark, S. R. (2011). Assessing the effectiveness of the Great Australian Bight Marine Park in protecting the endangered Australian sea lion *Neophoca cinerea* from bycatch mortality in shark gillnets. *Endanger. Species Res.* 14, 203–216. doi: 10.3354/esr00353

Hammond, P. S., Francis, T. B., Heinemann, D., Long, K. J., Moore, J. E., Punt, A. E., et al. (2021). Estimating the abundance of marine mammal populations. *Front. Mar. Sci.* 8:735770. doi: 10.3389/fmars.2021.735770

Jaramillo-Legoretta, A. M., Cardenas-Hinojosa, G., Nieto-Garcia, E., Rojas-Bracho, L., Thomas, L., Ver Hoef, J. M., et al. (2019). Decline towards extinction of Mexico's vaquita porpoise (*Phocoena sinus*). *R. Soc. Open Sci.* 6:190598. doi: 10.1098/rsos.190598

Johnson, A. F., Moreno-Báez, M., Giron-Nava, A., Corominas, J., Erisman, B., Ezcurra, E., et al. (2017). A spatial method to calculate small-scale fisheries effort in data poor scenarios. *PLoS One* 12:e0174064. doi: 10.1371/journal.pone.0174064

Julian, F., and Beeson, M. (1998). Estimates of marine mammal, turtle, and seabird mortality for two California gillnet fisheries: 1990-1995. *Fish. Bull.* 96, 271–284.

Kindt-Larsen, L., Dalskov, J., Stage, B., and Larsen, F. (2012). Observing incidental harbour porpoise *Phocoena phocoena* bycatch by remote electronic monitoring. *Endanger. Species Res.* 19, 75–83. doi: 10.3354/esr00455

Kirkwood, R. J., Goldsworthy, S. D., and Ward, T. M. (2020). *Assessment of dolphin interactions, effectiveness of the Code of Practice and fishing behavior in the South Australian sardine fishery: 2019-20. SARDI Research Report Series No. 1077*. Adeliaide: South Australian Research and Development Institute (Aquatic Sciences), 34.

Kroodsma, D. A., Mayorga, J., Hochberg, T., Miller, N. A., Boerder, K., Ferretti, F., et al. (2018). Tracking the global footprint of fisheries. *Science* 359, 904–908. doi: 10.1126/science.aao5646

Lewison, R. L., and Moore, J. E. (2012). *Improving Interview-Based Assessments of Sea Turtle and Marine Mammal Bycatch in West Africa: Putting Fishing Activity into a Socio-Economic Context. Project Report*. Washington, DC: NOAA Office of International Affairs.

Lewison, R. L., Crowder, L. B., Read, A. J., and Freeman, S. A. (2004). Understanding impacts of fisheries bycatch on marine megafauna. *Trends Ecol. Evol.* 19, 598–604. doi: 10.1016/j.tree.2004.09.004

Liggins, G. W., Bradley, M. J., and Kennelly, S. J. (1997). Detection of bias in observer-based estimates of retained and discarded catches from a multi species trawl fishery. *Fish. Res.* 32, 133–147. doi: 10.1016/S0165-7836(97)00053-2

Liu, M., Lin, M., Turvey, S., and Li, S. (2016). Fishers' knowledge as an information source to investigate bycatch of marine mammals in the South China Sea. *Anim. Conserv.* 20, 182–192. doi: 10.1111/acv.12304

Lonergan, M. (2011). Potential biological removal and other currently used management rules for marine mammal populations: a comparison. *Mar. Policy* 35, 584–589. doi: 10.1016/j.marpol.2011.01.020

Luck, C., Jessopp, M., Tully, O., Cosgrove, R., Rogan, E., and Cronin, M. (2020). Estimating protected species bycatch from limited observer coverage: a case study of seal bycatch in static net fisheries. *Glob. Ecol. Conserv.* 24:e01213. doi: 10.1016/j.gecco.2020.e01213

Mangi, S. C., Dolder, P. J., Catchpole, T. L., Rodmell, D., and de Rozarieux, N. (2015). Approaches to fully documented fisheries: practical issues and stakeholder perceptions. *Fish Fish.* 16, 426–452. doi: 10.1111/faf.12065

Mangi, S. C., Smith, S., and Catchpole, T. L. (2016). Assessing the capability and willingness of skippers towards fishing industry-led data collection. *Ocean Coast Manag.* 134, 11–19. doi: 10.1016/j.ocecoaman.2016.09.027

Manly, B. F. (2011). Bootstrapping with models for count data. *J. Biopharm. Stat.* 21, 1164–1176. doi: 10.1080/10543406.2011.607748

Margolis, S., and Alger, B. (2020). *The National Electronic Monitoring Workshop Report 2019| 2020*. Washington: NOAA Office of Science and Technology, 46.

Martin, S. L., Stohs, S. M., and Moore, J. E. (2015). Bayesian inference and assessment for rare-event bycatch in marine fisheries: a drift gillnet fishery case study. *Ecol. Appl.* 25, 416–429. doi: 10.1890/14-0059.1

Maxwell, S. M., Witt, M. J., Abitsi, G., Aboro, M. P., Agamboue, P. D., Asseko, G. M., et al. (2018). Sea turtles and survivability in demersal trawl fisheries: do comatose olive ridley sea turtles survive post-release?. *Anim. Biotelemetry* 6:11. doi: 10.1186/s40317-018-0155-1

McCluskey, S. M., and Lewison, R. L. (2008). Quantifying fishing effort: a synthesis of current methods and their applications. *Fish Fish.* 9, 188–200. doi: 10.1111/j.1467-2979.2008.00283.x

McElderry, H., McCullough, D., Schrader, J., and Illingworth, J. (2007). *Pilot Study To Test The Effectiveness Of Electronic Monitoring In Canterbury Fisheries*. Wellington: Science & Technical Publishing, 27.

Moore, J. E., Cox, T. M., Lewison, R. L., Read, A. J., Bjorkland, R., McDonald, S. L., et al. (2010). An interview-based approach to assess marine mammal and sea turtle captures in artisanal fisheries. *Biol. Conserv.* 143, 795–805. doi: 10.1016/j.biocon.2009.12.023

Moore, J. E., and Curtis, K. A. (2016). "Developing control rules for threatened bycatch species," in *Management Science in Fisheries: An Introduction to Simulation-Based Methods*, 278-297, eds C. T. T. Edwards and D. J. Dankel (London: Routledge), 20. doi: 10.4324/9781315751443-23

Moore, J. E., Curtis, K. A., Lewison, R. L., Dillingham, P. W., Cope, J. M., Fordham, S. V., et al. (2013). Evaluating sustainability of fisheries bycatch mortality for marine megafauna: a review of conservation reference points for data-limited populations. *Environ. Conserv.* 40, 329–344. doi: 10.1017/S037689291300012X

Moore, J. E., and Read, A. J. (2008). A Bayesian uncertainty analysis of cetacean demography and bycatch mortality using age-at-death data. *Ecol. Appl.* 18, 1914–1931. doi: 10.1890/07-0862.1

National Marine Fisheries Service [NMFS] (2011). *U.S. National Bycatch Report First Edition*. Washington: NOAA, 508.

National Marine Fisheries Service [NMFS] (2012a). *Guidelines for Distinguishing Serious from Non-Serious Injury of Marine Mammals Pursuant to the Marine Mammal Protection Act*. Maryland: National Marine Fisheries Service.

National Marine Fisheries Service [NMFS] (2012b). *Protected Resources Management Process for Distinguishing Serious from Non-Serious Injury of Marine Mammals*. Maryland: National Marine Fisheries Service.

National Marine Fisheries Service [NMFS] (2016). *Guidelines for Preparing Stock Assessment Reports Pursuant to Section 117 of the Marine Mammal Protection Act*. Maryland: National Marine Fisheries Service.

Northeast Fisheries Science Center [NEFSC] (2019). *Northeast Fisheries Observer Program Handbook for Fisheries Observers and Providers*. Maryland: NOAA Fisheries.

Northridge, S., Coram, A., Kingston, A., and Crawford, R. (2017). Disentangling the causes of protected-species bycatch in gillnet fisheries. *Conserv. Biol.* 31, 686–695. doi: 10.1111/cobi.12741

Olaya-Ponzone, L., Espada, R., Moreno, E. M., Marcial, I. C., and García-Gómez, J. C. (2020). Injuries, healing and management of common dolphins (*Delphinus delphis*) in human-impacted waters in the south Iberian Peninsula. *J. Mar. Biol. Assoc. U. K.* 100, 315–325. doi: 10.1017/S0025315420000090

Orphanides, C. D. (2009). Protected species bycatch estimating approaches: estimating harbor porpoise bycatch in U. S. northwestern Atlantic gillnet fisheries. *J. Northw. Atl. Fish. Sci.* 42, 55–76. doi: 10.2960/J.v42.m647

Pacific Islands Regional Office Observer Program [PIRO-OP] (2017). *Hawaii Longline Observer Program Field Manual, Version LM.17.02*. Washington: National Oceanic and Atmospheric Administration.

Page, B., McKenzie, J., McIntosh, R., Baylis, A., Morrissey, A., Calvert, N., et al. (2004). Entanglement of Australian sea lions and New Zealand fur seals in lost fishing gear and other marine debris before and after Government and industry attempts to reduce the problem. *Mar. Pollut. Bull.* 49, 33–42. doi: 10.1016/j.marpolbul.2004.01.006

Parga, M. L., Crespo-Picazo, J. L., Monteiro, D., García-Párraga, D., Hernandez, J. A., Swimmer, Y., et al. (2020). On-board study of gas embolism in marine turtles caught in bottom trawl fisheries in the Atlantic Ocean. *Sci. Rep.* 10:5561. doi: 10.1038/s41598-020-62355-7

Patterson, H., Hansen, S., and Larcombe, J. (2014). *A review of shark bycatch mitigation in tuna longline fisheries. Western and Central Pacific Fisheries Commission, Scientific Committee report, WCPFC-SC10-2014/EB-WP-05*. Kolonia: Western and Central Pacific Fisheries Commission.

Pauly, D., and Zeller, D. (2016). Catch reconstructions reveal that global marine fisheries catches are higher than reported and declining. *Nat. Commun.* 7:10244. doi: 10.1038/ncomms10244

Peltier, H., Authier, M., Dabin, W., Dars, C., Demaret, F., Doremus, G., et al. (2020). Can modelling the drift of bycaught dolphin stranded carcasses help identify involved fisheries? An exploratory study. *Glob. Ecol Conserv.* 21:e00843. doi: 10.1016/j.gecco.2019.e00843

Peltier, H., Authier, M., Deaville, R., Dabin, W., Jepson, P. D., van Canneyt, O., et al. (2016). Small cetacean bycatch as estimated from stranding schemes: the common dolphin case in the northeast Atlantic. *Environ. Sci. Policy* 63, 7–18. doi: 10.1016/j.envsci.2016.05.004

Peltier, H., Dabin, W., Daniel, P., Van Canneyt, O., Dorémus, G., Huon, M., et al. (2012). The significance of stranding data as indicators of cetacean populations at sea: modelling the drift of cetacean carcasses. *Ecol. Ind.* 18, 278–290. doi: 10.1016/j.ecolind.2011.11.014

Pettis, H. M., Rolland, R. M., Hamilton, P. K., Knowlton, A. R., Burgess, E. A., and Kraus, S. D. (2017). Body condition changes arising from natural factors and fishing gear entanglements in North Atlantic right whales *Eubalaena glacialis*. *Endanger. Species Res.* 32, 237–249. doi: 10.3354/esr00800

Pilcher, N. J., Adulyanukosol, K., Das, H., Davis, P., Hines, E., Kwan, D., et al. (2017). A low-cost solution for documenting distribution and abundance of endangered marine fauna and impacts from fisheries. *PLoS One* 12:e0190021. doi: 10.1371/journal.pone.0190021

Plet-Hansen, K. S., Eliasen, S. Q., Mortensen, L. O., Bergsson, H., Olesen, H. J., and Ulrich, C. (2017). Remote electronic monitoring and the landing obligation–some insights into fishers' and fishery inspectors' opinions. *Mar. Policy* 76, 98–106. doi: 10.1016/j.marpol.2016.11.028

Porter, R. D. (2010). Fisheries observers as enforcement assets: lessons from the North Pacific. *Mar. Policy* 34, 583–589. doi: 10.1016/j.marpol.2009.11.005

Prado, J. H. F., Secchi, E. R., and Kinas, P. G. (2013). Mark-recapture of endangered franciscana dolphin (*Pontoporia blainvillei*) killed in gillnet fisheries to estimate past bycatch time series of stranded carcasses in southern Brazil. *Ecol. Ind.* 32, 35–41. doi: 10.1016/j.ecolind.2013.03.005

Punt, A. E. (1999). "Evaluating the costs and benefits of alternative monitoring programmes for fisheries management," in *Proceedings of the International Conference on Integrated Fisheries Monitoring, Sydney, Australia, 1-5 February 1999*, ed. C. P. Nolan (Rome: FAO).

Punt, A. E., Sepúlveda, M., Siple, M. C., Moore, J., Francis, T. B., Hammond, P. S., et al. (2021). Assessing pinniped bycatch mortality with uncertainty in abundance and post-release mortality: a case study from Chile. *Fish. Res.* 235:105816. doi: 10.1016/j.fishres.2020.105816

Read, A. J. (2008). The looming crisis: interactions between marine mammals and fisheries. *J. Mamm.* 89, 541–548. doi: 10.1644/07-MAMM-S-315R1.1

Read, A. J., Drinker, P., and Northridge, S. (2006). Bycatch of marine mammals in US and global fisheries. *Conserv. Biol* 20, 163–169. doi: 10.1111/j.1523-1739.2006.00338.x

Read, A. J., Reynolds, J. E., Perrin, W. F., Reeves, R. R., Montgomery, S., and Ragen, T. J. (2005). "Bycatch and depredation," in *Marine Mammal Research: Conservation Beyond Crisis*, eds T. J. Ragen, J. E. Reynolds, W. F. Perrin, R. R. Reeves, and S. Montgomery (Baltimore: The John Hopkins Univ. Press), 5–17.

Reeves, R. R., McClellan, K., and Werner, T. B. (2013). Marine mammal bycatch in gillnet and other entangling net fisheries, 1990 to 2011. *Endanger. Species Res.* 20, 71–97. doi: 10.3354/esr00481

Rolland, R. M., Schick, R. S., Pettis, H. M., Knowlton, A. R., Hamilton, P. K., Clark, J. S., et al. (2016). Health of North Atlantic right whales *Eubalaena glacialis* over three decades: from individual health to demographic and population health trends. *Mar. Ecol. Prog. Ser.* 542, 265–282. doi: 10.3354/meps11547

Roman, J., Estes, J. A., Morissette, L., Smith, C., Costa, D., McCarthy, J. B., et al. (2014). Whales as marine ecosystem engineers. *Front. Ecol. Environ.* 12, 377–385. doi: 10.1890/130220

Sampson, D. B. (2011). The accuracy of self-reported fisheries data: Oregon trawl logbook fishing locations and retained catches. *Fish. Res.* 112, 59–75. doi: 10.1016/j.fishres.2011.08.012

Secchi, E. R. (2010). "Review on the threats and conservation status of franciscana, *Pontoporia blainvillei* (Cetacea, Pontoporiidae)," in *Biology, Evolution and Conservation of River Dolphins within South America and Asia*, eds M. Ruiz-García and J. Shostell (New York: Nova Science), 323–339.

Slooten, E. (2007). Conservation management in the face of uncertainty: effectiveness of four options for managing Hector's dolphin bycatch. *Endanger. Species Res.* 3, 169–179. doi: 10.3354/esr003169

Smith, H., and Basurto, X. (2019). Defining small-scale fisheries and examining the role of science in shaping perceptions of who and what counts: a systematic review. *Front. Mar. Sci.* 6:00236. doi: 10.3389/fmars.2019.00236

Snyder, H. T., and Erbaugh, J. T. (2020). Fishery observers address arctic fishery discards. *Environ. Res. Lett.* 15:0940c4. doi: 10.1088/1748-9326/aba57d

Soykan, C. U., Eguchi, T., Kohin, S., and Dewar, H. (2014). Prediction of fishing effort distributions using boosted regression trees. *Ecol. Appl.* 24, 71–83. doi: 10.1890/12-0826.1

Stacy, B. A., Keene, J. L., and Schroeder, B. A. (2016). *Report of the technical expert workshop: Developing national criteria for assessing post-interaction mortality of sea turtles in trawl, net, and pot/trap fisheries: Shepherdstown, West Virginia, USA, 18-22 August 2015*. Washington: NOAA.

Stock, B. C., Ward, E. J., Thorson, J. T., Jannot, J. E., and Semmens, B. X. (2019). The utility of spatial model-based estimators of unobserved catch. *ICES J. Mar. Sci.* 76, 255–267. doi: 10.1093/icesjms/fsy153

Suuronen, P., and Gilman, E. (2020). Monitoring and managing fisheries discards: new technologies and approaches. *Mar. Policy* 116:103554. doi: 10.1016/j.marpol.2019.103554

Taylor, B. L., Rojas-Bracho, L., Moore, J., Jaramillo-Legorreta, A., Ver Hoef, J. M., Cardenas-Hinojosa, G., et al. (2017). Extinction is imminent for Mexico's endemic porpoise unless fishery bycatch is eliminated. *Conserv. Lett.* 10, 588–595. doi: 10.1111/conl.12331

Tietze, U. (2016). *Technical And Socio-Economic Characteristics Of Small-Scale Coastal Fishing Communities, And Opportunities For Poverty Alleviation And Empowerment*. Rome: FAO Fisheries and Aquaculture.

Turvey, S. T., Pitman, R. L., Taylor, B. L., Barlow, J., Akamatsu, T., Barrett, L. A., et al. (2007). First human-caused extinction of a cetacean species?. *Biol. Lett.* 3, 537–540. doi: 10.1098/rsbl.2007.0292

van Helmond, A. T. M., Mortensen, L. O., Plet-Hansen, K. S., Ulrich, C., Needle, C. L., Oesterwind, D., et al. (2020). Electronic monitoring in fisheries: lessons from global experiences and future opportunities. *Fish Fish.* 21, 162–189. doi: 10.1111/faf.12425

Wade, P. R. (1998). Calculating limits to the allowable human-caused mortality of cetaceans and pinnipeds. *Mar. Mamm. Sci.* 14, 1–37. doi: 10.1111/j.1748-7692.1998.tb00688.x

Wade P. R., Long K. J., Francis T. B., Punt A. E., Hammond P. S., Heinemann D., et al. (2021). Best practices for assessing and managing bycatch of marine mammals. *Front. Mar. Sci.* 8:757330. doi: 10.3389/fmars.2021.757330

Wahlen, B. E., and Smith, T. D. (1985). Observer effect on incidental dolphin mortality in the Eastern Tropical Pacific tuna fishery. *Fish. Bull.* 83, 521–530.

Wakefield, C. B., Hesp, S. A., Blight, S., Molony, B. W., Newman, S. J., and Hall, N. G. (2018). Uncertainty associated with total bycatch estimates for rarely-encountered species varies substantially with observer coverage levels: informing minimum requirements for statutory logbook validation. *Mar. Policy* 95, 273–282. doi: 10.1016/j.marpol.2018.05.018

Walsh, W. A., Kleiber, P., and McCracken, M. (2002). Comparison of logbook reports of incidental blue shark catch rates by Hawaii-based longline vessels to fishery observer data by application of a generalized additive model. *Fish. Res.* 58, 79–94. doi: 10.1016/S0165-7836(01)00361-7

Wang, D., Zhang, X., Wang, K., Wei, Z., Würsig, B., Braulik, G. T., et al. (2006). Conservation of the baiji: no simple solution. *Conserv. Biol.* 20, 623–625. doi: 10.1111/j.1523-1739.2006.00471.x

Wells, R. S., Allen, J. B., Hofmann, S., Bassos-Hull, K., Fauquier, D. A., Barros, N. B., et al. (2008). Consequences of injuries on survival and reproduction of common bottlenose dolphins (*Tursiops truncatus*) along the west coast of Florida. *Mar. Mamm. Sci.* 24, 774–794. doi: 10.1111/j.1748-7692.2008.00212.x

Wells, R. S., Allen, J. B., Lovewell, G., Gorzelany, J., Delynn, R. E., Fauquier, D. A., et al. (2015). Carcass-recovery rates for resident bottlenose dolphins in Sarasota Bay, Florida. *Mar. Mamm. Sci.* 31, 355–368. doi: 10.1111/mms.12142

Whitty, T. S. (2016). Multi-methods approach to characterizing the magnitude, impact, and spatial risk of Irrawaddy dolphin (*Orcaella brevirostris*) bycatch in small-scale fisheries in Malampaya Sound, Philippines. *Mar. Mamm. Sci.* 32, 1022–1043. doi: 10.1111/mms.12322

Williams, R., Burgess, M. G., Ashe, E., Gaines, S. D., and Reeves, R. R. (2016). U.S. seafood import restriction presents opportunity and risk. *Science* 354, 1372–1374. doi: 10.1126/science.aai8222

Williams, R., Gero, S., Bejder, L., Calambokidis, J., Kraus, S. D., Lusseau, D., et al. (2011). Underestimating the damage: interpreting cetacean carcass recoveries in the context of the Deepwater Horizon/BP incident. *Conserv. Lett.* 4, 228–233. doi: 10.1111/j.1755-263X.2011.00168.x

Zhou, S. (2002). Estimating parameters of derived random variables: comparison of the delta and parametric bootstrap methods. *Trans. Am. Fish. Soc.* 131, 667–675. doi: 10.1577/1548-86592002131<0667:EPODRV<2.0.CO;2

Author Disclaimer: The views expressed are those of the authors and do not necessarily reflect the views of the Lenfest Ocean Program or the Pew Charitable Trusts.

frontiers in Marine Science

ORIGINAL RESEARCH
published: 13 December 2021
doi: 10.3389/fmars.2021.776965

Participatory Risk Assessment of Humpback Whale (*Megaptera novaeangliae*) and Leatherback Turtle (*Dermochelys coriacea*) Bycatch in Northern Peru

Anna B. Costanza[1]*, Chiara Guidino[2,3], Jeffrey C. Mangel[2,4], Joanna Alfaro-Shigueto[2,3,4], Gregory Verutes[5], Marjolaine Caillat[1], Aritree Samanta[6] and Ellen Hines[1,7]

[1] Estuary and Ocean Science Center, San Francisco State University, Tiburon, CA, United States, [2] ProDelphinus, Lima, Peru, [3] Carrera de Biologia Marina, Universidad Cientifica del Sur, Lima, Peru, [4] Centre for Ecology and Conservation, University of Exeter, Penryn, United Kingdom, [5] CRETUS-Department of Applied Economics, University of Santiago de Compostela, Santiago de Compostela, Spain, [6] School of Public Affairs and Civic Engagement, San Francisco State University, San Francisco, CA, United States, [7] Department of Geography & Environment, San Francisco State University, San Francisco, CA, United States

Edited by:
Andre Eric Punt,
University of Washington,
United States

Reviewed by:
Scott Robert Benson,
National Marine Fisheries Service
(NOAA), United States
Tara Sayuri Whitty,
Independent Researcher,
United States

***Correspondence:**
Anna B. Costanza
annabcostanza@gmail.com

Specialty section:
This article was submitted to
Marine Megafauna,
a section of the journal
Frontiers in Marine Science

***Received:** 14 September 2021*
***Accepted:** 09 November 2021*
***Published:** 13 December 2021*

Citation:
Costanza AB, Guidino C, Mangel JC, Alfaro-Shigueto J, Verutes G, Caillat M, Samanta A and Hines E (2021) Participatory Risk Assessment of Humpback Whale (Megaptera novaeangliae) and Leatherback Turtle (Dermochelys coriacea) Bycatch in Northern Peru. Front. Mar. Sci. 8:776965. doi: 10.3389/fmars.2021.776965

Uncertainties about the magnitude of bycatch in poorly assessed fisheries impede effective conservation management. In northern Peru, small-scale fisheries (SSF) bycatch negatively impacts marine megafauna populations and the livelihoods of fishers which is further elevated by the under-reporting of incidents. Within the last decade, accounts of entangled humpback whales (HBW) (*Megaptera novaeangliae*) off the northern coast of Peru have increased, while Eastern Pacific leatherback turtles (LBT) (*Dermochelys coriacea*) have seen over a 90% decline in nesting populations related in large part to bycatch mortality. By leveraging the experience and knowledge of local fishers, our research objectives were to use a low-cost public participation mapping approach to provide a spatio-temporal assessment of bycatch risk for HBW and LBT off two Peruvian fishing ports. We used an open-source, geographic information systems (GIS) model, the Bycatch Risk Assessment (ByRA), as our platform. Broadly, ByRA identifies high bycatch risk areas by estimating the intersection of fishing areas (i.e., stressors) with species habitat and evaluating the exposure and consequence of possible interaction between the two. ByRA outputs provided risk maps and gear risk percentages categorized as high, medium, and low for the study area and seven subzones for HBW in the austral winter and LBT in the austral summer. Overall, the highest bycatch risk for both species was identified within gillnet fisheries near the coast. Bycatch risk for most gear types decreased with distance from the coast. When we separated the ByRA model by port, our map outputs indicate that bycatch management should be port specific, following seasonal and spatial variations for HBW, and specific fishing gear impacts for HBW and LBT. Combined with direct bycatch mitigation techniques, ByRA can be a supportive and informative tool for addressing specific bycatch threats and marine megafauna conservation goals. ByRA supports a participatory framework offering rapid visual information via risk maps and replicable methods for areas with limited resources and data on fisheries and species habitat.

Keywords: bycatch, small-scale fisheries, participatory GIS (PGIS), bycatch risk assessment, ByRA, marine megafauna bycatch, risk modeling

INTRODUCTION

Fisheries bycatch, defined here as interactions of accidental capture, entanglement, injury, and mortality of non-target species, has been largely accepted as one of the primary threats and drivers of marine megafauna decline (sharks, marine mammals, seabirds, and sea turtles), pushing various species toward extinction (Hall and Roman, 2013; Hamer et al., 2013; Lewison et al., 2014; Hashimoto et al., 2015). Continued losses of individuals from bycatch alter food web dynamics, cause shifts in ecosystem function and services, and can further endanger already depleted or at-risk populations (Worm et al., 2006; Roman and McCarthy, 2010; Estes et al., 2011; Roman et al., 2014; McCauley et al., 2015; Kroodsma et al., 2018).

Megafauna bycatch is a high conservation concern for which there is often inadequate data (Figueiredo et al., 2020; Mannocci et al., 2020). Specifically, in data-poor regions, accessing data required for assessments may be difficult due to the natural complexities of fisheries, especially among artisanal or small-scale fisheries (SSF) (FAO, 2020; Verutes et al., 2020). SSF's are broadly defined as smaller vessels with lesser tonnage, that largely use manual labor as opposed to mechanical equipment, and fish predominately in neritic waters. As SSF's tend to involve fishers who often use more than one gear type, may move between ports with seasonal changes, and have no monitoring technologies, few data are available that capture the intricate details of their gear use, spatial extent, and experience, especially concerning bycatch (Berkes et al., 2001; Cashion et al., 2018; Castillo et al., 2018). SSF's, as do all fisheries, have varying amounts of bycatch, with many unknowns related to risk factors, spatial extent, and quantity (Alfaro-Shigueto et al., 2010, 2011; Alava et al., 2017; Gray and Kennelly, 2018).

Among less accessible SSF's, conducting surveys and hosting participatory workshops with fishers are methods that have proven promising in incorporating local knowledge in the data sharing process and aid in bycatch estimates and mitigation efforts (Mancini et al., 2012; Thiault et al., 2017; Ayala et al., 2019; Mason et al., 2019). Direct fisher interviews used for rapid assessments of bycatch have offered a low-cost and approximate measure of incidental capture of marine megafauna allowing for proper risk assessments, including spatial components of information (Moore et al., 2010; Pilcher et al., 2017).

To support effective conservation that truly includes communities, thorough analyses are needed of the social, biological, and economic factors involved in conservation efforts such as understanding various threats to species vulnerable to bycatch. This process of combining many factors to assess risk and possible consequences contribute to risk assessments (Holsman et al., 2017). Risk assessments are a quantification of an uncertainty which examines both a threat's probability of occurrence and the consequence of that threat (Gibbs and Browman, 2015). In an environmental context, these assessments often narrow in on anthropogenic threats to an environment or species of interest while also emphasizing geographic locations as an important component in quantifying levels of risk (Arkema et al., 2014; Brown et al., 2015; Breen et al., 2017). Within fisheries, for example, risk assessments supported by local knowledge help managers, local community, and stakeholders understand challenges attributed to diverse fishing practices and identify areas of conservation needs that may be overlooked if not well-examined, such as bycatch (Hobday et al., 2011).

In data-poor regions, stakeholder input and consultation not only fill data gaps but offer opportunities for communities to acquire ownership of resource management and resource-related decision-making (Yang and Pomeroy, 2017; Zolkafli et al., 2017; Castellanos-Galindo et al., 2018). Community ownership over conservation measures can be supported by researchers and local non-profits by collaborating with local fishers and community members to aid in documenting data, facilitating projects, and synthesizing community needs (Moore et al., 2017; Szostek et al., 2017; Chung et al., 2019; Brandt et al., 2020).

For example, participatory mapping exercises with stakeholders have encouraged the development of more holistic environmental analyses by using maps to initiate conversation and incorporate different perspectives and knowledge (Levine and Feinholz, 2015; Luizza et al., 2016; Leis et al., 2019). Online and digital mapping platforms such as Geographic Information Systems (GIS) offer a powerful tool for spatial analysis and visualizations to support the collection and organization of data from varying sources (Nelson and Burnside, 2019). GIS has been especially beneficial to the field of conservation when the relationship between people and environment can be explicitly visualized spatially and temporally (Noble et al., 2019).

This process of collaboration with community members, transferring their experiences and understanding of their environments via mapping exercises is called Participatory GIS (PGIS) (Dunn, 2007). PGIS is a low-cost method that has been used in many studies to guide ecosystem management (Croll et al., 2005; Levine and Feinholz, 2015; Strickland-Munro et al., 2016), estimate fishing effort (Thiault et al., 2017), provide location data for rare or endangered species distributions (Rajamani, 2013; Mason et al., 2019), assess anthropogenic threats to coastal environments (Moore et al., 2017; Castellanos-Galindo et al., 2018), and estimate the distribution and magnitude of bycatch (Moore et al., 2010; Pilcher et al., 2017). The various obstacles and challenges associated with identifying the risk of bycatch in SSF can benefit from incorporating different pieces of the puzzle by using PGIS (Lewison et al., 2018).

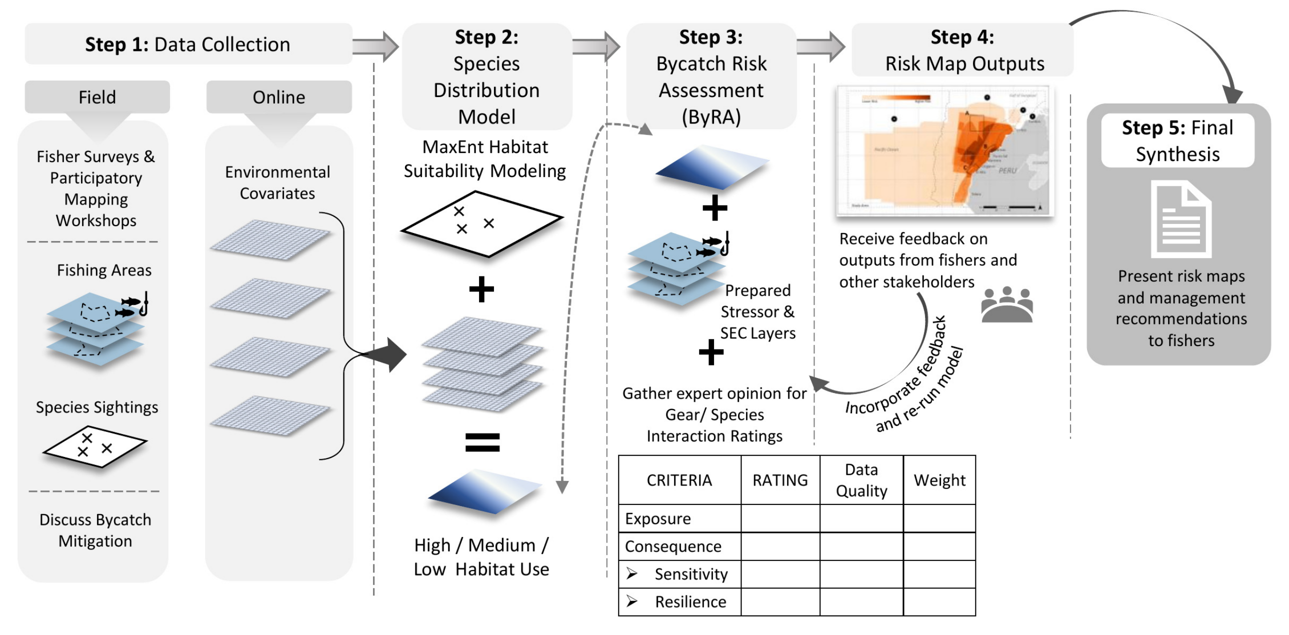
Problems as complex as bycatch can use risk assessments to examine opportunities for marine megafauna conservation among fisheries by simulating complex processes.

Risk assessments are more likely to reflect on-the-ground conditions if coastal community members are actively involved in the discussion and implementation of the risk assessment process (Campbell and Cornwell, 2008; Sawchuk et al., 2015; Visalli et al., 2020). The Bycatch Risk Assessment (ByRA) (**Figure 1**), a spatially explicit analysis that can integrate PGIS data collection methods, was first tested in several southeastern Asian fisheries (Hines et al., 2020; Verutes et al., 2020). The ByRA model offers a structural framework specifically for assessing bycatch in data-poor fisheries by making use of available information and incorporating expert opinion and local stakeholder input via fisher interviews to guide place-based management recommendations for reducing bycatch. Fisher interviews are inevitably dependent on fisher's experiences and willingness to contribute to the sharing of bycatch data (Arlidge et al., 2020). However, by including fisher input, final outputs are more likely to be applicable to end-users and can be tailored to fisher's needs (Scholz et al., 2004; Aburto-Oropeza et al., 2018). In areas with partial data due to limited personnel, training, and funding to support bycatch mitigation strategies, a participatory risk assessment framework, such as ByRA, may be an effective option to examine bycatch and initiate fisher input (Alava et al., 2017).

For this case study, we applied the ByRA model with the following specific objectives: (1) use PGIS to identify areas of fishing and high risk for bycatch of two marine megafauna, (2) provide a spatio-temporal assessment of bycatch risk, and (3) identify gaps in current data monitoring. The ByRA model, run through the freely downloadable software, can be broken down into four phases: (1) conduct a species distribution model for each species, (2) identify and prepare models of fisheries stressors, (3) complete interaction ratings from expert opinion (i.e., bycatch exposure and consequence criteria) for each species (**Table 1**), and (4) produce risk maps that interpret findings to non-expert stakeholders.

Case Study

Along the northern coast of Peru, two large oceanic current systems converge and mix to create the highly biodiverse Tropical Eastern Pacific Bioregion, hosting over 70% of Peru's marine biodiversity (Spalding et al., 2007; Hooker, 2016). This overlap of productive, nutrient-rich waters supports both diverse marine life and fishers who harvest the commercially valuable fish [e.g., tuna (*Thunnus albacares*), smooth hammerhead sharks (*Sphyrna zygaena*), swordfish (*Xiphias gladius*), thresher sharks (*Alopias* spp.), among others]. By 2015, an average of 5,100 SSF vessels were operating in the northern regions of Piura and Tumbes, Peru, providing vital sources of direct consumption of protein at local and national scales, as well as job security, employing over 27,500 fishers (Castillo et al., 2018). These waters also present a persistent risk of bycatch of non-targeted species (Alfaro-Shigueto et al., 2011; García-Godos et al., 2013). Various species of conservation concern within this area such as small cetaceans, sea turtles, sea birds, sharks, and large cetaceans have been documented as bycatch, which predominantly occurs in gillnets and longline gear in these regions. Gillnets act like underwater spiderwebs able to capture many marine megafaunal species that



CRITERIA	RATING	Data Quality	Weight
Exposure			
Consequence			
➢ Sensitivity			
➢ Resilience			

FIGURE 1 | Conceptual model of the Bycatch Risk Assessment (ByRA) process used in our case study. ByRA begins with data collection in Step 1 and ends with the Final Synthesis of the risk maps and recommended management strategies. Each step depicts the basic spatial layers necessary for completing the step and how the output of one process flows into the next. Fisher participation was prevalent in Step 1, 4, and 5 for data collection, data review, and output presentation and discussions. See Verutes et al. (2020) for further description on ByRA model including how ratings scores are assembled, and risk plots are determined.

TABLE 1 | Exposure and Consequence scoring criteria and definitions used to guide final ratings derived from Verutes et al. (2020).

Criteria	High risk (3)	Medium risk (2)	Low risk (1)	Description
Exposure (likelihood)				
Spatial overlap	>30% of species overlaps with gear	10–30% of species overlaps with gear	<10% of species overlaps with gear	The overlap by grid cell between the distribution in space of each species and gear is calculated by ByRA.
Intensity of gear use	High intensity	Medium intensity	Low intensity	Overlap between gear-type density and species distribution.
Likelihood of interaction between species and gear	High likelihood	Medium likelihood	Low Likelihood	The overlap between habitat suitability and intensity of gear use. The resulting encounter rates are ranked low to high (SEC)
Current status of management	No strategies identified or implemented	Management strategies identified, not implemented	Management strategies identified and implemented	Management strategies can limit the use of certain gears in certain areas, thereby mitigating negative impacts to species (SEC)
Likelihood of capture by gear	High likelihood	Medium likelihood	Low Likelihood	The “catchability” of species by gear includes behavior of animal during interaction, for example, dugong may roll around nets.
Temporal overlap (year)	All year (12 months)	Most of year (4–11 months)	Occasional (<4 months)	The duration of time that the species and gear overlap in space.
Temporal overlap (daily net soak time)	8 or more hours	4–7 h	0–4 h	The duration of time nets was set reported by the fishermen. Longer duration would mean greater risk.
Consequence–*sensitivity*				
Mortality	Lethal	Sub-lethal	Negligible	The severity (direct effect) of gear on mortality rate of a species
Life stages affected by gear	Adults only	Mixed	Juvenile	If a gear strands a species before they have the opportunity to reproduce, recovery is likely to be inhibited.
Consequence–*resilience*				
Age at maturity	>4 years	2–4 years	<2 years	Greater age at maturity corresponds to lower productivity.
Reproductive strategy	Long calving interval/high parental invest	medium calving interval/high parental invest	short calving interval/high parental invest	The extent to which a species protects and nourishes its offspring.
Population connectivity	Negligible exchange between the focal regional population and other populations	occasional movement/exchange between the focal regional population and other populations	regular movement/exchange between the focal regional population and other populations	The realized exchange with other populations based on spatial patchiness of distribution, degree of isolation, and potential dispersal capability; based on monitoring surveys or direct tracking estimates.
Local conservation status of species	Endangered	Threatened or of concern	Low concern	The conservation status of species (population level)

can become entangled and drown. This outcome is similar to that of baited longlines that attract unwanted catch via the bait hooked on the lines. Some government organizations interact with SSFs including the Dirección de Capitanías y Puertos (DICAPI) who is the regional government authority who patrols the coast, Pesquería del Ministerio de la Producción (PRODUCE) who monitor fisheries, and the Instituto del Mar del Perú (IMARPE) who is the research arm of PRODUCE. Though there is an active presence of these organizations in the region, there is currently little to no bycatch regulation enforcement, and bycatch incidents go unreported (Van Waerebeek et al., 1997; Mangel et al., 2010; Arlidge et al., 2020).

Two species that highlight this national issue are the Southeast Pacific humpback whale (HBW) (*Megaptera novaeangliae*), and the Eastern Pacific leatherback turtle (LBT) (*Dermochelys coriacea*). These two large, charismatic megafaunas have differing conservation status, yet share similar severe threats, including bycatch (Alfaro-Shigueto et al., 2008; Félix et al., 2011; García-Godos et al., 2013; Ortiz-Alvarez et al., 2020).

The Southeast Pacific HBW population is listed as Least Concern in the IUCN Red List (Cooke, 2018). García-Godos et al. (2013) published the only account of HBW bycatch in northern Peru, documenting a total of ten stranding events between 1995 and 2012 gathered from local news, online evidence, and direct observations. Of these events, nine were entanglements due to drift nets and one from a longline. There have since been continued accounts of entanglements as the humpbacks' coastal migratory route and the southern limit of their winter breeding grounds overlap with SSF operations in northern Peru (Rasmussen et al., 2007; Félix and Botero-Acosta, 2011; Guidino et al., 2014; Pacheco et al., 2021). This threat heightens from mid-July through October when whale presence peaks in northern Peru (Félix and Guzmán, 2014). Rope entanglements can be debilitating to whales by weakening their ability to swim, forage, and mate, increasing their susceptibility to infection (Félix et al., 2011; Moore and van der Hoop, 2012).

The Eastern Pacific population of LBT is listed as Critically Endangered by the International Union for the Conservation of Nature (IUCN) Red List, highlighting bycatch as a primary threat (Wallace et al., 2013). In the last three decades, the EP leatherback turtle population has experienced a greater than 90% decline (Spotila et al., 2000; Shillinger et al., 2008). Predominantly during the austral summer (November–May), adult EP leatherbacks travel from nesting sites in southern Mexico, Costa Rica, and Nicaragua, crossing paths with numerous fisheries, to forage for jellyfish, including to waters off northern Peru (Alfaro-Shigueto et al., 2011; Hoover et al., 2019). Juvenile EP leatherbacks may also be present year-round in and around the study area (Hoover et al., 2019). Leatherback turtle bycatch has been noted from strandings, on-board reporting, and shore-based surveys with fishers (Alfaro-Shigueto et al., 2018; Arlidge et al., 2020; Ortiz-Alvarez et al., 2020).

Bycatch, especially entanglements of large whales, also places burdens on fisher's funds, time and safety. Whale entanglements cause an estimated average loss of $300 USD to the fisher per gillnet pane, which can be a high financial loss given that each vessel has ca.30 panes (Alfaro-Shigueto et al., 2010). Fishers also lose time when fishing, spent on disentangling and discarding unwanted catch, and can risk their lives if they decide to engage in disentangling larger whales. De la Puente et al. (2020) demonstrates that bycatch of megafauna is a significant economic burden that has become debilitating to fishers over time. Due to Peru's dependence on fisheries and the rapid growth of the SSF industry, bycatch management is essential to support megafauna welfare and ecosystem health, as well as support the ability of fishers to continue their trade (De la Puente et al., 2020).

A 2017 provision under the United States Marine Mammal Protection Act (MMPA) will play a role in motivating the development of national regulations (Williams et al., 2016). This provision requires foreign fisheries that export seafood to the U.S. to develop regulatory management that includes estimates of marine mammal abundance, bycatch assessments as well as the implementation of mitigation efforts, and the establishment of programs to monitor and report bycatch (Oceanic National and Administration Atmospheric [NOAA], 2016). SSF's may struggle to comply with these provisions, though building relationships between managers and other stakeholders is encouraged to guide initial regional programs and data collection (Johnson et al., 2017). Although not all fishers in the northern Peru region export to U.S. markets, the presence of conservation measures may have a spill-over effect on SSF practices and handling of bycatch (Williams et al., 2016).

METHODS

Study Sites

We conducted the ByRA model in two fishing ports in northern Peru: Mancora (4° 06′ 38″ S, 81° 04′ 01″ W) and Cancas (3° 56′ 41″ S, 80° 56′ 25″ W) (**Figure 2**). The adjacent coastal and open-water study area spans over 125 mi (>200 km) of coast from Talara to Tumbes, Peru and west across the ocean to the 82° 40' W longitudinal coordinate, covering a total area of just over 40,000 km^2 (15,000 mi^2). These sites were selected because they fall within the core seasonal distribution of SP humpback whales and seasonal foraging range of EP leatherback turtles (Bailey et al., 2012; Guidino et al., 2014) and have well-established small-scale net fisheries with known but poorly quantified bycatch interactions (García-Godos et al., 2013). This region also has growing tourism industries (including whale-watching), and ProDelphinus, a Peru-based NGO and the local collaborator, has a few established contacts in the region. Additionally, the Tropical Eastern Pacific Bioregion has received national conservation interest for possible implementation of several marine protected areas in part due to a prominent seamount, locally known as Banco de Mancora, situated northwest approximately 60 km off the coast, considered an area of high importance for marine biodiversity (**Figure 2**; Nakandakari, 2012; Servicio Nacional de Áreas Naturales Protegidas por el Estado, n.d.). Predominant gear types operating out of these ports are gillnets, longlines, and handlines, with fewer vessels using purse seines (Guevara-Carrasco and Bertrand, 2017). The most recent national fisheries survey from 2018 reports 5,601

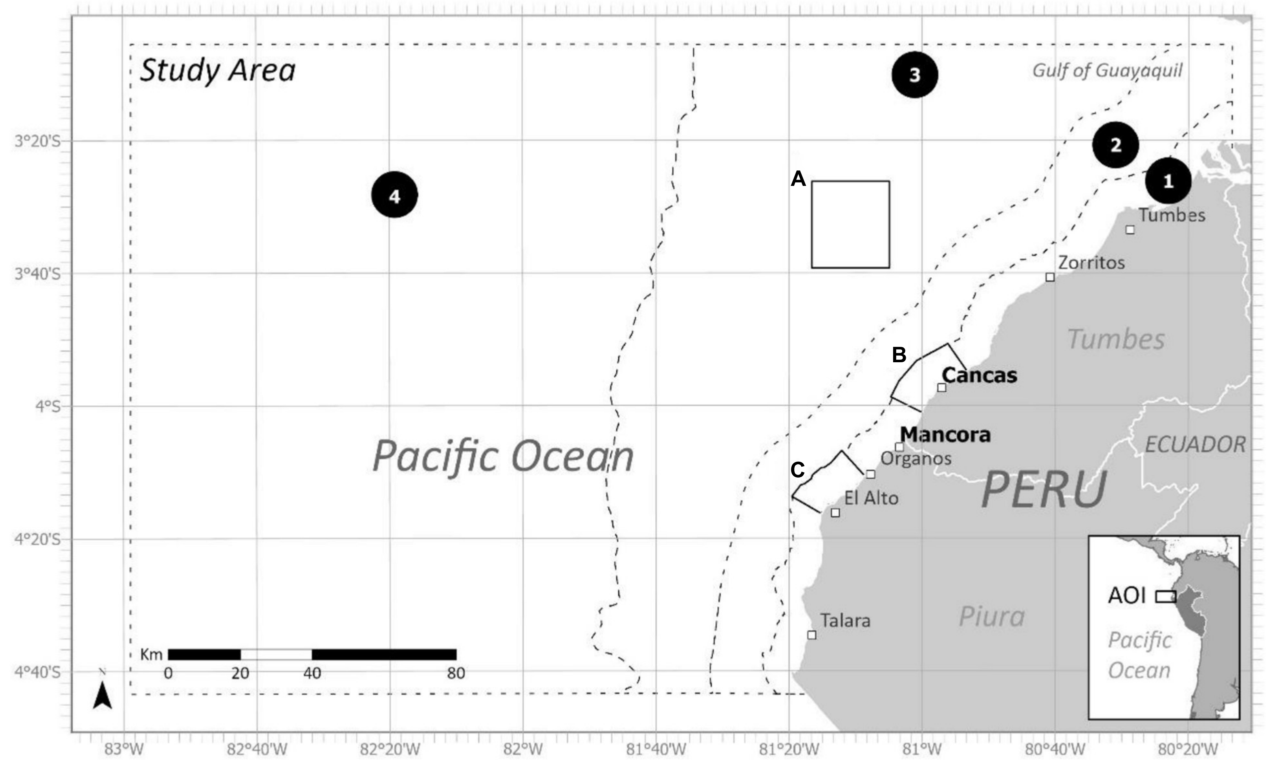


FIGURE 2 | Northern Peruvian coastline and the two fishing ports where interviews were conducted- Mancora and Cancas. Map is divided into study area and seven additional subzones. Zone (1) delineates the 5 nm artisanal fisheries boundary, Zone (2) delineates the 12 nm territorial seas boundary, Zone (3) delineates the cut-off for the bathymetric shelf, and Zone (4) contains the waters beyond the shelf. The second group of subzones were the boundaries of three proposed Marine Protected Areas including (A) Banco de Mancora, (B) Arrecifes de Punta Sal, and (C) Cabo Blanco–El Ñuro.

and 21,943 small-scale fishers operating out of Tumbes and Piura regions, respectively, with estimated numbers increasing over the last near decade (Castillo et al., 2018).

Initial Interviews and Workshop

A total of 85 semi-structured, one-on-one, GIS-based interviews were conducted by our team between July and August of 2019 in two ports, Mancora (n = 55) and Cancas (n = 30) (**Figure 2**). The questionnaire was based on Pilcher et al. (2017) and translated into Spanish. The questionnaire contained 72 questions that covered topics of basic demographics, fisheries data, previous animal sightings, vessel-animal interactions, and conservation perceptions (**Supplementary Appendix A**). For the PGIS part of the interview, using paper maps of the study area (**Supplementary Appendix B**), we requested that respondents draw polygons for fishing areas they used, and HBW and LBT habitat areas, and points for animal sightings and locations where they had entangled or seen entangled HBW or LBT. We first asked the fishers to mark the map where they sighted HBW and LBT from the previous year, previous 5 years, and then any remaining information from the fisher's lifetime. A table was used to record attributes of the point data (e.g., date, number of individuals, animal condition). We shared a base map with several local geographic references to help guide the fishers, including bathymetry contours of 200 m and names of coastal towns (**Supplementary Appendix B**). We documented and grouped fishing gear into a total of six common gear type categories: (1) gillnet surface, (2) gillnet bottom, (3) longline surface, (4) longline bottom, (5) purse seine, and (6) hook and line.

We used both purposive and snow-ball sampling methods for finding interview participants (Goodman, 1961; Denzin and Lincoln, 2018). Key contacts, who were either well-connected fishers, respected port leaders, or local officials, assisted in introducing us to interview participants. With their help, we specifically sought out captains to avoid duplicate information from individuals on the same vessel (83 and 76% of participants were captains from Mancora and Cancas, respectively). Additionally, we believed captains would be more familiar with, and knowledgeable of our study area due to their longer years of fishing experience. We sought to interview one third of vessels at each port with a focus on the gear types that have been documented previously in local bycatch events (longlines and gill nets). We estimated total vessels for both ports by conducting shore counts of vessels docked at the harbors (Mancora = ∼200, Cancas = ∼100). For our research, the sample group attained was meant for a qualitative analysis of the fishers at each port as well as attain key information regarding fishing activity. Two of the interview participants were not active fishers, but rather worked in the local whale-watching and tourism industry and only offered data on animal sightings. Interviews took under an hour to complete, were conducted in Spanish, and then translated into English for analysis. Each interview began with a statement describing the project goals, and our emphasis on using information for research purposes only. We also explained that the data collected would not be shared with other entities and would be used for research purposes only. This research protocol was approved by the San Francisco State University Institutional Review Board for Human-Subject research on June 20, 2019, Protocol number X19-30.

Interviews were divided into species-specific sections. We used a skip pattern where if a participant stated they never saw a humpback whale (Q17) or a leatherback turtle (Q47), we would end that section of the interview and continue to the next section or go to the final question (Q71). All interview responses were recorded using physical copies of the questionnaire where answers were marked, though three interviews were audio recorded with permission from fishers to recall details later for analysis. Additionally, in August 2019, we hosted a mapping workshop with fishers from Mancora. The goal of the meeting was to review the study area maps as a whole and have an open discussion on the solutions and concerns of bycatch.

Analysis of Interviews and Participatory Maps

All interview responses were transcribed into a spreadsheet, then coded to identify emergent themes from the qualitative responses (Saldanþa, 2009). Participatory maps from interviews were photographed and imported into a GIS where map images were georeferenced and overlaid atop a base map in the GIS. Hand drawn polygons and point features were digitized in order to transfer information as accurately as possible. Associated table data were attributed to each polygon (e.g., gear characteristics, months fished, target species, etc.) and animal sightings point data (e.g., date and time observed, number and condition of individuals, etc.) (**Supplementary Appendix B**). Fishery polygons were divided among two seasons: austral winter and austral summer, based on fisher input and oceanic conditions (Pennington et al., 2006; Bakun and Weeks, 2008). We recognize that seasonal transitions exist between winter and summer, but for simplicity in describing oceanic conditions and the regional peak whale season, we chose to capture two temporal scenarios. Winter season was defined as June through November, and summer season as December through May. Participants who declared they fished all year were attributed to both summer and winter groupings.

After digitizing the participatory paper maps, all category specific polygons were combined into groups. The groups were: (1) fishing areas by port, season, and gear type (example: Mancora, winter, longline surface) and (2) fisher-perceived habitat of humpback whales and leatherback turtles. We used a count of overlapping polygons on the groups to create density maps of all group specific overlapping polygons. These maps roughly identified where the fishers perceived the greatest fishing density area and the spatial extent of the study species' habitat within the study area (Appendices C, D). The second group of maps consisted of previous animal sightings as point data. These data were run through a kernel density estimation (KDE) that interpolates a surface by estimating the spatial extent and density of species presence (Kenchington et al., 2014) (**Supplementary Appendix E**). Species maps were printed in large poster format for group review, whereas gear maps were printed on standard letter size for individual reviewers.

Map Review Workshops

Researchers returned to northern Peru for mid-project map review workshops on the 22nd and 23rd of January 2020 in Cancas and Mancora, respectively. The goal of the workshops was to present data collection results and receive feedback on combined data input for corrections and clarifications into the ByRA model. Specific feedback focused on winter and summer season divisions, accuracy of gear use locations, and accuracy of species sightings and habitat.

In the workshops we presented maps containing all of the spatial data gathered from initial interviews. Participants were encouraged to invite other fishers to gather as much feedback as possible in the workshop, regardless of whether they had previously participated in the initial interviews conducted months before. For Cancas, both returning (3) and new participants (15) attended the workshop. We asked the fishers whether the maps generally reflected where they see the species. If not, they were asked to mark on the map the correct locations. The maps were marked using a dry-erase marker as appropriate or were noted OK if approved by participants. After this, we distributed individual, gear-specific paper maps to each participant depending on the type of fishing gear they used. Each paper showed a fisheries density map on one side and two questions on the back asking if the map reflects their own and other fisher's fishing areas, with an area to elaborate on their response. Papers were collected and photographed to be transcribed and later incorporated to revise or reaffirm input into the ByRA.

We hosted two smaller map reviews in Mancora. Three fishers and one whale-watching employee attended the map review. Four additional fishers were also identified and participated in a review on a fishing boat. The same questions were asked, and paper maps distributed, reviewed, marked, and re-collected to be photographed and transcribed.

Building Habitat Models

Sightings data gathered from our PGIS exercises were then used as input into presence-only models to determine the distributions of our study animals. We chose a maximum entropy likelihood model, MaxEnt, to model habitat preferences from a combination of environmental variables and known species occurrences (Phillips et al., 2017). Maxent is one of the most prevalent models of choice among ecologists due to its simple user interface, predictive power, and presence-only inputs. Maxent accepts smaller sample sizes at a minimum of 30 occurrence points to produce outputs with acceptable statistical power (Tobeña et al., 2016).

A total of 215 previous animal sighting points were collected for humpback whales, and 79 for leatherback turtles. We removed sighting points that had no specified month or year. Additional sighting points were removed that came from surveys where the interviewer did not feel confident in the fisher's ability to distinguish among the species reported (Question 80, $n = 5$). To remove spatial autocorrelation and reduce sample bias, we used the SDMtoolbox v2.0 to spatially rarefy the presence points with a 5 km buffer for humpbacks and a 2 km buffer for leatherbacks (Kramer-Schadt et al., 2013; Brown et al., 2017). These values were chosen based on the size of the study area, the distribution of the animal presence points, and the scale of which the species' have occupied space in the area (Guidino et al., 2014; Hoover et al., 2019). Due to a recommended minimum number of 30 occurrence points to show any reasonable statistical power within MaxEnt, we were unable to model either humpback habitat for the austral summer ($n = 27$), or leatherback habitat for austral winter ($n = 17$) (Tobeña et al., 2016). Total presence points used for humpback whales were reduced to 75 for the austral winter months (i.e., local whale season), and 35 presence points for leatherback turtles for the austral summer.

A total of eight candidate environmental variables were selected based on previous animal habitat suitability studies and features of ecological importance which have influence on physical processes that may signal safe havens for the species or promote prey availability (Fiedler et al., 2018). The eight included: sea surface temperature (sst), bathymetry (bathy), chlorophyll *a* (chlora), and k490 coefficient (k490), as well as GIS derived variables: Euclidean distance to bathymetric shelf break (distShelf), distance to shore (dist0), and distance to 200 m isobath (dist200) (Dransfield et al., 2014; Derville et al., 2018; Hoover et al., 2019; **Table 2**). All variables were plotted on a 1.67 km^2 grid of cells, the smallest cell size of the covariate data sources. This spatial scale helped capture the high resolution of fisher-drawn polygons to aid in management recommendations and the practical use of map outputs. Monthly averages of covariates were then averaged for each seasonal range, resulting in eight layers for each season with the same spatial extent, cell size, and coordinate system. Values were extracted for each presence sighting point from appropriate months and years, then were modeled against randomized background points as pseudo-absences in the MaxEnt program. Final distribution models were assessed using two metrics: (1) area under the receiver operating characteristic curve (AUC ROC), and (2) a true skills statistic (TSS) test, in addition to visual comparisons between the Maxent output maps if AUC and TSS were comparable between the final models. Model selection and evaluation is described further in **Supplementary Material** under Species Distribution Models.

Bycatch Risk Assessment

Assessing Risk to Species

The Bycatch Risk Assessment (ByRA) model was adapted from the Habitat Risk Assessment tool (version 3.8.9) made freely available from the Stanford Natural Capital Project InVEST software suite.[1] The overall function of the ByRA model is to identify and assess bycatch risk, built on the combination of spatially explicit geospatial layers of both species' distribution and fishing areas (Hines et al., 2020; Verutes et al., 2020). Within the ByRA model, the software requires following a template to add direct numerical input by the user to give objective weights to each category of gear type and their impact on a species (**Table 1**). These weights are incorporated into ByRA where cumulative risk to a species is calculated from an exposure and consequence matrix (Stephenson et al., 2020). Exposure is the degree a species experiences stress, injury, or mortality due to an anthropogenic source or activity, which, in this case is exposure to fishing gear (Samhouri and Levin, 2012). For our research, exposure was measured by seven specific risk factors that contribute to fisheries bycatch, of which four relied on spatially explicit criteria (SEC) data. Rather than a numerical input, the input in the table for the SEC required a file path to a preformatted GIS layer depicting the criteria. These four SEC were (1) spatial overlap between the species and stressors, (2) likelihood of animal interaction with gear type based on the output of the species distribution layers and it's overlap with fishing density, (3) temporal overlap of gear soak time, and (4) intensity of fishing area used as a proxy for the number of vessels or gear in the water at a given time.

Consequence is the impact to a species from a possible interaction with fishing gear. Consequence criteria contain two subcategories: sensitivity and resilience (Sharp et al., 2019). Respectively, these categories provide an opportunity to include information on the vulnerability, population dynamics, and health of a species, as well as the traits of a species or population that would encourage a recovery from a fishing interaction (Hobday et al., 2011). Criteria for species-specific *sensitivity* to a stressor includes: (1) life stages affected bycatch, and (2) the severity of the possible interaction. Criteria for species-specific *resilience* to a stressor includes: (1) age of maturity, (2) reproductive strategy, (3) population connectivity, and (4) local conservation status (**Supplementary Tables 3**, **4**).

Expert Literature Input

Following the criteria definitions and rating scheme provided by ByRA and previously applied by Hines et al. (2020) and Verutes et al. (2020), all collaborators discussed and assigned scores for each category and binned gear type. Final scores were reviewed among the research team on the basis of their expert knowledge, interview responses, field observations, and previous literature. The ratings given ranged between 1 (low risk) to 3 (high risk) or marked as 0 if a matrix score was unavailable or not applicable (**Supplementary Tables 3–5**).

Preparing Bycatch Risk Assessment Data Input

Fishing area polygons were prepared for each geospatial fishing area layer (Verutes et al., 2020). If the information exists, the ByRA tool allows users to create spatially explicit criteria (SEC) scores that vary throughout the study area and for each species-gear interaction. For our research we incorporated three SEC: temporal overlap of gear soak time, intensity of fishing area, and likelihood of species/gear interaction.

For temporal overlap of gear soak time, 49% ($n = 39$) of the gillnet fishing areas identified from the interviews had distinguishable variation on length of soak time. Longer soak times, alluding to overnight or half day sets, may create higher

---

[1] naturalcapitalproject.stanford.edu/software/invest

TABLE 2 | Candidate environmental variables used in model selection process.

Predictor	Description	Unit	Data type	Resolution	Source
sst	Sea surface temperature	C^{o}	climatology	4 km^2	National Oceanic and Atmospheric Administration (NOAA) MODIS (https://oceandata.sci.gsfc.nasa.gov/MODIS-Aqua/)
chlora	Chlorophyll-*a*	mgm^{-3}	climatology	4 km^2	National Oceanic and Atmospheric Administration (NOAA) MODIS (https://oceandata.sci.gsfc.nasa.gov/MODIS-Aqua/)
kd490	KD 490 diffuse attenuation coefficient	m^{-1}	climatology	4 km^2	National Oceanic and Atmospheric Administration (NOAA) MODIS (https://oceandata.sci.gsfc.nasa.gov/MODIS-Aqua/)
bathy	Bathymetry	M	topographic	1.67 m^2	NASA Earth Observations (NEO), GEBCO (https://neo.sci.gsfc.nasa.gov/view.php?datasetId=GEBCO_BATHY)
dist0	Euclidean distance to shore	m	Derived topographic	1.67 m^2	Interpolated in GIS
distShelf	Euclidean distance to shelf edge	m	Derived topographic	1.67 m^2	Interpolated in GIS
dist200	Euclidean distance to 200 m isobath	m	Derived topographic	1.67 m^2	Interpolated in GIS
slope	Slope derived from bathymetry	–	Derived topographic	1.67 m^2	Interpolated in GIS

risk for an animal interaction (Shester and Micheli, 2011). When we recorded a soak time of 6 h or more for gillnet fishing polygons, a rating score of 3 was assigned to the *rating* field, though those with soak times between 3 and 6 h were scored a 2 and time less than 3 h scored a 1 and unknown a zero.

For the intensity criteria, we generated the GIS layer by creating individual polygons for each intersecting fisher-drawn polygon and recorded as attributes the number of overlapping polygons in each polygon. For example, in areas where there are ten overlapping fishing areas, a newly created polygon delineated from the mutual overlap was given the value 10 in the *rating* field of the gear-specific GIS layer. Values were reclassified into three categories using the classification algorithm Natural Breaks, which divides similar values together and maximizes the differences between the groups, to fit the risk value schema (1–3). More popular or dense fishing areas were then captured in the model as a proxy for determining higher density areas of gear use.

For the third SEC for likelihood of species/gear interaction, we overlaid the categorized species distribution and *intensity* SEC layers. The resulting encounter rates were scored in the *rating* field and were classified into three categories similarly as above. These SEC and related ratings helped differentiate specific fishing areas and identify possible higher bycatch risk zones.

Bycatch Risk Assessment was run a total of six times. We ran the model for both species using all fishing areas, then we divided the fishing areas by port and ran the models again. Two groupings of subzones were created to assess risk (**Figure 2**). These groupings were chosen to give practical applications of risk outputs based on distance to shore and potential regional protected areas. The first grouping of subzones included four zones delineated by the 5 nm artisanal fisheries boundary (1), the 12 nm territorial seas boundary (2), the cut-off for the bathymetric shelf (3) and the waters beyond the shelf (4). The second group of subzones were the boundaries of three proposed Marine Protected Areas including Banco de Mancora (A), Arrecifes de Punta Sal (B), and Cabo Blanco–El Ñuro (C). Risk percentages for each gear type within each subzone were calculated and classified as low, medium, and high risk (**Figures 3A**, **4**, **5A**, **6**). These percentages were determined by the combination of the spatial input data (i.e., fishing areas, and SDM layers) combined with the rating scores, associated weights and data quality ratings assigned by our team in the exposure and consequence tables. As part of the ByRA model outputs, exposure and consequence (ExC) scores were also estimated and plotted for each gear and species combination (**Figures 3B**, **5B**).

Characterizing Data Uncertainties and Importance

To address the various sources of data availability and quality used to support the model, we incorporated uncertainty metrics and importance weights for each data source and category bin in the exposure and consequence table. Transparency in data quality within risk assessments is important for stakeholder awareness so model outputs are perceived appropriately and data gaps can be clearly defined for future research (Harwood, 2000). A column specifically for scoring data quality was presented to the user of the ByRA model to allocate quality of data source and importance of criteria used to substantiate the risk scores given. Additionally, data input was characterized visually as part of the final risk maps. Data inputs were given a color scoring of green, yellow or red for four categories of our data: (1) animal sightings distribution, (2) habitat suitability, (3) fishing occurrence/gear type densities, and (4) bycatch/stranding data (**Table 3**; Hines et al., 2020). Among these categories, inputs were tagged green if they contained substantial data, yellow for limited data availability, and red for unknown or incomplete data. Stoplight figures were incorporated in the maps. By incorporating uncertainties regarding inputs, the model outputs offered more realistic perceptions of our findings and provided quick visual cues for communicating data standards with stakeholders.

RESULTS

Fishing Areas

During the PGIS section of the interviews, 85 interview respondents drew a total of 104 polygons that represented their fishing grounds. This total area spanned 1 degree 30’S of latitude (3 degrees 15’- 4 degrees 44’S) and 2 degrees 10’W of longitude

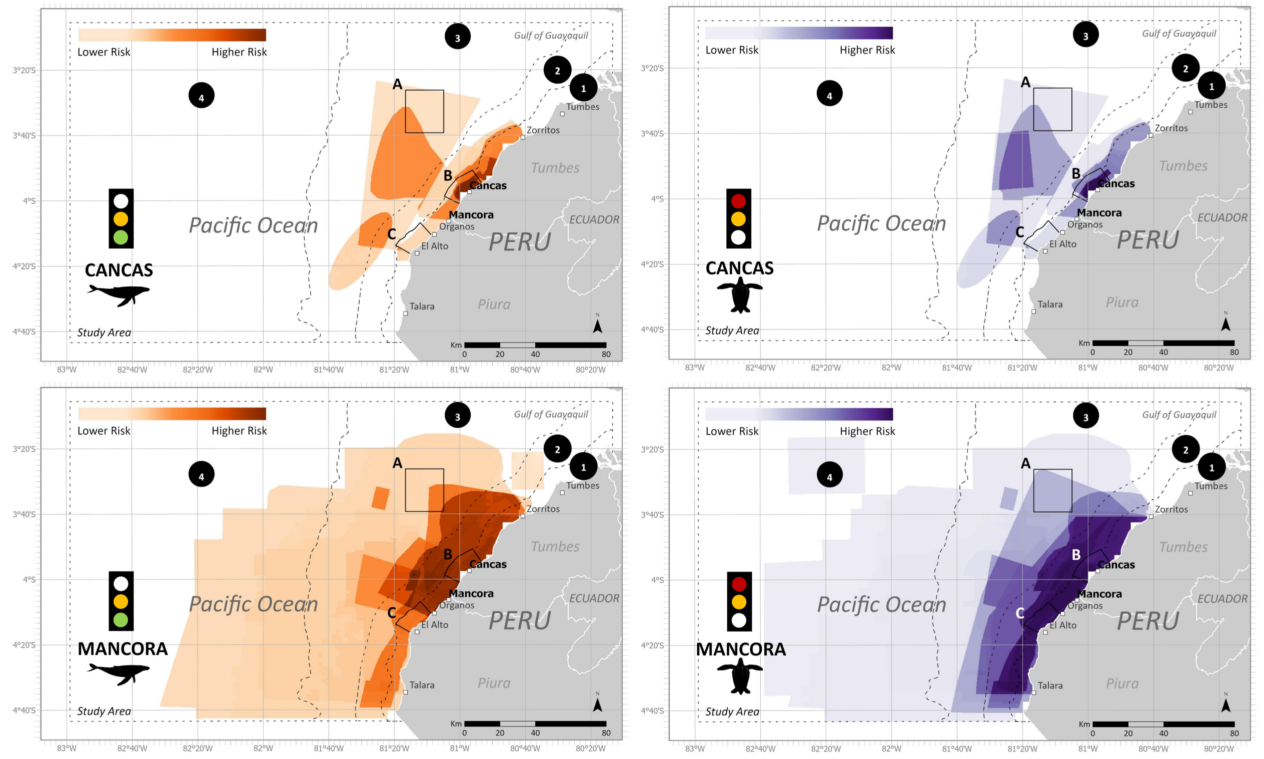


FIGURE 3 | Final estimated bycatch risk split by ports (Mancora and Cancas) for humpback whales and leatherback turtles during the austral winter (June-November) and austral summer (December-May), respectively. Subzones include (1) Artisanal fishing zone (5 nm), (2) Territorial seas zone (12 nm), (3) Shelf zone, (4) Beyond shelf zone, (A) Banco de Mancora, (B) Arrecifes de Punta Sal, and (C) Cabo Blanco–El Ñuro. Data uncertainty depicted in stoplight and described further in **Table 3**.

(82 degrees 45’W–80 degrees 35’W) covering an overall area of 43,000 km^2 (**Figure 2**). Fishing grounds for all gear types covered an area of ∼7,000 km^2 for Cancas and ∼37,000 km^2 for Mancora. The greatest overlap of fishing polygons was for the surface gillnets, for a total of 25 overlapping polygons during the austral summer season (**Supplementary Appendix D**). The distance to the center of the densest fishing area from shore was 32 km from the port of Mancora and 3 km from the port in Cancas. Most participants identified their fishing behavior as year-round (63%), with 15% exclusively fishing during the winter months and 23% during the summer months.

Fisher Demographics

Fisher demographics between Mancora and Cancas varied in several characteristics, although all fishers were men. Our sample group in Mancora on average fished in larger vessels (mean vessel length = 10 meters), employed larger crew sizes (mean = 5 fishers), and were younger (mean age = 43), compared to Cancas (mean vessel length = 6 meters, crew size = 3 fishers, and mean age = 53). The fishers we interviewed ranged from 25 to 72 years of age. The number of years reported as working in the fisheries trade varied from 2 to 58 years (Mancora mean = 21 years, Cancas mean = 34 years). Due to complex fishing schedules and limited access to all fishers, we are aware that representative samples for extrapolation of fisher responses were not attainable. Rather, sample group responses and results portray a qualitative analysis of the fishers at each port.

Conservation Perceptions

We asked several questions regarding conservation perceptions during the one-on-one interviews to gauge fishers’ understanding of the population growth or declines over time and attitudes toward HBW and LBT populations (**Supplementary Appendix A**). For fisher’s who had been fishing for more than 35 years, 85% (n = 20) of respondents stated that there were either more or the same number of humpback whales in the area. This was the same percentage for those who had been fishing for <15 years (n = 20), and 91.5% for folks who had been fishing between 15 and 35 years (n = 40). Comments included legal protections, prohibitions on hunting, and access to anchovies as prey. Two-thirds of the respondents (n = 58) stated there will always be humpback whales off the coast of Peru. Of these, 18.4% (n = 16) of respondents believed this was due to fishers not hunting humpbacks, 16% (n = 14) of respondents cited humpback migrations, 7% (n = 6) mentioned that they reproduce, and only 5% (n = 4) of respondents gave a scenario stating humpback whale populations would only be sustainable if they were protected. Sixty percent (n = 57) of respondents agreed that humpback whales were important to very important for the ocean. However, 18.6% (n = 16) stated humpbacks

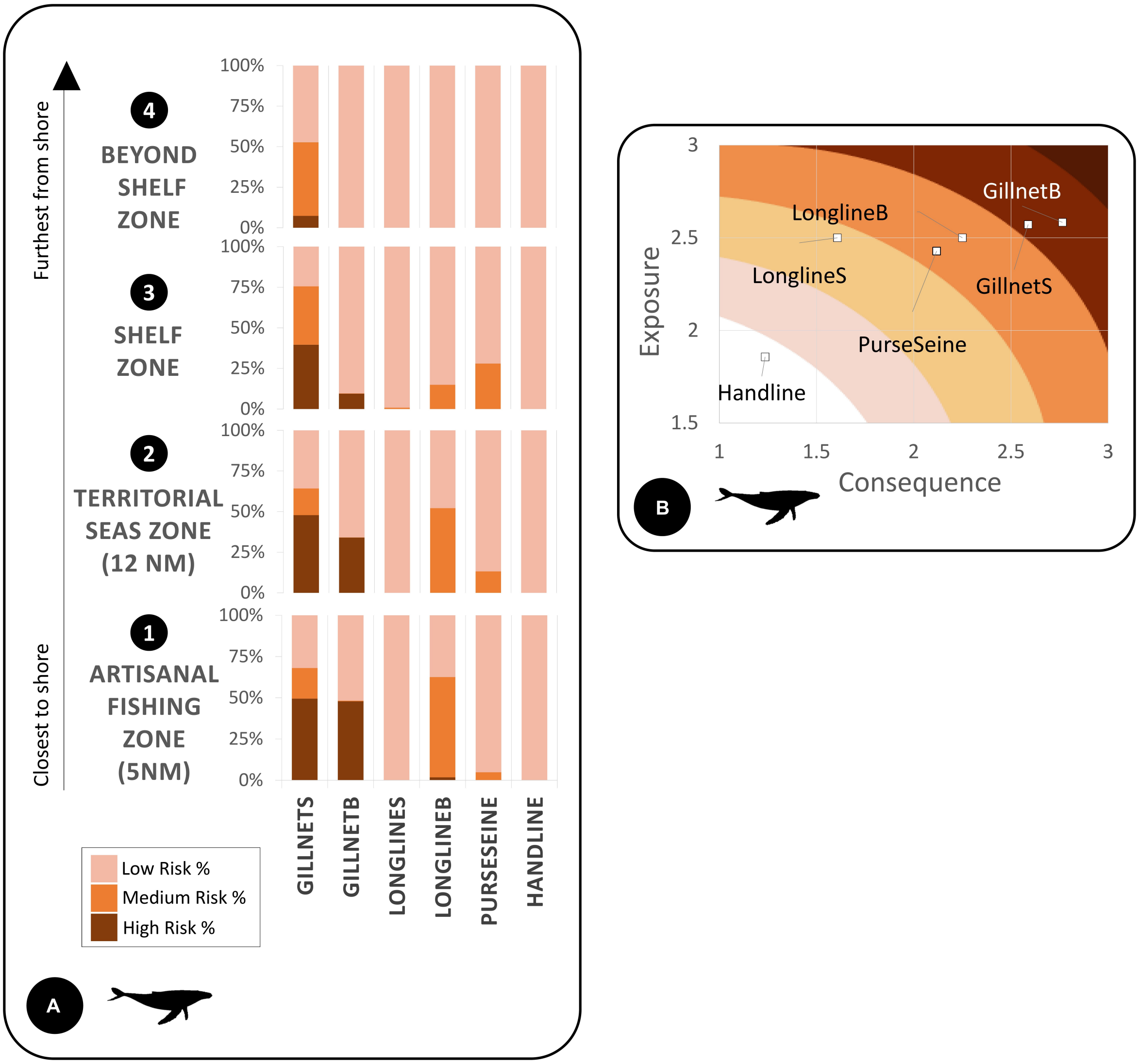


FIGURE 4 | (A) We calculated humpback whale bycatch risk percentages for each subzone by gear type divided into low, medium, and high risk. **(B)** Exposure and consequence plots depict gear impact on species based on the spatial exposure or consequence criteria ratings (**Supplementary Material**).

were not important to the ocean, because whales damage their fishing gear or have no function in the ocean. Those that did believe humpbacks were important, mentioned that the ocean is the species' natural habitat, and that the whales support local tourism (**Table 4**).

As for the 67 respondents who stated they had seen a leatherback turtle in their life, those who had been fishing for more than 35 years, 50% (*n* = 6) believed leatherback turtle numbers have decreased or remained about the same with four individuals unsure. Overall, respondents were split between whether they believed leatherback turtle populations were increasing (*n* = 6), decreasing (*n* = 12), or were unsure (*n* = 11). Seventy-six percent (*n* = 51) of respondents believed that having leatherback turtles in the ocean is important (Q 68) (**Supplementary Appendix A**). When asked why, the fishermen mentioned the species' role in the ecosystem and food chain, tourism, and their beauty (**Table 4**). For both Cancas and Mancora, 36.8% of respondents (*n* = 32) stated there would always be leatherback turtles in the ocean, because of legal protections, no hunting, and prey availability. Only two fishers believed there would be a time when there would be no leatherbacks in the ocean, stating that they were being found further offshore, and are nearing extinction.

Synthesizing Fisher Knowledge and Perspectives

Fisher participation throughout different points of the research were key in providing a two-way conversation with the participants, allowing us to prepare the input data and present the outputs of the risk assessment more accurately. Both interviews and mapping workshops offered an opportunity for fishers to

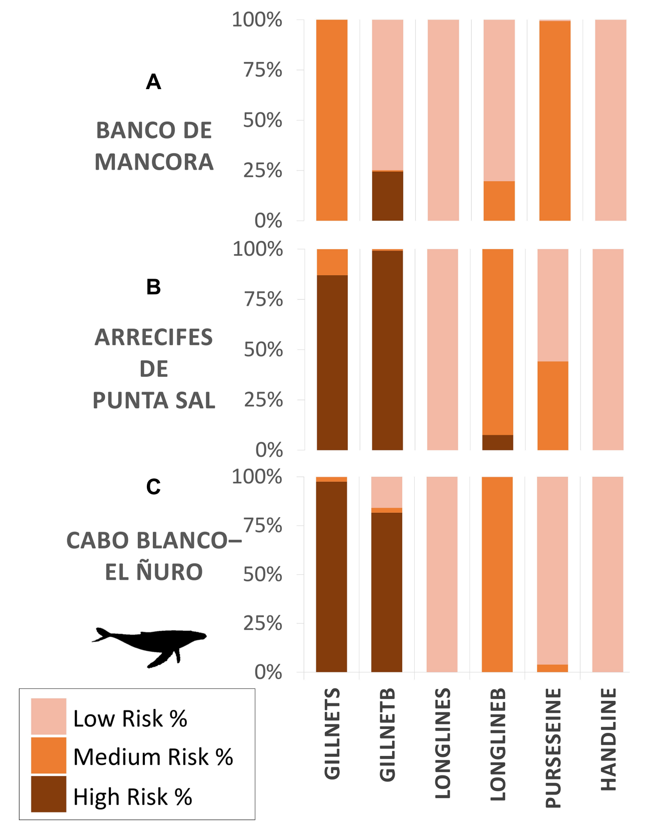


FIGURE 5 | Bycatch risk percentages calculated for humpback whales for each gear type. Subzones assessed are composed of proposed marine protected areas: **(A)** Banco de Mancora, **(B)** Arrecifes de Punta Sal, and **(C)** Cabo Blanco–El Ñuro, as part of the Grau Tropical Marine Reserve.

express their concerns and experiences with bycatch in a one-on-one and group setting. For example, at the first workshop, our research team presented on HBW and LBT population statuses, migrations, and the benefits of the species to the ecosystem. In response, fishers attending the meeting were able to share their knowledge on the species presence in the area via group mapping, their concerns about gear loss from entanglements, and talked about possible solutions to bycatch such as avoiding high density areas of HBW during the season, switching gear types or using bycatch reduction technologies such as acoustic pingers. For our second trip to the ports, fishers helped review the maps of fishing area and habitat layers. Fishers emphasized data that reflected the use of the area and of their peers, and in contrast were able to reject data that did not align with their experience and knowledge of the area. In our final meeting, after a year we initiated the project, we realized that the majority of the fishers had already decided to change gear from surface gillnets to longline, and bottom set nets.

With these group meetings fishers provided valuable feedback, so the input data into ByRA reflected closer the reality of their experiences. Additionally, hearing the fishers' perspectives on the severity and high cost of losing their gear (estimated at $300 per pane of net) helped us synthesize the ByRA outputs and management recommendations knowing what changes they would be willing to make to reduce unwanted interactions (Alfaro-Shigueto et al., 2010). Because of these meetings, fishers expressed that they felt valued and that their opinions on management issues mattered. Fishers agreed that more formal conversations between the fishing associations, port administration and each other within a participatory framework could galvanize interest and initiate collaborative ideas in how to mitigate bycatch.

Species Distribution Model

Final contributing environmental covariates from greatest to least contributing were Euclidean distance to shore (dist0) (*PI* = 72.7%), chlorophyll-*a* (chlora) (*PI* = 16.3%), and slope (*PI* = 11%) (**Table 5**). Predicted habitat was close to the coast within 50 km and above the shelf break (**Figure 2**). The final SDM results for the two species differed in parameter settings and most environmental variables. In the austral winter SDM for humpback whales, the final Maxent model used features Linear, Quadratic and Hinge (LQH), with a 1.5 regularization parameter, and showed predictive performance with an AUC of 0.88 and a TSS of 0.89 (**Table 6**).

The greatest variable contributor for the leatherback SDM was distance to 200 m depth (dist200) (*PI* = 72.7%), following chlorophyll-*a* (chloral) (*PI* = 21.8%) and Euclidean distance to shelf (dist0) (*PI* = 5.5%) (**Table 5**). For leatherback turtles, the best SDM for the summer season used only the Hinge feature within Maxent, with a 2.5 regularization parameter and scoring similar performance metrics with an AUC of 0.86 and a TSS score of 0.85 (**Table 6**). Leatherback turtle habitat preference showed highest within a 200 m depth, which includes the elevated seamount area known as Banco de Mancora. Both SDMs showed possible evidence of sampling bias with the highest predicted habitat preference nearest to the study sites (**Figure 7**).

Uncertainty Scores

We determined separate uncertainty scores for each category and species based on standards from Hines et al. (2020). Red represents high uncertainty, yellow for medium and green for low. All species sightings data was retrieved from interviews (as opposed to formal transect surveys) which we labeled a blend of yellow and red for HBW and LBT. From these sightings, we were able to construct habitat suitability models using a collection of environmental variables to predict high, medium, and low habitat preferences. Because of the fewer sightings gathered for LBT, we labeled the output risk maps with a stronger uncertainty (labeled yellow and red). Fishing areas were also gathered from our PGIS exercise, which incorporated temporal and spatial variations in gear use. From these we were able to estimate fishing area densities and therefore labeled this criterion as a blend of green and yellow. Although we gathered information on previous bycatch incidents from the fisher interviews, we were not able to give estimates on bycatch rate for either species or ports. Due to this, our data quality on bycatch was labeled a blend of green and yellow for HBW. For LBT, interviewed reported bycatch was gathered from fewer fishers which we labeled a blend of yellow and red to signal greater uncertainty.

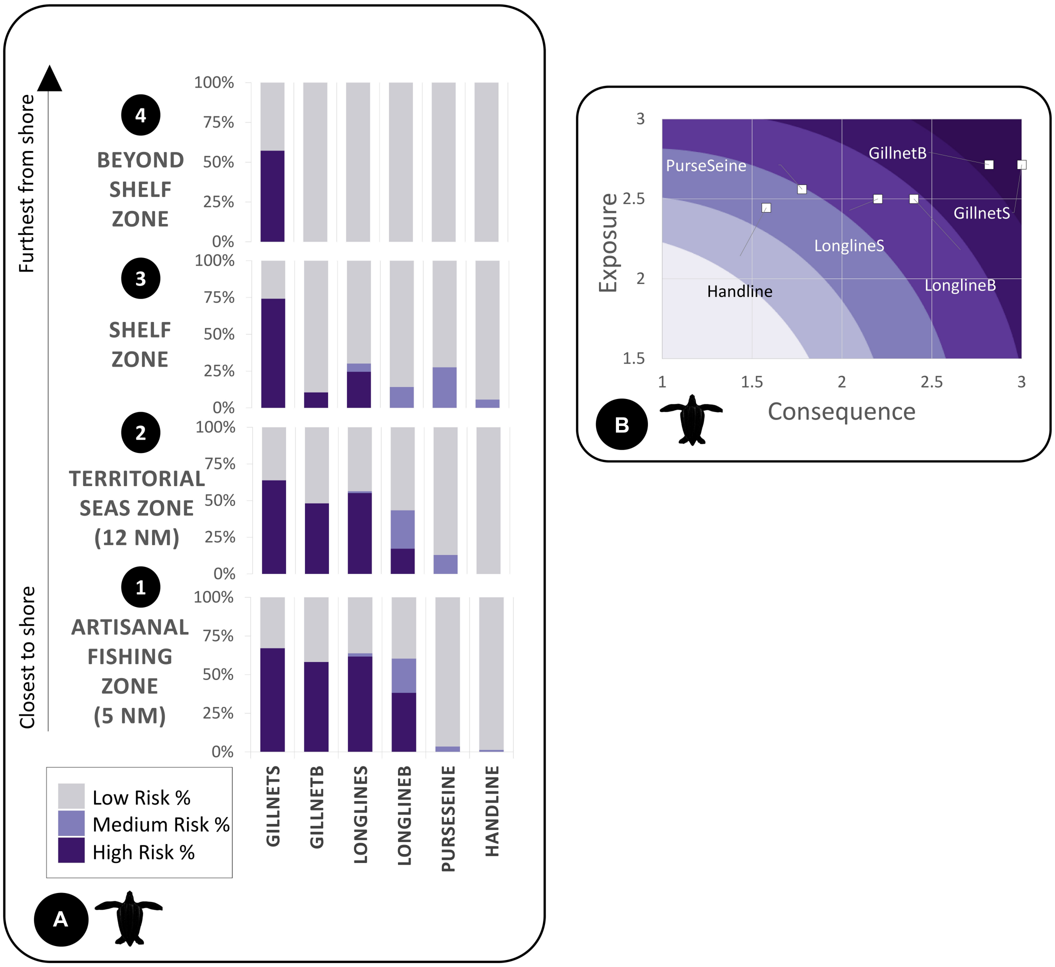
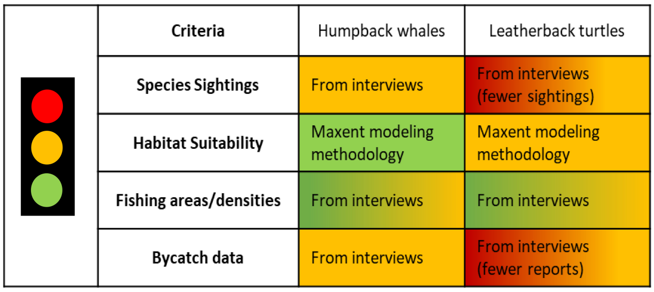


FIGURE 6 | **(A)** We calculated bycatch risk percentages for leatherback turtles for each subzone by gear type divided into low, medium, and high risk. **(B)** Exposure and consequence plots depict gear impact on species based on the spatial exposure or consequence criteria ratings (**Supplementary Material**).

TABLE 3 | We assigned uncertainty signals using the stoplight approach from Hines et al. (2020).



	Criteria	Humpback whales	Leatherback turtles
	Species Sightings	From interviews	From interviews (fewer sightings)
	Habitat Suitability	Maxent modeling methodology	Maxent modeling methodology
	Fishing areas/densities	From interviews	From interviews
	Bycatch data	From interviews	From interviews (fewer reports)

Coloration signals data quality and uncertainty in final outputs.

All uncertainty criteria were considered for stoplight labels shown in the final bycatch risk maps (**Figures 3, 8, 9**). For the HBW bycatch maps, we labeled the overall data uncertainty a green and yellow label. For LBT, our overall uncertainty score was yellow and red.

Bycatch Risk Estimates

The ByRA outputs are risk plots and a series of GIS map layers classified by the modeled amount of bycatch risk (**Figures 3, 8, 9**). Spatially assessed bycatch risk followed patterns associated with high habitat suitability of the study species and where there was high overlap among the seven fishing gear stressors and the spatially explicit criteria (SEC) (likelihood of interaction with species, intensity, and temporal soak time). The HBW risk map showed that the greatest risk was centralized off the coast of Mancora and Cancas, and medium to lower risk further from shore (**Figure 8**). The highest bycatch risk areas for LBT were present in zones 1, 2, and 3, predominantly centered off the coast between the towns of Cancas and El Alto (**Figure 9**).

Model outputs that were split by port showed stark differences in the spatial distribution of risk between Cancas and Mancora (**Figure 3**). For both species, the Cancas risk maps had a smaller range of risk closer to the shore (**Figure 3**). The risk outputs for Mancora cover a larger region with highest risk spanning a larger coastal area.

TABLE 4 | Summary of responses to questions 40 and 68 on the perceived importance of species.

Question *Do you think having (species) in the ocean is (or I don't know)*	**Sample responses of explanations**	
	Humpback whale	**Leatherback turtle**
1. Not important	Lost and damaged gear; inconvenience; whales have no function	They aren't usable
2. Slightly important	Not good for fishing; tourism; harms fishermen and their boats; possibility for entanglements	*The ocean is their habitat*
3. Important	The ocean in their home; tourism; part of the ecosystem	They are beautiful; the ocean is their habitat/home; they clean the ocean; they maintain biodiversity; tourism
4. Fairly important	The ocean is their habitat; they are innocent	People eat them; they are part of the ecosystem; they clean the ocean; they signal where there are fish
5. Very important	They are part of the food chain; Feces is food for the fish; they scare the sea lions; tourism	They are part of the ecosystem; part of the food chain; they eat the bad fish/waste; tourism

TABLE 5 | Selected environmental variables after pruning the covariates (**Supplementary Material**) as well as permutation importance used for final species distribution models.

Species	Presence (n)	Season	chlora[a]	dist0[b]	distShelf[c]	dist200[d]	Slope[e]
Humpback whale	75	Winter*	16.3	72.7	–	–	11
Leatherback turtle	35	Summer*	21.8	–	5.5	72.7	–

(–) indicates variable was not used in final model.
**Winter season (June–November), summer season (December–May).*
[a] *Chlorophyll-a.*
[b] *Euclidean distance to shore.*
[c] *Euclidean distance to shelf edge.*
[d] *Euclidean distance to 200 m isobath.*
[e] *Slope derived from bathymetry.*

TABLE 6 | Model parameters and performance statistics of final models calculated from biomod2 package for R.

Species	Presence (n)	Season	Training AUC	Test AUC mean (standard deviation)	TSS (mean)	Regularization parameter	Features
Humpback whale	75	Winter*	0.88	0.87 (0.06)	0.89	1.5	LQH
Leatherback turtle	35	Summer*	0.86	0.87 (0.05)	0.85	2.5	H

**Winter season (June–November), summer season (December–May).*

For the exposure and consequence (ExC) scores, points plotted higher along the x-axis (consequence) and y-axis (exposure) posed the greatest risk to the species (**Figures 4B**, **5B**). For example, points that had high exposure but lower consequence from a species/gear occurrence were plotted more in the left side of the plot (i.e., handline). Overall, gillnets (surface and bottom) were the riskiest gear type for both species within the ExC plots with similar exposure scores and very small differences in consequence. Bottom longlines were second in overall risk (**Figures 4B**, **5B**). Handlines showed the lowest risk in the ExC plots for both species, in all subzones and the overall study area.

Numbered Zones

Following the spatial distribution of fishing grounds reported by fishers, bycatch risk for most gear types decreased with distance from the coast. For the LBT risk map, there were large proportions of higher risk areas within all subzones for each gear category (**Figure 6A**). For the HBW risk map, out of the four zones, the largest percentage of highest risk was for bottom gillnets in zone A (**Figure 4A**).

Proposed Marine Protected Areas

Within the three proposed MPA boundaries, the highest bycatch risk for both species occurred in zone B, Arrecifes de Punta, with the highest risk being associated to surface set gillnets (HBW 87%, LBT 99%), bottom set gillnets (HBW 99%, LBT 99%) and bottom set longlines (HBW 8%, LBT 99%), and additionally surface set longlines (99%) solely for LBT (**Figures 5**, **10**). Closely following was zone C, Cabo Blanco–El Ñuro, with similar risk percentages. Risk assessed in zone A, Banco de Mancora, differed most between the species. Bycatch risk for HBW was greatest for bottom gillnets (24% high risk), though there was 100% medium risk for surface gillnets and purse seine gear types (**Figure 5**). Compared to the LBT risk map within zone A, highest risk percentages for LBT were present for surface gillnets (100%) and longlines (87%) with a 99% medium risk for purse seines (**Figure 10**).

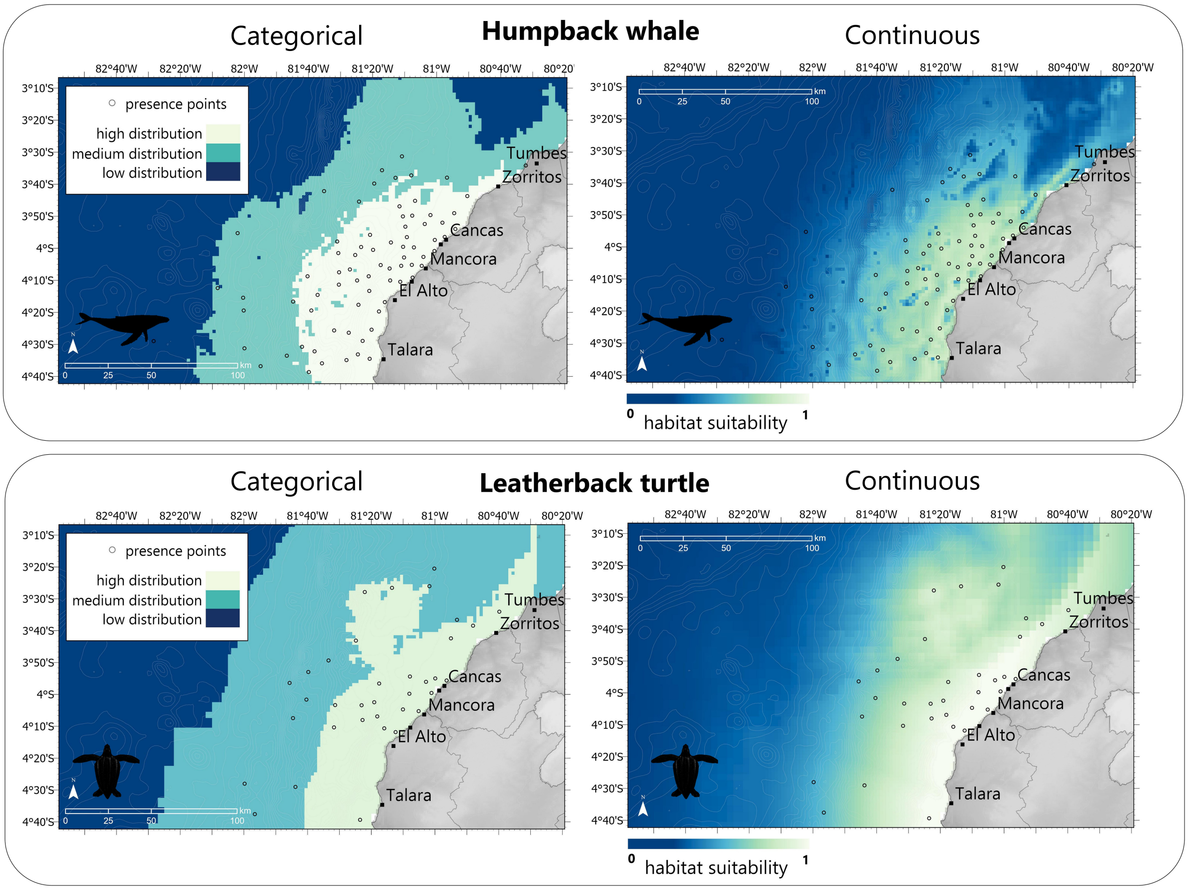


FIGURE 7 | Final Maxent model outputs for humpback whales and leatherback turtles were built based on interview responses from PGIS mapping. The top row are the habitat suitability predictions as continuous data with values ranging between 0 and 1. The second row of maps are discretized into three classification levels using the Relative Occurrence Rate (ROR). The low category ranges from 0 to 10% of the maximum ROR. The medium category ranges from 10% of the maximum ROR to 50% of the maximum ROR, and the high category ranges from 50% of the maximum ROR to the maximum ROR.

DISCUSSION

By applying the Bycatch Risk Assessment (ByRA) model we were able to map areas of high bycatch risk for two marine megafauna species in two highly active artisanal fishing ports in northern Peru. Our assessment outputs and participatory approach are applicable to both localized bycatch mitigation and broader bycatch reduction efforts in data deficient areas as well as generating initial bycatch estimates by identifying areas of bycatch concern and building connections with fishers.

Bycatch Risk Assessment offered a way to engage fishers and provide a platform for collaboration. The project stimulated discussions in the fishing community and provided repeated opportunities for feedback. This was done by using PGIS and returning to the study sites to talk with fishers about the research. We found it especially valuable to collaborate with local organizations, to build upon established relationships and familiarity with the communities. Local groups help access key points of contacts both with direct fisher communities or with other respected members among fisher organizations, admin of the ports, knowledge holders, and/or experts. Historically, bycatch management and legislation in Peru has come from a top-down approach with a heavier focus on small cetaceans (Van Waerebeek et al., 1997, 2002). Engaging key stakeholders (i.e., the fishers) can help strengthen local capacity for assessing large cetacean entanglements (Johnson et al., 2017; Aburto-Oropeza et al., 2018). Overall, we received positive feedback from the map outputs from fishers, who expressed a specific interest in the temporal overlap of the species distributions and fishing areas. For both seasons, our results showed similar geographic areas of high bycatch risk for HBW and LBT. Similar to other studies, higher risk was concentrated nearer to shore where coastal fisheries overlap with productive habitats. For our study area, highest risk was identified within 20 nm offshore and spanned the coastal distance from Talara to Zorritos (**Figures 3**, **8**, **9**). These areas are reflective of the data we used as input into the model. High bycatch risk follows patterns of high habitat preference, fishing intensity, and likelihood of species/gear interaction.

We identified gillnets as the riskiest gear type for both species. Gillnets (surface or demersal) are lethal to many marine species (D'Agrosa et al., 2000; Mangel et al., 2010;

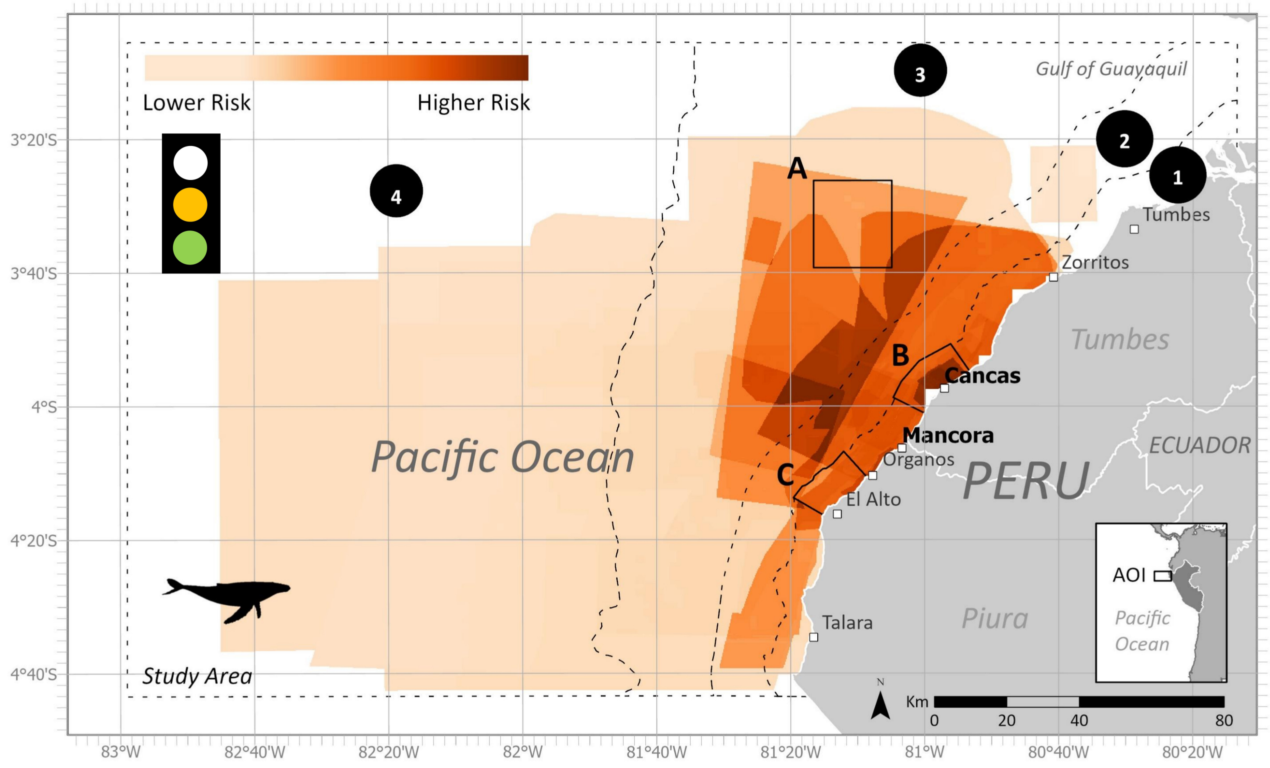


FIGURE 8 | Final estimated bycatch risk within the study area and subzones for humpback whales during the austral winter (June-November). This model output included fishing polygons from both ports. Subzones are represented by black dotted and solid lines. (1) Artisanal fishing zone (5 nm), (2) Territorial seas zone (12 nm), (3) Shelf zone, (4) Beyond shelf zone, (A) Banco de Mancora, (B) Arrecifes de Punta Sal, and (C) Cabo Blanco–El Ñuro. Data uncertainty depicted in stoplight and described further in **Table 3**.

Alfaro-Shigueto et al., 2011; Reeves et al., 2013). They create large obstacles for foraging and migratory species (Brown et al., 2015). Also, in alignment with other studies, respondents mentioned that entanglements of HBW in gillnets reported in our study did not always result in mortality. Continued entanglement in gear after being released or separated from a vessel can yield sublethal conditions and excessive energy costs, which in some cases eventually leads to death (van der Hoop et al., 2017). Within the boundaries of the study area specifically, demersal gillnets ranked as the highest risk for HBW, and surface gillnets ranked most risky for LBT. All proposed marine protected areas (Zones A, B, C) contained some level of high risk of bycatch for one or more gear types, specifically gillnets for HBW and longlines and gillnets for LBT.

More refined spatial information was revealed when we analyzed our data by port. Different characteristics between the ports may provide insight into why one port experienced greater risk for entanglements. For Cancas, bycatch risk was centered nearby to the port, and most bycatch risk diminished beyond 5 nm. In Mancora, overall fishing activity covered larger coastal and further offshore areas. Fishers had more days at sea, used thickly threaded gillnets, fished with longer nets, more hooks, and fished for an array of larger target species (sharks, tuna). It was important to separate data to visually map the spatial differences between the ports. The port-specific maps revealed localized differences in fishing areas and, consequently, bycatch risk.

Different characteristics from each ports' fisheries may have also impacted fishers' perspectives about the longevity, ecological function and importance of humpbacks and leatherbacks to marine biodiversity in Peruvian waters. Responses and comments from the subsample of fishers in Mancora indicated that they considered humpback whales less important than the subsample of fishers from Cancas. This could be because fishers from Mancora experienced more entanglements where the whale was found entangled on site. This could create negative perceptions for the fishers of Mancora, who directly see humpbacks as a nuisance or threat to their fishing and livelihood. This differs from fishers from Cancas who were often left with a large hole in their net and less evidence as to which animal created it. Regardless, most fishers believed that HBW and LBT are important for the ocean. These responses are derived from a subsample and might not be representative of the communities as a whole. Nonetheless, this shows the complexity of positive and negative attitudes between fishers and their relationships with marine megafauna (Seminara et al., 2019). Any mitigation planning within fishing communities going forward should consider these differences in ports and

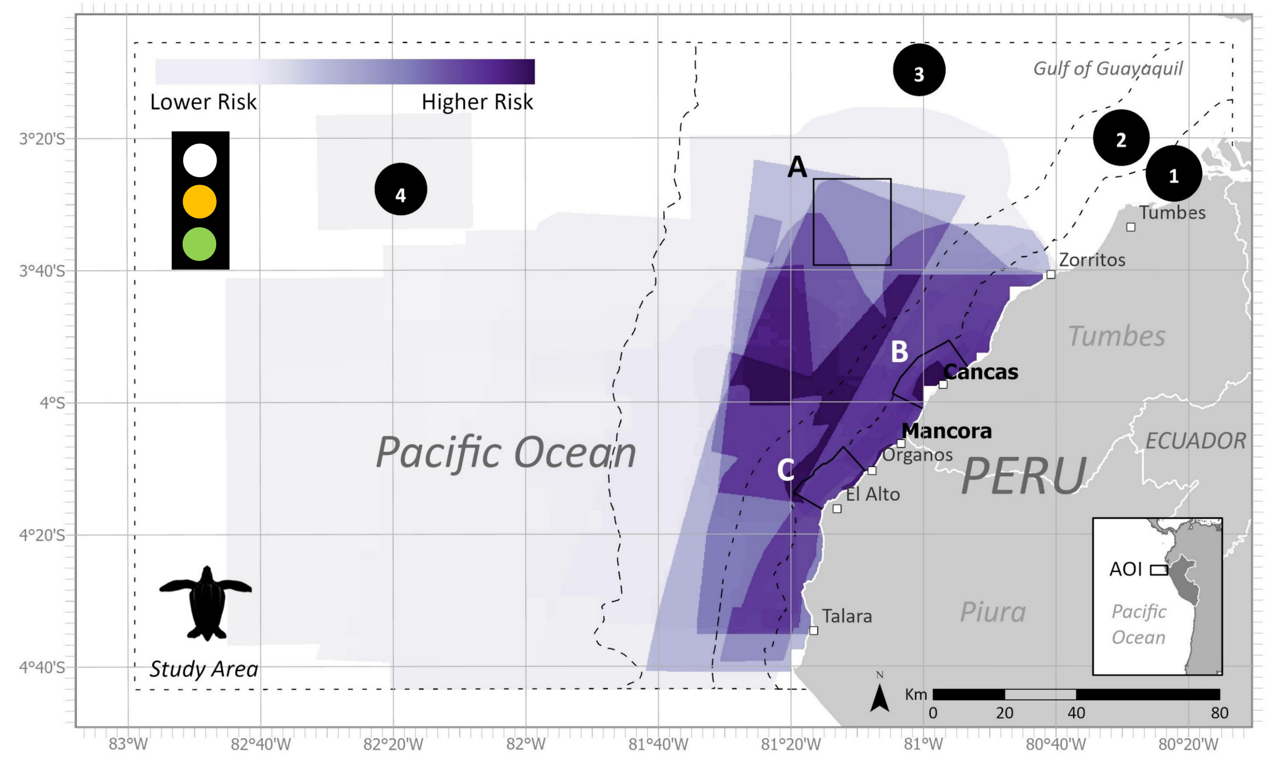


FIGURE 9 | Final estimated bycatch risk within the study area and subzones for leatherback turtles during the austral summer (December-May) with all fishing polygons. (1) Artisanal fishing zone (5 nm), (2) Territorial seas zone (12 nm), (3) Shelf zone, (4) Beyond shelf zone, (A) Banco de Mancora, (B) Arrecifes de Punta Sal, and (C) Cabo Blanco–El Ñuro. Data uncertainty depicted in stoplight and described further in **Table 3**.

attitudes that can impact effective management or fishers' willingness to participate.

Limitations and Biases

Our results showed greatest risk nearest to the ports we assessed, which can be attributed to higher densities of fishing activity but also biased reporting and recall error. Naturally, there is sample bias in the sightings data since fishers would see animals where they fish, unless the animal was seen in passing from fishing grounds to the port. Our use of a combination of purposive and snowball sampling methods was drawn from our positionality as researchers, access of field site/fishers and their ability and willingness to engage with the research team, and field-specific recommendations of the partner institute Pro-Delphinus. Therefore, the results of our study are neither representative of other fishers/fishery locations nor are our results generalizable. However, in our study, 70 of the 85 participants had over 15 years of experience in the trade and a thorough understanding of their local marine environment from within a fisheries context. Our goal was to conduct a qualitative exploration into the spatial extent of bycatch with little aim in generalizing results to the broader community. We were able to gather consensus-based insights from our target population in both fisheries data and fisher perspectives specific to our study sites.

Furthermore, there is no full way to assess the accuracy of the animal identification and fishing locations that the fishers marked on the maps. One way we addressed this was by using vetting tools to assure that interview participants were speaking of humpbacks and leatherbacks when asked questions about these species (Pilcher et al., 2017). Additionally, our methods relied on recall, or the fishers' memories, to report on previous animal sightings and interactions. Based on a study by O'Donnell et al. (2012), fishers' recall error was shown to be less when inquiring about rare, positive, or extreme events (O'Donnell et al., 2012). We used methods from Moore et al. (2010) and Pilcher et al. (2017) to reduce this error. For example, one way we did this was by inquiring about events in specific moments in time (i.e., within the past year, within the past five years and beyond 5 years).

Another limitation in our data collection was in incorporating fisher-derived opportunistic sightings into the species distribution models. Often, predictive models of species distributions rely on the quality of data used as input in the model (Derville et al., 2018). Our process for building these models was through participatory GIS (PGIS). For us, PGIS filled in gaps where data were either non-existent or lacking (Thiault et al., 2017). Several studies have adopted PGIS methods for various marine conservation efforts (for example, Levine and Feinholz, 2015; Moore et al., 2017). In addition to species data, tracking vessel coverage, movement, and effort of SSF is difficult, if not impossible to obtain. Our incorporation of PGIS to map not only fisheries but species sightings required additional layers of caution that we incorporated in our methodology to reduce

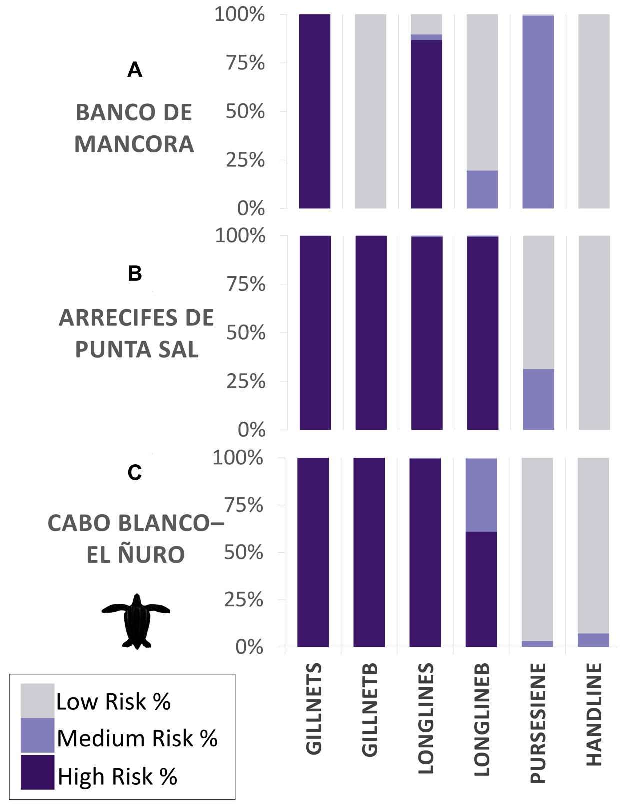


FIGURE 10 | Bycatch risk percentages calculated for leatherback turtles for each gear type. Subzones assessed are composed of proposed marine protected areas: **(A)** Banco de Mancora, **(B)** Arrecifes de Punta Sal, and **(C)** Cabo Blanco–El Ñuro, as part of the Grau Tropical Marine Reserve.

bias and consequent misidentification of habitat (van Strien et al., 2013; Pennino et al., 2019).

Lastly, PGIS methods used to collect data on animal sightings and previous bycatch incidents have given us insights into lesser-known habitat use, especially when systematic transect surveys are logistically infeasible, prohibitively expensive or when detection probability is low (Di Febbraro et al., 2018). The latter could be the case for the EP leatherback turtles, which are rare given their small population size and predominantly use pelagic habitat (Giraud et al., 2016). Using PGIS for species occurrence is an effective route for creating a baseline understanding of species distribution with data uncertainty (Rocchini et al., 2011; Guillera-Arroita et al., 2015).

Bycatch Within a Regional Context

Interviews with fishers suggested higher numbers of HBW entanglements than had been previously documented (Félix et al., 2011; García-Godos et al., 2013). To date, Peruvian SSF bycatch estimates have relied on various short-term collection methods completed by government or non-profit agencies (Mangel et al., 2010; Alfaro-Shigueto et al., 2011; Arlidge et al., 2020). These include onboard observers, fisher interviews, and early testing of remote electronic monitoring (Bartholomew et al., 2018; Ayala et al., 2019; Campbell et al., 2020). Interviews and mapping workshops helped identify areas of greatest risk and improve quantifications of entanglements. From our research, focused on two ports (of the 100+ landing sites along the Peruvian coast), about half of the fishers (n = 41, Mancora = 30, Cancas = 11) self-reported a total range of between 64 and 100 HBW bycatch events in 2018 alone. Of these, 64 were mapped with an estimated location of interaction. Of the 85 fishermen interviewed, 28 individual fishers said they have entangled more than 10 HBW in their lifetime and for LBT, only 4 individual fishers reported accidentally capturing more than 10 in their lifetime. These larger estimates of bycatch far surpass previous counts and provide some evidence for a large number of bycatch events that may be occurring but lack an avenue for continuous reporting.

As our ByRA outputs suggests, there is a wide coastal area identified as medium to high bycatch risk for both species, with a high likelihood that this area will increase in size. According to a recent study on the growth and economic status of the Peruvian SSF fleet, SSF fishing effort is increasing, covering larger fishing areas, while landing less catch (De la Puente et al., 2020). If this national trend of vessel expansion and fishing effort is reflective in our study area, larger areas of habitat would be expected to be at risk as well. Recent review of the Southeast Pacific HBW population, reveals that as a whole, HBW are arriving a month earlier to the northern tropical waters (Avila et al., 2020). This would add to a longer temporal overlap with fisheries as HBW's approach their wintering habitat, especially putting adult whales at higher risk (Pacheco et al., 2021).

Trends in SSF status may also impact LBT's. While the majority of LBT life history is pelagic (Hoover et al., 2019), based on our findings, these turtles also occur in neritic waters, close enough to shore to overlap with artisanal fisheries. Final SDM outputs for leatherback turtles showed greater preference to coastal regions within the depths of 200 m. Though coastal presence has been less commonly reported, nearshore sightings of LBT's have been documented previously via bycatch monitoring or surveys in coastal SSF in northern Peru, western Mexico, the central Americas and Colombia (Alfaro-Shigueto et al., 2008; Arlidge et al., 2020; Ortiz-Alvarez et al., 2020). Given the critical status of the EP leatherback turtle, even a few fishing-related deaths (like those reported here) could further drive declines in the population (Arlidge et al., 2020; Laúd, 2020).

Future Directions

There is no one path to bycatch reduction. The ByRA model and risk maps offer a more accurate way of understanding and framing that enable more informed decisions about how to reduce bycatch. Our outputs seek to guide conversations leading to effective marine management planning. However, these outputs can be viewed as flexible with an option to be continuously adjusted and updated. Especially within the exposure and consequence table, as new information arises on the local fisheries and species, revising ByRA inputs would allow for refined and more accurate risk assessment scenarios. For example, from our interviews, we discovered greater variations between gear types that could be included in the model to further parse out consequences of bycatch interaction between more specified gear variations and individual species. Additionally, as De la Puente et al. (2020) notes, since 2015, gillnet and handline fishers have consistently made below minimum wage, while those using trawl nets, purse seine, longlines and squid gigs have made more. For our research we did not assess bycatch risk for two of these gear types (trawl nets due to their illegal status in the area, and squid jigs). It may be important to include additional gear types that have been identified as fisheries with potentially greater profits that could become more favorable financially to fishers in the near future.

Bycatch Risk Assessment is a robust tool that can facilitate greater empowerment of local fishers and lead to a variety of management and legislative actions. Any form of successful mitigation efforts benefit from fishers' interest and participation beginning with the early steps of planning (Campbell and Cornwell, 2008; Aburto-Oropeza et al., 2018; Twichell et al., 2018). To truly involve fishers in bycatch mitigation process, ByRA provides structured opportunities to engage in sharing information with the possibility to be representative, and having fishers influence the outcomes. If management tools such as dynamic and seasonal fishing bans and MPA no-take areas are not immediately possible or desired, various practices such as limited soak times, net patrolling, and safe post-capture handling can be applied to areas of highest identified bycatch. This way, limited resources and personnel can be directed to where they can be most effective and where key stakeholder buy-in is strongest. As previous studies suggest, fisher participation is strongly dependent on the process and facilitation of collaboration among parties (Reed, 2008). We learned from our conservations with fishers that they had already taken voluntary actions to minimize HBW entanglements during whale season and continued to do so after our project, such as switching gear types, fishing further offshore, or temporarily stopping fishing to avoid HBW's.

Future use of the ByRA model could expand beyond our study area or be applied to other ports along the coast and extend risk analysis to other taxa and species (Hines et al., 2020). Within our study site are several other marine megafauna species with similar bycatch vulnerability, such as small cetaceans or other species of sea turtles. Future habitat modeling should link climatology data to species presence by incorporating El Nino and El Nina seasonal variations to better reflect oceanographic variations (Estrella Arellano and Swartzman, 2010). It may also be desirable to incorporate sightings data from local whale-watching companies as input into species models and build upon local knowledge of whale entanglements (Pacheco et al., 2021).

Bycatch Risk Assessment is a model that is strengthened through careful, representative participation that combined several means of data collection from community members, marine scientists, existing databases, and previous research within the area. ByRA outputs offer spatial, and temporal specific recommendations by gear type, subzone, and species to better engage future management and support marine policy. ByRA can help identify possible other scenarios helpful for fishers and animals, such as switching to other economic activities during the whale entangle season, or temporarily switching to other fishing gears. The tool can help fishers act upon diversification based on real data. Bycatch reduction requires multifaceted approaches, working alongside agencies and across broader regions and countries. Given the pyramidal structure of traditional management measures by developing countries with high numbers of SSF, ByRA could also serve as a method where fishermen have a voice and can be part of local management measures.

DATA AVAILABILITY STATEMENT

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

ETHICS STATEMENT

The studies involving human participants were reviewed and approved by the San Francisco State University Institutional Review Board, Protocol number X19-30. Written informed consent for participation was not required for this study in accordance with the national legislation and the institutional requirements.

AUTHOR CONTRIBUTIONS

AC, CG, and JA-S: fieldwork, logistics, and interviews. AC, MC, and GV: data analysis. AC, CG, EH, JM, and JA-S: study design. AC, CG, JM, JA-S, GV, EH, and AS: writing and editing the manuscript. All authors contributed to the manuscript and approved the submitted version.

FUNDING

This research was funded by the Rufford Foundation grant 27741-1, the American Cetacean Society, San Francisco Chapter-Jonathon Stern Research grant, the Society for Conservation Biology Marine Section Small Grant Program, and a NSF Research Traineeship Grant (award no. DGE-1633336) through the Estuary and Ocean Science Center, at San Francisco State University.

ACKNOWLEDGMENTS

We have the greatest gratitude for all of the fishers who participated in the interviews and workshops in Mancora, Cancas, and Los Organos. We especially want to thank Carlos Belupu, who was crucial during fisher interviews. We thank the ProDelphinus staff who helped with the data collection in the field: Clara Ortiz, Kelly Dias, Adriana Gonzalez, Elizabeth Campbell, and Daniela Thorne.

SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2021.776965/full#supplementary-material

REFERENCES

Aburto-Oropeza, O., López-Sagástegui, C., Moreno-Báez, M., Mascareñas-Osorio, I., Jiménez-Esquivel, V., Johnson, A. F., et al. (2018). Endangered species, ecosystem integrity, and human livelihoods. *Conserv. Lett.* 11:e12358. doi: 10.1111/conl.12358

Alava, J., Tatar, B., José Barragán, M., Castro, C., Rosero, P., Denkinger, J., et al. (2017). Mitigating cetacean bycatch in coastal Ecuador: governance challenges for small-scale fisheries. *Mar. Policy* 110:102769. doi: 10.1016/j.marpol.2017.05.025

Alfaro-Shigueto, J., Dutton, P. H., Van Bressem, M., and Mangel, J. (2008). Interactions between leatherback turtles and peruvian artisanal fisheries. *Chelonian Conserv. Biol.* 6, 129–134.

Alfaro-Shigueto, J., Mangel, J. C., Bernedo, F., Dutton, P. H., Seminoff, J. A., and Godley, B. J. (2011). Small-scale fisheries of Peru: a major sink for marine turtles in the Pacific. *J. Appl. Ecol.* 48, 1432–1440. doi: 10.1111/j.1365-2664.2011.02040.x

Alfaro-Shigueto, J., Mangel, J. C., Darquea, J., Donoso, M., Baquero, A., Doherty, P. D., et al. (2018). Untangling the impacts of nets in the southeastern Pacific: rapid assessment of marine turtle bycatch to set conservation priorities in small-scale fisheries. *Fish. Res.* 206, 185–192. doi: 10.1016/j.fishres.2018.04.013

Alfaro-Shigueto, J., Mangel, J. C., Pajuelo, M., Dutton, P. H., Seminoff, J. A., and Godley, B. J. (2010). Where small can have a large impact: structure and characterization of small-scale fisheries in Peru. *Fish. Res.* 106, 8–17. doi: 10.1016/j.fishres.2010.06.004

Arkema, K. K., Verutes, G., Bernhardt, J. R., Clarke, C., Rosado, S., Canto, M., et al. (2014). Assessing habitat risk from human activities to inform coastal and marine spatial planning: a demonstration in Belize. *Environ. Res. Lett.* 9:114016. doi: 10.1088/1748-9326/9/11/114016

Arlidge, W. N. S., Squires, D., Alfaro-Shigueto, J., Booth, H., Mangel, J. C., and Milner-Gulland, E. J. (2020). A mitigation hierarchy approach for managing sea turtle captures in small-scale fisheries. *Front. Mar. Sci.* 7:49. doi: 10.3389/fmars.2020.00049

Avila, I. C., Dormann, C. F., García, C., Payán, L. F., and Zorrilla, M. X. (2020). Humpback whales extend their stay in a breeding ground in the Tropical Eastern Pacific. *ICES J. Mar. Sci.* 77, 109–118. doi: 10.1093/icesjms/fsz251

Ayala, L., Ortiz, M., and Gelcich, S. (2019). Exploring the role of fishers knowledge in assessing marine megafauna bycatch: insights from the Peruvian longline artisanal fishery. *Anim. Conserv.* 22, 251–261. doi: 10.1111/acv.12460

Bailey, H., Benson, S. R., Shillinger, G. L., Bograd, S. J., Dutton, P. H., Eckert, S. A., et al. (2012). Identification of distinct movement patterns in Pacific leatherback turtle populations influenced by ocean conditions. *Ecol. Appl.* 22, 735–747. doi: 10.1890/11-0633

Bakun, A., and Weeks, S. J. (2008). The marine ecosystem off Peru: what are the secrets of its fishery productivity and what might its future hold? *Progr. Oceanogr.* 79, 290–299. doi: 10.1016/j.pocean.2008.10.027

Bartholomew, D. C., Mangel, J. C., Alfaro-Shigueto, J., Pingo, S., Jimenez, A., and Godley, B. J. (2018). Remote electronic monitoring as a potential alternative to on-board observers in small-scale fisheries. *Biol. Conserv.* 219, 35–45. doi: 10.1016/j.biocon.2018.01.003

Berkes, F., Mahon, R., Mcconney, P., Pollnac, R., and Pomeroy, R. (2001). *Managing Small-scale Fisheries*. Ottawa: IDRC.

Brandt, K., Graham, L., Hawthorne, T., Jeanty, J., Burkholder, B., Munisteri, C., et al. (2020). Integrating sketch mapping and hot spot analysis to enhance capacity for community-level flood and disaster risk management. *Geogr. J.* 186, 198–212. doi: 10.1111/geoj.12330

Breen, P., Brown, S., Reid, D., and Rogan, E. (2017). Where is the risk? Integrating a spatial distribution model and a risk assessment to identify areas of cetacean interaction with fisheries in the northeast Atlantic. *Ocean Coast. Manag.* 136, 148–155. doi: 10.1016/j.ocecoaman.2016.12.001

Brown, J. L., Bennett, J. R., and French, C. M. (2017). SDMtoolbox 2.0: the next generation Python-based GIS toolkit for landscape genetic, biogeographic and species distribution model analyses. *PeerJ* 5:e4095. doi: 10.7717/peerj.4095

Brown, S. L., Reid, D., and Rogan, E. (2015). Spatial and temporal assessment of potential risk to cetaceans from static fishing gears. *Mar. Policy* 51, 267–280. doi: 10.1016/j.marpol.2014.09.009

Campbell, E., Pasara-Polack, A., Mangel, J. C., and Alfaro-Shigueto, J. (2020). Use of small cetaceans as bait in small-scale fisheries in Peru. *Front. Mar. Sci.* 7:534507. doi: 10.3389/fmars.2020.534507

Campbell, L. M., and Cornwell, M. L. (2008). Human dimensions of bycatch reduction technology: current assumptions and directions for future research. *Endanger. Species Res.* 5, 325–334. doi: 10.3354/esr00172

Cashion, T., Al-Abdulrazzak, D., Belhabib, D., Derrick, B., Divovich, E., Moutopoulos, D. K., et al. (2018). Reconstructing global marine fishing gear use: catches and landed values by gear type and sector. *Fish. Res.* 206, 57–64. doi: 10.1016/j.fishres.2018.04.010

Castellanos-Galindo, G. A., Chong-Montenegro, C., Baos, R. A., Zapata, L. A., Tompkins, P., Graham, R. T., et al. (2018). Using landing statistics and fishers' traditional ecological knowledge to assess conservation threats to Pacific goliath grouper in Colombia. *Aquat. Conserv.* 28, 305–314. doi: 10.1002/aqc.2871

Castillo, G., Fernández, J., Medina, A., and Guevara-Carrasco, R. (2018). Third structural survey of the artisanal fishery in the Peruvian littoral. *Gen. Results Inf. Inst. Mar. Peru* 45, 299–388.

Chung, M.-K., Lu, D.-J., Tsai, B.-W., and Chou, K.-T. (2019). Assessing effectiveness of PPGIS on Protected Areas by Governance Quality: a case study of community-based monitoring in wu-wei-kang wildlife refuge, Taiwan. *Sustainability* 11:4154. doi: 10.3390/su11154154

Cooke, J. G. (2018). *Megaptera novaeangliae. IUCN Red List Threatened Species* 2018:e.T13006A50362794.

Croll, D. A., Marinovic, B., Benson, S., Chavez, F. P., Black, N., Ternullo, R., et al. (2005). From wind to whales: trophic links in a coastal upwelling system. *Mar. Ecol. Progr. Series* 289, 117–130. doi: 10.3354/meps289117

D'Agrosa, C., Lennert-Cody, C. E., and Vidal, O. (2000). Vaquita bycatch in Mexico's artisanal gillnet fisheries: driving a small population to extinction. *Conserv. Biol.* 14, 1110–1119. doi: 10.1046/j.1523-1739.2000.98191.x

De la Puente, S., López de la Lama, R., Benavente, S., Sueiro, J. C., and Pauly, D. (2020). Growing into poverty: reconstructing peruvian small-scale fishing effort Between 1950 and 2018. *Front. Mar. Sci.* 7:681. doi: 10.3389/fmars.2020.00681

Denzin, N. K., and Lincoln, Y. S. (2018). *The SAGE Handbook of Qualitative Research (Fifth edition.)*. Thousand Oaks: SAGE.

Derville, S., Torres, L. G., Iovan, C., and Garrigue, C. (2018). Finding the right fit: comparative cetacean distribution models using multiple data sources and statistical approaches. *Divers. Distrib.* 24, 1657–1673. doi: 10.1111/ddi.12782

Di Febbraro, M., Sallustio, L., Vizzarri, M., De Rosa, D., De Lisio, L., Loy, A., et al. (2018). Expert-based and correlative models to map habitat quality: which gives better support to conservation planning? *Glob. Ecol. Conserv.* 16:e00513. doi: 10.1016/j.gecco.2018.e00513

Dransfield, A., Hines, E., McGowan, J., Holzman, B., Nur, N., Elliott, M., et al. (2014). Where the whales are: using habitat modeling to support changes in shipping regulations within national marine sanctuaries in central California. *Endanger. Species Res.* 26, 39–57. doi: 10.3354/esr00627

Dunn, C. E. (2007). Participatory GIS - A people's GIS? *Progr. Hum. Geogr.* 31, 616–637. doi: 10.1177/0309132507081493

Estes, J. A., Terborgh, J., Brashares, J. S., Power, M. E., Berger, J., Bond, W. J., et al. (2011). Trophic downgrading of planet earth. *Science* 333, 301–306. doi: 10.1126/science.1205106

Estrella Arellano, C. E., and Swartzman, G. (2010). The Peruvian artisanal fishery: changes in patterns and distribution over time. *Fish. Res.* 101, 133–145. doi: 10.1016/j.fishres.2009.08.007

FAO (2020). *The State of the World Fisheries and Aquaculture*. Rome: FAO. doi: 10.4060/ca9229en

Félix, F., and Botero-Acosta, N. (2011). Distribution and behaviour of humpback whale mother-calf pairs during the breeding season off Ecuador. *Mar. Ecol. Progr. Series* 426, 277–287. doi: 10.3354/meps08984

Félix, F., and Guzmán, H. M. (2014). Satellite tracking and sighting data analyses of Southeast Pacific humpback whales (*Megaptera novaeangliae*): is the migratory route coastal or oceanic? *Aquat. Mamm.* 40, 329–340. doi: 10.1578/AM.40.4.2014.329

Félix, F., Muñoz, M., Falconí, J., Botero, N., and Haase, B. (2011). Entanglement of humpback whales in artisanal fishing gear in Ecuador. *J. Cetacean Res. Manage.* 3, 283–290.

Fiedler, P. C., Redfern, J. V., Forney, K. A., Palacios, D. M., Sheredy, C., Rasmussen, K., et al. (2018). Prediction of large whale distributions: a comparison of presence-absence and presence-only modeling techniques. *Front. Mar. Sci.* 5:419. doi: 10.3389/fmars.2018.00419

Figueiredo, I., Maia, C., Lagarto, N., and Serra-Pereira, B. (2020). Bycatch estimation of Rajiformes in multispecies and multigear fisheries. *Fish. Res.* 232:105727. doi: 10.1016/j.fishres.2020.105727

García-Godos, I., Van Waerebeek, K., Alfaro-Shigueto, J., and Mangel, J. C. (2013). Entanglements of large cetaceans in Peru: few records but high risk. *Pac. Sci.* 67, 523–532. doi: 10.2984/67.4.3

Gibbs, M. T., and Browman, H. I. (2015). Introduction: risk assessment and risk management: a primer for marine scientists. *ICES J. Mar. Sci.* 72, 992–996. doi: 10.1093/icesjms/fsu232

Giraud, C., Calenge, C., Coron, C., and Julliard, R. (2016). Capitalizing on opportunistic data for monitoring relative abundances of species. *Biometrics* 72, 649–658. doi: 10.1111/biom.12431

Goodman, L. A. (1961). Snowball Sampling. *Ann. Math. Stat.* 32, 148–170.

Gray, C. A., and Kennelly, S. J. (2018). Bycatches of endangered, threatened and protected species in marine fisheries. *Rev. Fish Biol. Fish.* 28, 521–541. doi: 10.1007/s11160-018-9520-7

Guevara-Carrasco, R., and Bertrand, A. (2017). *Atlas de la Pesca Artesanal del Mar del Perú. Edición IMARPE-IRD*. 183. Available online at: http://biblioimarpe.imarpe.gob.pe/handle/123456789/3167 (accessed January 2, 2021).

Guidino, C., Llapapasca, M. A., Silva, S., Alcorta, B., and Pacheco, A. S. (2014). Patterns of Spatial and Temporal Distribution of Humpback Whales at the Southern Limit of the Southeast Pacific Breeding Area. *PLoS One* 9:e112627. doi: 10.1371/journal.pone.0112627

Guillera-Arroita, G., Lahoz-Monfort, J. J., Elith, J., Gordon, A., Kujala, H., Lentini, P. E., et al. (2015). Is my species distribution model fit for purpose? Matching data and models to applications. *Glob. Ecol. Biogeogr.* 24, 276–292. doi: 10.1111/geb.12268

Hall, M., and Roman, M. (2013). *Bycatch and Non-Tuna Catch In The Tropical Tuna Purse Seine Fisheries of the World. FAO Fisheries and Aquaculture Technical Paper, (568)*. Rome: FAO.

Hamer, D. J., Goldsworthy, S. D., Costa, D. P., Fowler, S. L., Page, B., and Sumner, M. D. (2013). The endangered Australian sea lion extensively overlaps with and regularly becomes by-catch in demersal shark gill-nets in South Australian shelf waters. *Biol. Conserv.* 157, 386–400. doi: 10.1016/j.biocon.2012.07.010

Harwood, J. (2000). Risk assessment and decision analysis in conservation. *Biol. Conserv.* 95, 219–226. doi: 10.1016/S0006-3207(00)00036-7

Hashimoto, M., Shirakihara, K., and Shirakihara, M. (2015). Effects of bycatch on the population viability of the narrow-ridged finless porpoises in Ariake Sound and Tachibana Bay, Japan. *Endanger. Species Res.* 27, 87–94. doi: 10.3354/esr00658

Hines, E., Ponnampalam, L., Junchompoo, C., Peter, C., Vu, L., Huynh, T., et al. (2020). Getting to the bottom of bycatch: a GIS-based toolbox to assess the risk of marine mammal bycatch. *Endanger. Species Res.* 42, 37–57. doi: 10.3354/esr01037

Hobday, A. J., Smith, A. D. M., Stobutzki, I. C., Bulman, C., Daley, R., Dambacher, J. M., et al. (2011). Ecological risk assessment for the effects of fishing. *Fish. Res.* 108, 372–384. doi: 10.1016/j.fishres.2011.01.013

Holsman, K., Samhouri, J., Cook, G., Hazen, E., Olsen, E., Dillard, M., et al. (2017). An ecosystem-based approach to marine risk assessment. *Ecosyst. Health Sustain.* 3:e01256. doi: 10.1002/ehs2.1256

Hooker, Y. (2016). *Expediente tecnico Zona Reservada Mar Pacifico Tropical Peruano/Technical Report for the Peruvian Tropical Pacific Sea Reserve*. Peru: SERNANP.

Hoover, A. L., Liang, D., Alfaro-Shigueto, J., Mangel, J. C., Miller, P. I, Morreale, S. J., et al. (2019). Predicting residence time using a continuous-time discrete-space model of leatherback turtle satellite telemetry data. *Ecosphere* 10:e02644. doi: 10.1002/ecs2.2644

Johnson, A. F., Caillat, M., Verutes, G. M., Peter, C., Junchompoo, C., Long, V., et al. (2017). Poor fisheries struggle with U.S. import rule. *Science* 355, 1031–1032. doi: 10.1126/science.aam9153

Kenchington, E., Murillo, F. J., Lirette, C., Sacau, M., Koen-Alonso, M., Kenny, A., et al. (2014). Kernel density surface modelling as a means to identify significant concentrations of vulnerable marine ecosystem indicators. *PLoS One* 9:e109365. doi: 10.1371/journal.pone.0109365

Kramer-Schadt, S., Niedballa, J., Pilgrim, J. D., Schröder, B., Lindenborn, J., Reinfelder, V., et al. (2013). The importance of correcting for sampling bias in MaxEnt species distribution models. *Divers. Distrib.* 19, 1366–1379. doi: 10.1111/ddi.12096

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

Laúd, O. P. O. (2020). Enhanced, coordinated conservation efforts required to avoid extinction of critically endangered eastern Pacific leatherback turtles. *Sci. Rep.* 10:4772. doi: 10.1038/s41598-020-60581-7

Leis, M., de, O., Devillers, R., Medeiros, R. P., and Chuenpagdee, R. (2019). Mapping fishers' perceptions of marine conservation in Brazil: an exploratory approach. *Ocean Coast. Manag.* 167, 32–41. doi: 10.1016/j.ocecoaman.2018.09.017

Levine, A. S., and Feinholz, C. L. (2015). Participatory GIS to inform coral reef ecosystem management: mapping human coastal and ocean uses in Hawaii. *Appl. Geogr.* 59, 60–69. doi: 10.1016/j.apgeog.2014.12.004

Lewison, R. L., Crowder, L. B., Wallace, B. P., Moore, J. E., Cox, T., Zydelis, R., et al. (2014). Global patterns of marine mammal, seabird, and sea turtle bycatch reveal taxa-specific and cumulative megafauna hotspots. *Proc. Natl. Acad. Sci.* 111, 5271–5276. doi: 10.1073/pnas.1318960111

Lewison, R. L., Johnson, A. F., and Verutes, G. M. (2018). Embracing complexity and complexity-awareness in marine megafauna conservation and research. *Front. Mar. Sci.* 5:207. doi: 10.3389/fmars.2018.00207

Luizza, M. W., Wakie, T., Evangelista, P. H., and Jarnevich, C. S. (2016). Integrating local pastoral knowledge, participatory mapping, and species distribution modeling for risk assessment of invasive rubber vine (*Cryptostegia grandiflora*) in Ethiopia's Afar region. *Ecol. Soc.* 21:22. doi: 10.5751/ES-07988-210122

Mancini, A., Koch, V., Seminoff, J. A., and Madon, B. (2012). Small-scale gill-net fisheries cause massive green turtle Chelonia mydas mortality in Baja California Sur, Mexico. *Oryx* 46, 69–77. doi: 10.1017/S0030605310001833

Mangel, J. C., Alfaro-Shigueto, J., Van Waerebeek, K., Cáceres, C., Bearhop, S., Witt, M. J., et al. (2010). Small cetacean captures in Peruvian artisanal fisheries: high despite protective legislation. *Biol. Conserv.* 143, 136–143. doi: 10.1016/j.biocon.2009.09.017

Mannocci, L., Forget, F., Tolotti, M. T., Bach, P., Bez, N., Demarcq, H., et al. (2020). Predicting bycatch hotspots in tropical tuna purse seine fisheries at the basin scale. *Glob. Ecol. Conserv.* 24:e01393. doi: 10.1016/j.gecco.2020.e01393

Mason, J. G., Alfaro-Shigueto, J., Mangel, J. C., Brodie, S., Bograd, S. J., Crowder, L. B., et al. (2019). Convergence of fishers' knowledge with a species distribution model in a Peruvian shark fishery. *Conserv. Sci. Pract.* 1:e13. doi: 10.1111/csp2.13

McCauley, D. J., Pinsky, M. L., Palumbi, S. R., Estes, J. A., Joyce, F. H., and Warner, R. R. (2015). Marine defaunation: animal loss in the global ocean. *Science* 347:1255641.

Moore, J. E., Cox, T. M., Lewison, R. L., Read, A. J., Bjorkland, R., McDonald, S. L., et al. (2010). An interview-based approach to assess marine mammal and sea turtle captures in artisanal fisheries. *Biol. Conserv.* 143, 795–805. doi: 10.1016/j.biocon.2009.12.023

Moore, M. J., and van der Hoop, J. M. (2012). The painful side of trap and fixed net fisheries: chronic entanglement of large whales. *J. Mar. Biol.* 2012, 1–4. doi: 10.1155/2012/230653

Moore, S. A., Brown, G., Kobryn, H., and Strickland-Munro, J. (2017). Identifying conflict potential in a coastal and marine environment using participatory mapping. *J. Environ. Manag.* 197, 706–718. doi: 10.1016/j.jenvman.2016.12.026

Nakandakari, A. (2012). *Determinación de áreas Prioritarias Para la Conservación de la Biodiversidad Marina del Perú*. Ph.D. thesis. Lima: Universidad Peruana Cayetano Heredia.

Nelson, K., and Burnside, N. G. (2019). Identification of marine management priority areas using a GIS-based multi-criteria approach. *Ocean Coast. Manag.* 172, 82–92. doi: 10.1016/j.ocecoaman.2019.02.002

Noble, M. M., Harasti, D., Pittock, J., and Doran, B. (2019). Linking the social to the ecological using GIS methods in marine spatial planning and management to support resilience: a review. *Mar. Policy* 108:103657. doi: 10.1016/j.marpol.2019.103657

Oceanic National and Administration Atmospheric [NOAA] (2016). Oceanic national and administration atmospheric. *Fed. Regist.* 81:54390.

O'Donnell, K. P., Molloy, P. P., and Vincent, A. C. J. (2012). Comparing fisher interviews, logbooks, and catch landings estimates of extraction rates in a small-scale fishery. *Coast. Manag.* 40, 594–611. doi: 10.1080/08920753.2012.727734

Ortiz-Alvarez, C., Pajuelo, M., Grados, D., Abrego, M. E., Rebeca Barragán-Rocha, A., Barrantes, M., et al. (2020). Rapid assessments of leatherback small-scale fishery bycatch in internesting areas in the Eastern Pacific Ocean. *Front. Mar. Sci.* 6:813. doi: 10.3389/fmars.2019.00813

Pacheco, A., Llapapasca, M., López-Tejada, N., Silva, S., and Alcorta, B. (2021). Modeling breeding habitats of humpback whales *Megaptera novaeangliae* as a function of group composition. *Mar. Ecol. Progr. Series* 666, 203–215. doi: 10.3354/meps13686

Pennington, J. T., Mahoney, K. L., Kuwahara, V. S., Kolber, D. D., Calienes, R., and Chavez, F. P. (2006). Primary production in the eastern tropical Pacific: a review. *Progr. Oceanogr.* 69, 285–317. doi: 10.1016/j.pocean.2006.03.012

Pennino, M. G., Paradinas, I., Illian, J. B., Muñoz, F., Bellido, J. M., López-Quílez, A., et al. (2019). Accounting for preferential sampling in species distribution models. *Ecol. Evol.* 9, 653–663. doi: 10.1002/ece3.4789

Phillips, S. J., Anderson, R. P., Dudík, M., Schapire, R. E., and Blair, M. E. (2017). Opening the black box: an open-source release of Maxent. *Ecography* 40, 887–893. doi: 10.1111/ecog.03049

Pilcher, N. J., Adulyanukosol, K., Das, H., Davis, P., Hines, E., Kwan, D., et al. (2017). A low-cost solution for documenting distribution and abundance of endangered marine fauna and impacts from fisheries. *PLoS One* 12:e0190021. doi: 10.1371/journal.pone.0190021

Rajamani, L. (2013). Using community knowledge in data-deficient regions: conserving the Vulnerable dugong Dugong dugon in the Sulu Sea, Malaysia. *Oryx* 47, 173–176. doi: 10.1017/S0030605312000154

Rasmussen, K., Allen, J. M., Palacios, D. M., Calambokidis, J., Saborı′o, M. T., Dalla Rosa, L., et al. (2007). Southern Hemisphere humpback whales wintering off Central America: insights from water temperature into the longest mammalian migration. *Biol. Lett.* 3, 302–305. doi: 10.1098/rsbl.2007.0067

Reed, M. S. (2008). Stakeholder participation for environmental management: a literature review. *Biol. Conserv.* 141, 2417–2431. doi: 10.1016/j.biocon.2008.07.014

Reeves, R. R., McClellan, K., and Werner, T. B. (2013). Marine mammal bycatch in gillnet and other entangling net fisheries, 1990 to 2011. *Endanger. Species Res.* 20, 71–97. doi: 10.3354/esr00481

Servicio Nacional de Áreas Naturales Protegidas por el Estado (n.d.). *Propuesta de Reserva Nacional mar Tropical de Grau. Reserva Nacional Mar Tropical de Grau.* Available online at: https://www.sernanp.gob.pe/reserva-nacional-mar-tropical-de-grau (accessed April 10, 2019).

Rocchini, D., Hortal, J., Lengyel, S., Lobo, J. M., Jimenez-Valverde, A., Ricotta, C., et al. (2011). Accounting for uncertainty when mapping species distributions: the need for maps of ignorance. *Prog. Phys. Geogr.* 35, 211–226. doi: 10.1177/0309133311399491

Roman, J., and McCarthy, J. J. (2010). The whale pump: marine mammals enhance primary productivity in a coastal basin. *PLoS One* 5:e13255. doi: 10.1371/journal.pone.0013255

Roman, J., Estes, J. A., Morissette, L., Smith, C., Costa, D., McCarthy, J., et al. (2014). Whales as marine ecosystem engineers. *Front. Ecol. Environ.* 12, 377–385. doi: 10.1890/130220

Saldanþa, J. (2009). *The Coding Manual for Qualitative Researchers.* Thousand Oaks: SAGE Publications.

Samhouri, J. F., and Levin, P. S. (2012). Linking land-and sea-based activities to risk in coastal ecosystems. *Biol. Conserv.* 145, 118–129.

Sawchuk, J. H., Beaudreau, A. H., Tonnes, D., and Fluharty, D. (2015). Using stakeholder engagement to inform endangered species management and improve conservation. *Mar. Policy* 54, 98–107. doi: 10.1016/j.marpol.2014.12.014

Scholz, A., Bonzon, K., Fujita, R., Benjamin, N., Woodling, N., Black, P., et al. (2004). Participatory socioeconomic analysis: drawing on fishermen's knowledge for marine protected area planning in California. *Mar. Policy* 28, 335–349. doi: 10.1016/j.marpol.2003.09.003

Seminara, C. I., Barbosa-Filho, M. L. V., and Le Pendu, Y. (2019). Interactions between cetaceans and artisanal fishermen from Ilhéus, Bahia – Brazil. *Biota Neotropica* 19, 1–13. doi: 10.1590/1676-0611-bn-2019-0742

Sharp, R., Douglass, J., Wolny, S., Arkema, K., Bernhardt, J., Bierbower, W., et al. (2019). *InVEST 3.7.0 User Guide. Collaborative publication by The Natural Capital Project, Stanford University, the University of Minnesota, The Nature Conservancy, and the World Wildlife Fund).* Stanford: Stanford University.

Shester, G. G., and Micheli, F. (2011). Conservation challenges for small-scale fisheries: bycatch and habitat impacts of traps and gillnets. *Biol. Conserv.* 144, 1673–1681. doi: 10.1016/j.biocon.2011.02.023

Shillinger, G. L., Palacios, D. M., Bailey, H., Bograd, S. J., Swithenbank, A. M., Gaspar, P., et al. (2008). Persistent leatherback turtle migrations present opportunities for conservation. *PLoS Biol.* 6:e171. doi: 10.1371/journal.pbio.0060171

Spalding, M. D., Fox, H. E., Allen, G. R., Davidson, N., Ferdaña, Z. A., Finlayson, M., et al. (2007). Marine ecoregions of the world: a bioregionalization of coastal and shelf areas. *BioScience* 57, 573–583. doi: 10.1641/B570707

Spotila, J. R., Reina, R. D., Steyermark, A. C., Plotkin, P. T., and Paladino, F. V. (2000). Pacific leatherback turtles face extinction. *Nature* 405, 529–530. doi: 10.1038/35014729

Stephenson, F., Goetz, K., Sharp, B. R., Mouton, T. L., Beets, F. L., Roberts, J., et al. (2020). Modelling the spatial distribution of cetaceans in New Zealand waters. *Divers. Distrib.* 26, 495–516. doi: 10.1111/ddi.13035

Strickland-Munro, J., Kobryn, H., Brown, G., and Moore, S. A. (2016). Marine spatial planning for the future: using Public Participation GIS (PPGIS) to inform the human dimension for large marine parks. *Mar. Policy* 73, 15–26. doi: 10.1016/j.marpol.2016.07.011

Szostek, C. L., Murray, L. G., Bell, E., and Kaiser, M. J. (2017). Filling the gap: using fishers' knowledge to map the extent and intensity of fishing activity. *Mar. Environ. Res.* 129, 329–346. doi: 10.1016/j.marenvres.2017.06.012

Thiault, L., Collin, A., Chlous, F., Gelcich, S., and Claudet, J. (2017). Combining participatory and socioeconomic approaches to map fishing effort in smallscale fisheries. *PLoS One* 12:e0176862. doi: 10.1371/journal.pone.0176862

Tobeña, M., Prieto, R., Machete, M., and Silva, M. A. (2016). Modeling the potential distribution and richness of cetaceans in the Azores from fisheries observer program data. *Front. Mar. Sci.* 3:202. doi: 10.3389/fmars.2016.00202

Twichell, J., Pollnac, R., and Christie, P. (2018). Lessons from Philippines MPA management: social ecological interactions, participation, and MPA performance. *Environ. Manag.* 61, 916–927. doi: 10.1007/s00267-018-1020-y

van der Hoop, J., Corkeron, P., and Moore, M. (2017). Entanglement is a costly life-history stage in large whales. *Ecol. Evol.* 7, 92–106. doi: 10.1002/ece3.2615

van Strien, A. J., van Swaay, C. A., and Termaat, T. (2013). Opportunistic citizen science data of animal species produce reliable estimates of distribution trends if analysed with occupancy models. *J. Appl. Ecol.* 50, 1450–1458. doi: 10.1111/1365-2664.12158

Van Waerebeek, K., Alfaro-Shigueto, J., Montes, D., Onton, K., Santillan, L., Van Bressem, M. F., et al. (2002). *Fisheries Related Mortality of Small Cetaceans in Neritic Waters of Peru in 1999–2001. International Whaling Commission, Scientific Committee Document SC/54/SM10.* Cambridge: International Whaling Commission.

Van Waerebeek, K., Van Bressem, M. F., Félix, F., Alfaro-Shigueto, J., García-Godos, A., Chávez-Lisambart, L., et al. (1997). Mortality of dolphins and porpoises in coastal fisheries off Peru and southern Ecuador in 1994. *Biol. Conserv.* 81, 43–49. doi: 10.1016/S0006-3207(96)00152-8

Verutes, G., Andrew, F., Johnson Marjolaine, C., Ponnampalam, L. S., Peter, C., Vu, L., et al. (2020). Using GIS and stakeholder involvement to innovate marine mammal bycatch risk assessment in data-limited fisheries. *PLoS One* 15:e0237835. doi: 10.1371/journal.pone.0237835

Visalli, M. E., Best, B. D., Cabral, R. B., Cheung, W. W. L., Clark, N. A., Garilao, C., et al. (2020). Data-driven approach for highlighting priority areas for protection in marine areas beyond national jurisdiction. *Mar. Pol.* 122:103927. doi: 10.1016/j.marpol.2020.103927

Wallace, B. P., Tiwari, M., and Girondot, M. (2013). Dermochelys coriacea (East Pacific Ocean subpopulation). *IUCN Red List Threatened Species* 2013:e.T46967807A46967809.

Williams, R., Burgess, M. G., Ashe, E., Gaines, S. D., and Reeves, R. R. (2016). U.S. seafood import restriction presents opportunity and risk. *Science* 354, 1372–1374. doi: 10.1126/science.aai8222

Worm, B., Barbier, E. B., Beaumont, N., Duffy, J. E., Folke, C., Halpern, B. S., et al. (2006). Impacts of biodiversity loss on ocean ecosystem services. *Science* 314, 787–790. doi: 10.1126/science.1132294

Yang, D., and Pomeroy, R. (2017). The impact of community-based fisheries management (CBFM) on equity and sustainability of small-scale coastal fisheries in the Philippines. *Mar. Policy* 86, 173–181. doi: 10.1016/j.marpol.2017.09.027

Zolkafli, A., Liu, Y., and Brown, G. (2017). Bridging the knowledge divide between public and experts using PGIS for land use planning in Malaysia. *Appl. Geogr.* 83, 107–117. doi: 10.1016/j.apgeog.2017.03.013

Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

frontiers in Marine Science

ORIGINAL RESEARCH
published: 24 December 2021
doi: 10.3389/fmars.2021.795953

Evaluating Strategies for Managing Anthropogenic Mortality on Marine Mammals: An R Implementation With the Package RLA

Mathieu Genu[1], Anita Gilles[2], Philip S. Hammond[3], Kelly Macleod[4], Jade Paillé[1], Iosu Paradinas[3], Sophie Smout[3], Arliss J. Winship[5] and Matthieu Authier[1,6]*

[1] Observatoire Pelagis, UMS 3462, CNRS-La Rochelle Université, La Rochelle, France, [2] Institute for Terrestrial and Aquatic Wildlife Research, University of Veterinary Medicine Hannover, Büsum, Germany, [3] Sea Mammal Research Unit, Scottish Oceans Institute, University of St Andrews, St Andrews, United Kingdom, [4] Joint Nature Conservation Committee, Aberdeen, United Kingdom, [5] CSS Inc., Fairfax, VA, United States, [6] ADERA, Pessac, France

Edited by:
Lars Bejder,
University of Hawaii at Manoa,
United States

Reviewed by:
Debra Palka,
Northeast Fisheries Science Center
(NOAA), United States
Claudio Vasapollo,
Istituto Superiore per la Protezione é la
Ricerca Ambientale (ISPRA), Italy

***Correspondence:**
Matthieu Authier
matthieu.authier@univ-lr.fr

Specialty section:
This article was submitted to
Marine Megafauna,
a section of the journal
Frontiers in Marine Science

***Received:** 15 October 2021*
***Accepted:** 24 November 2021*
***Published:** 24 December 2021*

Citation:
Genu M, Gilles A, Hammond PS,
Macleod K, Paillé J, Paradinas I,
Smout S, Winship AJ and Authier M
(2021) Evaluating Strategies for
Managing Anthropogenic Mortality on
Marine Mammals: An R
Implementation With the Package
RLA. Front. Mar. Sci. 8:795953.
doi: 10.3389/fmars.2021.795953

Bycatch, the undesirable and non-intentional catch of non-target species in marine fisheries, is one of the main causes of mortality of marine mammals worldwide. When quantitative conservation objectives and management goals are clearly defined, computer-based procedures can be used to explore likely population dynamics under different management scenarios and estimate the levels of anthropogenic removals, including bycatch, that marine mammal populations may withstand. Two control rules for setting removal limits are the Potential Biological Removal (PBR) established under the US Marine Mammal Protection Act and the Removals Limit Algorithm (RLA) inspired from the Catch Limit Algorithm (CLA) developed under the Revised Management Procedure of the International Whaling Commission. The PBR and RLA control rules were tested in a Management Strategy Evaluation (MSE) framework. A key feature of PBR and RLA is to ensure conservation objectives are met in the face of the multiple uncertainties or biases that plague real-world data on marine mammals. We built a package named RLA in the R software to carry out MSE of control rules to set removal limits in marine mammal conservation. The package functionalities are illustrated by two case studies carried out under the auspices of the Oslo and Paris convention (OSPAR) (the Convention for the Protection of the Marine Environment of the North-East Atlantic) Marine Mammal Expert Group (OMMEG) in the context of the EU Marine Strategy Framework Directive. The first case study sought to tune the PBR control rule to the conservation objective of restoring, with a probability of 0.8, a cetacean population to 80% of carrying capacity after 100 years. The second case study sought to further develop a RLA to set removals limit on harbor porpoises in the North Sea with the same conservation objective as in the first case study. Estimation of the removals limit under the RLA control rule was carried out within the Bayesian paradigm. Outputs from the functions implemented in the package RLA allows the assessment of user-defined performance metrics, such as time to reach a given fraction of carrying capacity under a given level of removals compared to the time needed given no removals.

Keywords: bycatch, conservation, management, marine mammal, PBR, RLA, R

INTRODUCTION

Marine mammal conservation requires understanding and assessing the consequences of anthropogenic activities, in particular removals (e.g., bycatch; Wade et al., 2021), at the population level. Bycatch, the non-intentional capture or killing of marine mammals in commercial or recreational fisheries, is one of the major threats to marine mammals (Avila et al., 2018) and small-sized cetaceans in particular (Reeves et al., 2013; Gray and Kennelly, 2018; Brownell et al., 2019; Rogan et al., 2021). Managing bycatch, or more generally any anthropogenic removal of marine mammals is paramount, lest the examples of the baiji (*Lipotes vexillifer*) and the vaquita (*Phocoena sinus*) be repeated. An appropriate framework for managing anthropogenic activities and their impact should include remedial and timely actions when objectives are not met. Conservation actions that rely only on detection of statistically significant population decline are inoperant: statistical significance will be evidenced too late to enact corrective measures to prevent decline or extinction (Gerrodette, 1987; Cooke, 1994; Wade, 1998; Taylor et al., 2007; Williams et al., 2008; Authier et al., 2020). Early warnings must be identified for pro-active prevention of the population decline of marine mammals. This philosophy underlies the approach enshrined in the US Marine Mammal Protection Act (MMPA, see **Table 1** for abbreviations) *via* the management strategy known as Potential Biological Removal (PBR; Wade, 1998).

The MMPA has legal teeth because, among others, it spells out a clear quantitative conservation objective (CO) and lays out management objectives and remedial measures to meet the CO. In contrast, a critical gap hindering marine mammal conservation in the European Union (EU) is the lack of (i) a legally-binding CO for marine mammals and (ii) management objectives with respect to human-caused mortality (ICES, 2020b; Rogan et al., 2021). In 2010, the International Council for the Exploration of the Sea (ICES) asked the European Commission (EC) for explicit conservation and management objectives for marine mammal populations (ICES, 2010) but this was not acted upon (see ICES, 2013, pages 35–37 for further discussion). Lacking an explicit CO, the simplest, but also the crudest, approach for assessing the impact of anthropogenic removals on marine mammal populations is to consider a fixed percentage of total abundance as a threshold. For instance, the Agreement on the Conservation of Small Cetaceans of the Baltic, North East Atlantic, Irish and North Seas (ASCOBANS[1]) passed two resolutions, one in 2000 (Resolution 3.3 on Incidental Take of Small Cetaceans) and the other in 2006 (Resolution 5.5 on Incidental Take of Small Cetaceans) which

- defines "unacceptable interactions" as being, in the short term, a total anthropogenic removal above 1.7% of the best available estimate of abundance (Res.3.3); and
- underlines the intermediate precautionary objective to reduce by-catches to less than 1% of the best available population estimate (Res.3.3 and Res.5.5).

These resolutions make use of a fixed percentage approach to set removal limits due to anthropogenic activities. The EC accepted the ICES (2010) advice to use such an approach (Anonymous, 2010), although without endorsing any of the technical elements within the advice as policy. The fixed percentage approach has been used for small cetaceans, based on the best available recent estimates of abundance and bycatch levels (ICES, 2020b). Several European member states used this approach in their assessment of "Good Environmental Status" (GES) as required by the Marine Strategy Framework Directive (Anonymous, 2008), the overarching instrument to ensure the sustainable use of marine ecosystems in the EU (Korpinen et al., 2021). The advantage of using a fixed percentage of abundance to manage removals lies in its simplicity: only a single estimate of removals and a single estimate of abundance are required. The calculations are transparent, simple, and can be easily followed by all stakeholders. A major shortcoming, however, is how this approach (i) fails to incorporate other information about the population (e.g., life-history parameters) and (ii) does not account for potential errors or bias in estimates or for epistemic uncertainty (i.e., uncertainty about population dynamics; Winship, 2009). Another shortcoming is how, in practice, there is often a temporal mismatch between the available removals and abundance estimates. A conservative approach is to set a removals limit as the management objective which represents an upper bound not to be exceeded. This is the approach followed by the US MMPA and the PBR management strategy (Wade, 1998).

A management strategy is an agreed-upon set of rules for determining thresholds beyond which a CO runs the risk of not being met with unacceptably high probability (Punt, 2006; Winship, 2009; Bunnefeld et al., 2011; Kaplan et al., 2021). This strategy defines management objectives in the form of thresholds that managers can monitor from available data, with the management objectives that these thresholds are not exceeded. As epitomized by the example of whaling, an important scientific insight in the development of precautionary management was the realization that the process of evaluating a management strategy (Management Strategy Evaluation, MSE) was possible with modeling and simulations (Cooke, 1994; Hilborn and Mangel, 1997). MSE thus needs generative models, that is models that can generate (synthetic) data that are similar to observed, and crucially, currently available data. These models need to be more than simple curve-fitting devices and should be infused with ecological realism as much as possible to sustain their long-term use for management. These models are data-generating mechanisms: they can reproduce and simulate the dynamics of an ecological system such as a population subjected to anthropogenic removals on top of natural processes such as density dependence. With these models, scientists can evaluate the performance of management actions in "what-if," or counterfactual, scenarios to set efficient management objectives. Important, the latter will be gauged against observable and available data (e.g., abundance and bycatch estimates, along with their uncertainties) only and not from unknown quantities (e.g., true abundance; Cooke, 1994). Uncertainties in the underlying model, potential biases and uncertainty in the observed data must

---

[1] https://www.ascobans.org/en/species/threats/bycatch

TABLE 1 | Abbreviation used in the paper.

Abbreviation	Meaning
ABC	Approximate Bayesian Computation
ASCOBANS	Agreement on the Conservation of Small Cetaceans of the Baltic, North-East Atlantic, Irish and North Seas
CLA	Catch Limit Algorithm
CO	Conservative Objective
EC	European Commission
GES	Good Environmental Status
ICES	International Council for the Exploration of the Sea
IPL	Internal Protection Level
IWC	International Whaling Commission
MCMC	Markov Chain Monte Carlo
MMPA	Marine Mammal Protection Act
MNPL	Maximum Net Productivity Level
mPBR	modified Potential Biological Removal
MSE	Management Strategy Evaluation
MSFD	Marine Strategy Framework Directive
OMMEG	OSPAR Expert Group on Marine Mammals
OSPAR	Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR stands for Oslo and Paris convention)
PBR	Potential Biological Removal
RMP	Revised Management Procedure
SCANS	Small Cetaceans in European Atlantic waters and the North Sea

be considered in order to ensure that a management strategy is robust to those. Uncertainty should thus be incorporated into management procedures (Punt, 2006) and neither be dismissed as noise nor used to postpone corrective measures by strategic use of ignorance (Mangel et al., 1996; Punt, 2006; Rayner, 2012). Uncertainties and potential biases justify also conservatism in thresholds and rules to avoid running the risk of missing the CO (Mangel et al., 1996).

A management strategy should cover all aspects of management in accordance with pre-specified objectives, including data and analysis requirements, a mathematical formula for calculating thresholds, and a set of rules for all expected situations. Thresholds in the context of marine mammal conservation will take the form of a removals limit, that is an annual maximum number of animals whose removal would not result in excessive depletion of the population. MSE thus requires several components, including:

1. one or several unambiguous quantitative CO;
2. a data simulator (or operating model) to emulate the dynamics of the marine mammal population and the effects of anthropogenic activities on this population;
3. a control rule, whose computation accounts for the expected quantity and quality of observable data, to set a removals limit beyond which the impact of human activities runs the risk of failing the aforementioned CO; and
4. performance metrics, necessarily context-dependent and reflecting the trade-off between the potentially multiple CO defined previously.

All the above are necessary to project forward in time the population dynamics (that is, numbers of animals at each time step, according to population models operating within the data simulator). For each management strategy, the selected control rule is applied: performance metrics are monitored and ultimately assessed with respect to the CO. Items (1) and (4) should be agreed upon by all stakeholders or taken from national or international law if available and transferable. Scientists alone should not be expected or forced to set the CO (Mangel et al., 1996), lest they engage (willingly or not) in "stealth advocacy" which may jeopardize the policy process (Pielke, 2007). Items (2) and (3) fit more squarely under the remit of scientists, whose task is to test a large panel of realistic scenarios to buffer the management strategy against uncertainties and potential biases in the available data. MSE is thus computer intensive as it needs tuning *via* simulations. Tuning means in the MSE context to find, with a large number of simulations, parameter values of the control rule that meet the CO. Running a large number of simulations has become rather mundane because of the power of modern computers. In practice, however, coding an adequate data simulator may present a daunting task. In addition, to minimize duplication of effort by research groups, and to enhance reproducibility, a common tool is desirable. This is precisely our goal in developing the RLA package for statistical software `R` which has become the *lingua franca* of statistical computing for a wide community including many scientists and managers.

Recently, tools to easily run MSE for marine mammals have been developed in the context of the US so-called "import rule" (Williams et al., 2016). These new regulations of the MMPA Import Provision require any nation exporting seafood products to the USA to establish a comparability finding for fisheries that have incidental or intentional mortality and serious injury of marine mammals (Wade et al., 2021). A comparability finding is a demonstration of equivalence in marine mammal conservation effectiveness to those governing bycatch in US fisheries. It requires, among other things, the calculation of a bycatch limit under the PBR control rule. Compliance with these new regulations may be challenging, especially for developing nations (Johnson et al., 2016). Fortunately, tools[2] to assist in determining PBR for fisheries of nations exporting sea products to the US have been developed (Siple et al., 2021). Implicit in this context is the acceptance of the MMPA CO: "a population will remain at, or recover to, its maximum net productivity level MNPL (typically 50% of the populations carrying capacity), with 95% probability, within a 100-year period" (Wade, 1998). PBR has been extensively tested (Wade, 1998; Punt et al., 2020a) and its robustness is well established. Yet, MSE is entirely dependent on the CO: if the latter change, a new MSE needs to be carried out. This justifies the need for an applied tool to easily re-run simulations when needed, and to possibly consider control rules other than PBR.

We describe below our RLA package which includes a set of functions to carry out MSE using contemporary population dynamics models for marine mammals species (Punt, 2016). Documentation on MSE for managing marine mammal removals is abundant, yet there is a comparatively dearth of applied tools to carry out MSE (but see Brandon et al., 2017). This gap motivated the development of the package. The manuscript format will be unusual in meshing together in the main text equations and `R` command lines. This choice is motivated to ease the mapping from the principles of MSE to its application for readers not yet familiar with MSE in practice. Our contribution is to illustrate its use *via* two cases studies on cetaceans, building on the work of Wade (1998) and Winship (2009) among others. The first case study considers a management strategy under the PBR control rule, then called modified PBR (mPBR), and illustrates tuning it to another CO than the US MMPA, namely the ASCOBANS short-term practical sub-objective "to restore and/or maintain stocks/populations to 80% or more of the carrying capacity" (Res.3.3). The second case study focuses on furthering the development of a Removals Limit Algorithm (RLA) to set limits to anthropogenic mortality of harbor porpoises (*Phocoena phocoena*) in the North Sea (Winship, 2009; Hammond et al., 2019). The RLA is similar in concept to the Catch Limit Algorithm (CLA) of the International Whaling Commission (IWC)'s Revised Management Procedure (RMP[3]; Boyce, 2000). We first introduce notations, then detail of the data simulators currently implemented in the RLA package before carrying out the tuning of management strategies with respect to the quantitative interpretation of the ASCOBANS CO made by the Oslo and Paris convention (OSPAR) (Convention for the Protection of the Marine Environment of the North-East Atlantic[4]) expert group on marine mammals (OMMEG). The article closes on possible extensions of the package.

MATERIALS AND METHODS

Installation

The RLA package for statistical software `R` (R Core Team, 2020) can be installed from https://gitlab.univ-lr.fr/pelaverse/rla by typing the following in an `R` (version $\geq$ 4.0.0) console:

```
remotes::install_gitlab(host =
              "https://gitlab.univ-lr.fr",
              repo = "pelaverse/rla"
              )
library(RLA)
```

Notation

Notations are detailed in **Table 2**. Greek letters denote random variables, and the bold font is used to flag a vector of parameters. Let $\mathcal{U}(l, u)$ denote the uniform distribution bounded by real numbers l and u. Let $\log\mathcal{N}(\mu, \sigma)$ denotes the log-normal distribution of location parameter μ and scale parameter σ. The mean of a log-normal random variable y is a function of both the location and scale parameters: $\mathbb{E}[y] = e^{\mu+\frac{\sigma^2}{2}}$. Let $\mathrm{Dir}(\boldsymbol{\alpha})$ denotes the Dirichlet distribution of concentration parameters $\boldsymbol{\alpha}$. Let $\mathrm{Bin}(N, \pi)$ and $\mathrm{Multin}(N', \boldsymbol{\pi}')$ denote, respectively, the binomial distribution of parameters $N \in \mathbb{N}^*, \pi \in [0, 1]$ and multinomial distribution of parameters $N' \in \mathbb{N}^*$, $\forall \pi'$, $\pi' \in [0, 1]$ such that $\sum \pi' = 1$.

Potential Biological Removal

Wade (1998) developed a pragmatic approach to set limits to anthropogenic mortality of small cetaceans and pinnipeds with minimal data requirements named PBR. The formula for the removal control rule (the so called "harvest rule" in fisheries science) behind PBR is:

$$\mathrm{PBR} = 0.5 \times \mathrm{R_{max}} \times \mathrm{N_{min}} \times \mathrm{F_r} \tag{1}$$

where $\mathrm{R_{max}}$ is the maximum theoretical or estimated productivity rate of the population (the annual *per capita* rate of increase in a population resulting from additions due to reproduction, less losses due to natural mortality), $\mathrm{N_{min}}$ is the minimum population estimate in numbers of animals (i.e., the 20th percentile of the best available abundance estimate, usually the most recent one, assuming a lognormal distribution; Wade, 1998), and $\mathrm{F_r}$ is a recovery factor between 0.1 and 1.0.

For small cetaceans, $\mathrm{R_{max}}$ is difficult to estimate in practice but the value of 4% has been used as a default (Wade, 1998). $\mathrm{F_r}$ is most often chosen below 1 (Punt et al., 2018) to (i) account for the current depletion level of the population (the more depleted, the lower) and (ii) allow for some protection against bias and uncertainties in the data: the use of $\mathrm{F_r} < 1.0$ buffers against uncertainties that might prevent population recoveries, such as biases in the estimation of $\mathrm{N_{min}}$ and $\mathrm{R_{max}}$. Wade (1998) determined in a MSE designed for the US MMPA the default value $\mathrm{F_r} = 0.5$ for populations that are depleted, threatened, or of unknown status. The $\mathrm{F_r}$ value can be increased up to 1.0 when populations are well studied and biases in the estimation of $\mathrm{N_{min}}$ and other parameters are thought to be negligible (Punt et al., 2020a). The different values used in

---

[2] https://github.com/mcsiple/mmrefpoints

[3] https://iwc.int/rmp2

[4] https://www.ospar.org/convention

TABLE 2 | Notations.

Name	Type	Meaning
K	Integer	Total carrying capacity (in number of individuals)
K^{1+}	Integer	Adult carrying capacity (in number of individuals)
N_t	Integer	Total abundance (in number of individuals) at time *t*
$\hat{N}_t$	Integer	Estimated total abundance (in number of individuals) at time *t*
cv_t	Positive real	Coefficient of variation associated with $\hat{\mathrm{N}}_{\mathrm{t}}$
N_t^{1+}	Integer	Abundance excluding calves of the year (in number of individuals) at time *t*
N_t^x	Integer	Abundance of *x*-years old individuals at time *t*
$N_{t,f}^x$	Integer	Abundance of *x*-years old females at time *t*
$N_{t,m}^x$	Integer	Abundance of *x*-years old males at time *t*
C_t	Integer	Bycatch or removals (in number of individuals) at time *t*
B_t	Integer	Births (in number of calves) at time *t*
D_t	Positive real	Depletion at time *t*: ratio of either N_t over K or N_t^{1+} over K^{1+}
L	Positive real	Maximum longevity
$\boldsymbol{\phi}$	Probability	Vector of length $L + 1$ of age-specific survival probabilities
m	Proportion	Vector of length $L + 1$ of proportions of mature females of a given age
$\boldsymbol{\eta}$	Positive real	Vector of length $L + 1$ of age-specific relative vulnerabilities to bycatch
p_{K}	Proportion	Vector of length $L + 1$ with the stable age structure
r	Positive real	Current population growth rate
MNP	Positive real	Maximum Net Productivity: the maximum possible *per capita* rate of increase per year
$\mathrm{b_t}$	Proportion	Density-dependent birth rate of female calves per female
MNPL	Proportion	Maximum Net Productivity Level
z	Positive real	Shape parameter of the Generalized Logistic Population Growth model
$\mathrm{R_{max}}$	Positive real	Maximum theoretical or estimated productivity rate; related to MNP
$\mathrm{F_r}$	Proportion	Recovery factor
$\mathrm{N_{min}}$	Integer	Minimum population estimate; related to $\hat{N}_t$
IPL	Proportion	Internal Protection Level; a fraction of K
w	Positive real	Weight for the likelihood (Equation 4)
cv_{env}	Positive real	Coefficient of variation associated with environmental stochasticity
ρ	$-1 < \rho < 1$	Correlation coefficient in the first order random walk model to simulate environmental stochasticity

Equation (1) were determined by tuning the PBR control rule to the MMPA CO: "a population will remain at, or recover to, its maximum net productivity level MNPL (typically 50% of the population's carrying capacity), with 95% probability, within a 100-year period." With a different CO, new default values should be determined using MSE (that is, simulations).

The operating model (data simulator) for carrying out simulations with the PBR control rule is a deterministic, age-aggregated, generalized logistic (Pella-Tomlinson) model of population dynamics (refer to **Table 2** for notation; Punt, 2016), implemented in the function `pellatomlinson_pbr()`. A call to the function requires several user-specified inputs, such as the MNPL, to set the appropriate value of parameter z controlling density-dependence. The MNPL corresponds to the level of population depletion at which the population reaches its Maximum Net Productivity (MNP), the maximum renewal rate of the population. The user only needs to specify MNPL: parameter z in **Algorithm 1** can be derived with a call to function `inverse_MNPL()`. Alternatively, the users can directly specify z: for example, $z = 2.40$ is often used for cetaceans to set MNPL at $0.6K$ (Wade, 1998).

The operating model (**Algorithm 1**) for carrying out simulations with the PBR control rule assumes that the coefficient of variation of the abundance estimates is sampled from a uniform distribution with a user-defined upper bound, but a lower bound of 10%. This is an assumption about realistic levels of precision that may be achieved on empirical estimates of marine mammal abundance. For example, all estimates from the SCANS-III surveys of marine mammals in the Northeast Atlantic had coefficients of variation larger than 10% (Hammond et al., 2021).

The following code snippet launches a population dynamics simulation, starting at a depletion level of $D_0 = 5\%$ of K. The population is allowed to grow for 150 years to reach K before removals start and impact the population for 50 years. Removals are generated by randomly drawing a number of caught animals from a uniform distribution (and rounding down to the nearest integer). These removals will deplete the

Algorithm 1 | Pella-Tomlinson age-aggregated population dynamics model.

```
Require: $K > 0, \mathrm{MNPL}, \mathrm{MNP}, D_0, CV$
  $N_0 \leftarrow K \times D_0$                                                          ▷ Initial condition
  for t in 1:T do
    $N_t \leftarrow \max(0, N_{t-1}\left(1 + \mathrm{R}_{\max}\left(1 - D_{t-1}^z\right)\right) - C_t)$                   ▷ Population dynamics
    $D_t \leftarrow \frac{N_t}{K}$                                                         ▷ Depletion
    if Survey takes place at time $t$ then
      $cv_t \sim \mathcal{U}(0.1, CV)$                                                ▷ Coefficient of variation
      $\hat{N}_t \sim \log\mathcal{N}\Big(\log N_t - 0.5 \log\left(1 + cv_t^2\right), \sqrt{\log\left(1 + cv_t^2\right)}\Big)$   ▷ Estimated abundance
    end if
  end for
```

population and can be later used to estimate a removals limit if it can be assumed that these initial removals, which are taking place before implementation of a control rule, can be estimated and are available.

```
set.seed(123)
hp <- pellatomlinson_pbr(burnin = 150,
                         depletion0 = 0.05,
                         Rmax = 0.04,
                         catches = floor
                        (runif(50, 1e3, 5e3)),
                         seed_id = 20210219
                         )
summary_plot(hp, lower_zero = TRUE)
```

A call to `summary_plot()` generates **Figure 1** to display the population dynamics: the gray area shows the period in which removals are taking place and may deplete the population.

The function `forward_pbr()` allows PBR to be tuned to a CO. The function requires the output from `pellatomlinson_pbr()` and will carry forward the population dynamics using Equation 1 and **Algorithm 1** to set limits to anthropogenic removals C_t:

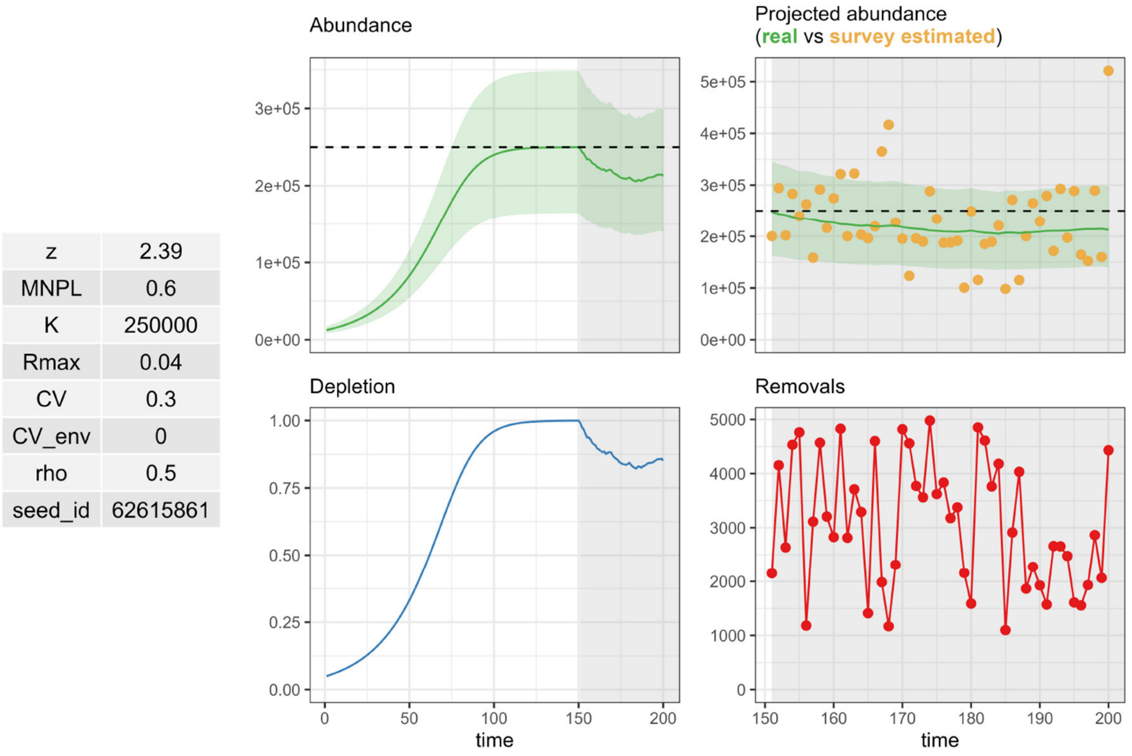
```
pbr_simul <- forward_pbr(pbrlist = hp,
                         distribution =
                          "truncnorm",
                         frequency = 6,
                         horizon = 100,
                         q = 0.2,
                         F_r = 0.5
                         )
```

The user needs to specify a time/management horizon, a frequency at which surveys are carried out to estimate population abundance, and a distribution to generate the removals (e.g., a truncated normal distribution; Punt et al., 2021). If unspecified, $\mathrm{R}_{\max}$ is recycled from the previous call to `pellatomlinson_pbr()`. Other arguments, detailed in the documentation available with `?pellatomlinson_pbr`, can be specified to perform "robustness" trials.

To illustrate the capabilities of the implemented functions, a modifed PBR (mPBR) was tuned to the CO "a population should be able to recover to or be maintained at 80% of carrying capacity, with probability 0.8, within a 100-year period." This CO is a quantitative interpretation from the OMMEG of the ASCOBANS interim objective "to restore and/or maintain stocks/populations to 80% or more of the carrying capacity" (ASCOBANS, 2000). The same MSE as Wade (1998) was carried out (with $K = 10,000$), except that the frequency at which survey estimates were assumed to be collected was set to 6 years to match the MSFD reporting cycle. The computation of $\mathrm{N}_{\min}$ was computed as the 20% of a log-normal distribution of mean $\hat{N}_t$ and associated coefficient of variation cv, which can be computed with the function `PBR()`. Note that this function relies on directly using quantiles of a log-normal distribution and is slightly different from the calculations presented in Wade (1998) which use quantiles from a normal distribution and exponentiation. Tuning was achieved by evaluating the same base case scenario and "robustness" trials of Wade (1998) with respect to parameter $\mathrm{F_r}$. A base case scenario is a reference situation whereby uncertainties are minimal (e.g., life history or population parameters, such as $\mathrm{R}_{\max}$ or depletion, are known with confidence) and the data are assumed without bias (e.g., no systematic error in abundance estimates). On the other hand, robustness trials address deviations from this base case scenario: "the performance of calculating the PBR in various ways was evaluated under simulations involving plausible flaws in the data or assumptions, such as substantial biases in the abundance or mortality estimates" (Wade, 1998; page 7). For each scenario/trial, 1,000 simulations were run (Williams et al., 2008), and the final depletion level of the population was monitored after 100 years of using equation (1) to set limits to anthropogenic removals. All simulations were carried out on a Dell laptop latitude 5400 (Intel(R) Core(TM) i7-8665U CPU @ 1.90GHz, 2112MHz, 4 cores, 8 logical processors, 32Gb RAM).

Removals Limit Algorithm

An RLA is derived from the CLA of the IWC's RMP. The RLA is comprised of a statistical model which is fitted to a time series of estimates of abundance and bycatch to estimate population growth rate r and current depletion D_T (with T corresponding to the last survey estimate of abundance), both of which are then used in the calculation of a control rule (removals limit). The removals control rule under the RLA, as a fraction of the latest



z	2.39
MNPL	0.6
K	250000
Rmax	0.04
CV	0.3
CV_env	0
rho	0.5
seed_id	62615861

FIGURE 1 | Output from a call to `summary_plot()` on a `pellatomlinson_pbr()` output. Inputs are summarized on the left. Population dynamics are displayed in the middle, with either abundance (top) or depletion (bottom). Simulated survey estimates (top) and removals are displayed on the right subfigures.

abundance estimate, is given by Equation 2:

$$\text{removals limit} = r \times \max\left(0, D_T - \text{IPL}\right) \tag{2}$$

where the Internal Protection Level (IPL) is the depletion threshold below which the limit is set to 0. A removals limit of zero is thus used if the estimated current population depletion is less than the IPL, and a non-nil limit, based on estimated stock productivity, is used otherwise (Boyce, 2000). An important difference in the control rule between RLA and PBR is that RLA requires both a time series of abundance estimates and anthropogenic removals (e.g., bycatch). These data are used to estimate the current depletion D_T and population growth rate r in the following model (Boyce, 2000):

$$N_t = N_{t-1} - C_{t-1} + r \times N_{t-1} \times (1 - D_{t-1} \times D_{t-1}) \tag{3}$$

Particularly, the quantities N_t are not parameters in Equation (3), but quantities that have a deterministic relationship with the unknown parameters r and D_T. The latter corresponds to the current population depletion level at the time T of the most recent survey estimate: $N_T = \text{K} \times D_T$ gives the final condition for the abundance process in Equation (3) (and thus the model can derive the abundances backward until $t = 0$). The initial condition is $D_0 = 1$ (that is $N_0 = K$): the population is assumed to be at carrying capacity for $t \leq 0$, that is before the start of the time series of estimated anthropogenic removals C_t.

The likelihood $\ell(N_t|r, D_T)$ of a datum N_t under the model specified in Equation (3) is a weighted (with weight w) log-normal probability density function (Boyce, 2000; Aldrin et al., 2008):

$$\ell(N_t|r, \text{D}_T) = \left(\frac{\exp\left(-\frac{\left(\log\left(N_t\right) - \mu_t\right)^2}{2 \times \sigma_t^2}\right)}{N_t \times \sigma_t \times \sqrt{2\pi}}\right)^w \tag{4}$$

The IWC's CLA down-weights the likelihood during model fitting, which represents a departure from the Bayesian paradigm. This down-weighting of the likelihood was found to stabilize the variance in removals limit and improve the performance of the CLA (Cooke, 1999). The rationale for down-weighting information from new data is to limit the speed at which the management procedure responds to feedback. In the RLA, down-weighting of the likelihood is also possible, and is set to $w = 1/16$ by default (as in the CLA). The likelihood can only be evaluated for the years t^* in which survey estimates $\hat{N_{t^*}}$ are available. For those years, $\sigma_{t^*} = \sqrt{\log\left(1 + cv_{t^*}^2\right)}$ and $\mu_{t^*} = \log\left(\hat{N_{t^*}}\right) - \frac{\log\left(1 + cv_{t^*}^2\right)}{2}$. The full likelihood is $\mathcal{L} = \prod_{t^*} \ell(N_{t^*}|r, \text{D}_T)$.

Estimation of the parameters of model (Equation 3) is carried out in a Bayesian framework, and was coded in `Stan`

Algorithm 2| Pella-Tomlinson age-disaggregated population dynamics model.

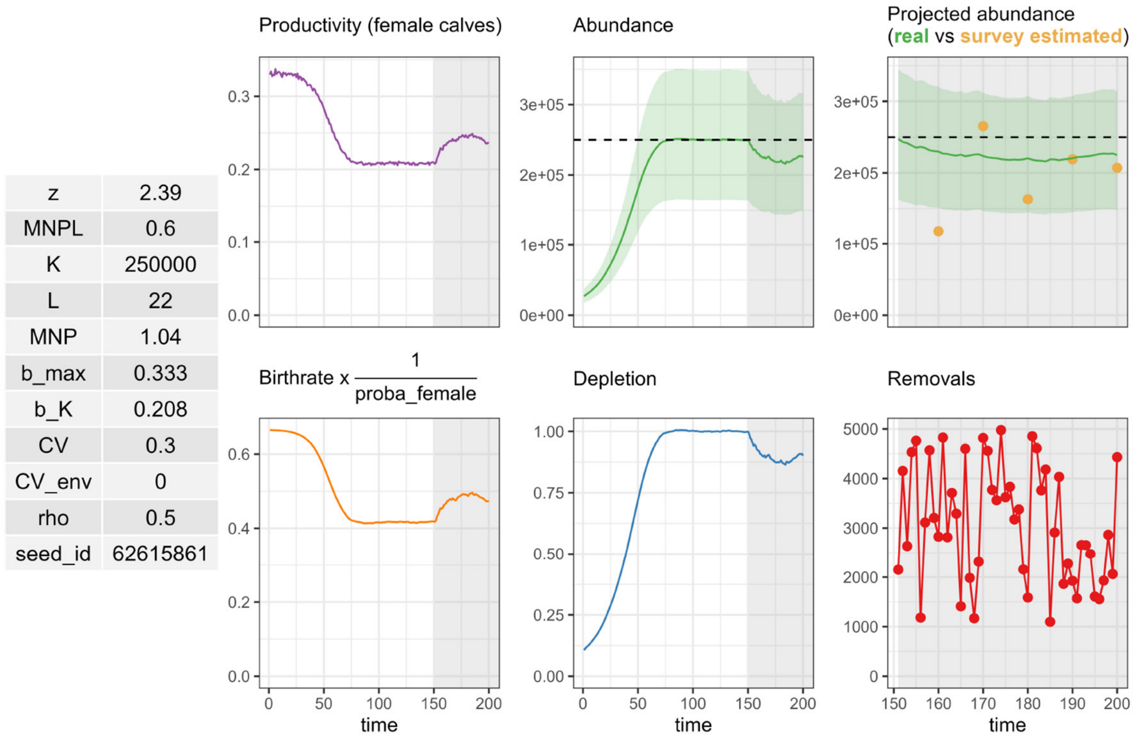
```
Require: $K > 0, L > 0, \boldsymbol{\phi}, \boldsymbol{\eta}, \mathbf{m}$, MNPL, MNP, $CV$
Ensure: length($\boldsymbol{\phi}$) = length($\boldsymbol{\eta}$) = length($\mathbf{m}$) = $L + 1$        ▷ Includes 0-year old
  $b_{\max} \leftarrow \lambda^{-1}(\text{MNP})$        ▷ Solve for birth rate corresponding to the MNP
  $b_K \leftarrow \lambda^{-1}(0)$        ▷ Solve for equilibrium birth rate
  $K^{1+} \leftarrow K\left(1 - \frac{1}{\sum\left(\prod(1,\phi_1),\ldots,\prod(1,\phi_1,\ldots,\phi_L)\right)}\right)$        ▷ Adult carrying capacity, excluding calves
  if $N_{0,f}$ and $N_{0,m}$ not specified and $D_0$ specified then
    $N_0 \leftarrow K \times D_0$        ▷ Initial condition
    $N_{0,f}, N_{0,m} \leftarrow 0.5 \times N_0$        ▷ Initial number of females and males
    $\mathbf{N^x_{0,f}} \sim \text{Multin}(N_{0,f}, \mathbf{p_K})$        ▷ Initial number of females of age $x$
    $\mathbf{N^x_{0,m}} \sim \text{Multin}(N_{0,m}, \mathbf{p_K})$        ▷ Initial number of males of age $x$
  end if
  for t in 1:T do
    $b_t \leftarrow \max\left(0, b_K + (b_{\max} - b_K) \times \left(1 - D^z_{t-1}\right)\right)$        ▷ Density-dependent birth rate
    $C_t \leftarrow \min(C_t, N_{t-1})$        ▷ Ensure removals do not exceed population size
    $\boldsymbol{\pi} \sim \text{Dir}(\mathbf{N^x_{t-1}} \times \boldsymbol{\eta})$        ▷ Age-specific removals proportions
    $\mathbf{R^x_t} \sim \text{Multin}(C_t, \boldsymbol{\pi})$        ▷ Age-specific removals
    for x in 1:L do
      $N^x_{t,f} \sim \text{Bin}\left(\max\left(0, N^{x-1}_{t-1,f} - 0.5R^{x-1}_t\right), \phi_{x-1}\right)$        ▷ Female survival
      $N^x_{t,m} \sim \text{Bin}\left(\max\left(0, N^{x-1}_{t-1,m} - 0.5R^{x-1}_t\right), \phi_{x-1}\right)$        ▷ Male survival
      $M^x_{t,f} \sim \text{Bin}(N^x_{t,f}, m_x)$        ▷ Number of mature females of age $x$
      $B^x_t \sim \text{Bin}(M^x_{t,f}, 2b_t)$        ▷ Number of calves from mature females of age $x$
    end for
    $N^0_t \leftarrow \sum_{x=1}^{L} B^x_t$        ▷ Total number of calves
    $N^0_{t,f} \sim \text{Bin}(N^0_t, 0.5)$        ▷ Female calves
    $N^0_{t,m} \leftarrow \max(0, N^0_t - N^0_{t,f})$        ▷ Male calves
    $N_t \leftarrow \sum_{x=0}^{L} N^x_t$        ▷ True abundance
    $N^{1+}_t \leftarrow \sum_{x=1}^{L} N^x_t$        ▷ Abundance, excluding calves
    $D_t \leftarrow \frac{N^{1+}_t}{K^{1+}}$        ▷ Depletion
    if Survey takes place at time $t$ then
      $cv_t \sim \mathcal{U}(0.1, CV)$        ▷ Coefficient of variation
      $\hat{N}_t \sim \log\mathcal{N}\left(\log N_t - 0.5\log\left(1 + cv^2_t\right), \sqrt{\log\left(1 + cv^2_t\right)}\right)$        ▷ Estimated abundance
    end if
  end for
```

(Carpenter et al., 2017). Stan uses Hamiltonian dynamics in Markov chain Monte Carlo (MCMC) to sample values from the joint posterior distribution of D_T and r (Carpenter et al., 2017). From this sample, the posterior distribution of Equation (2) is easily obtained and a decision analysis is carried out on which quantile to use to summarize this posterior. With the RLA, tuning is done by selecting a quantile to summarize the posterior distribution of Equation (2). Ultimately, this quantile corresponds to a number of animals, but selecting a quantile allows a dispassionate assessment as the user need not work directly on a number of animals: the number will only be revealed at a later stage.

The model code in `Stan` syntax is stored as text data in a dataframe within the RLA package and can be accessed with:

```
library(rstan)
data(rlastan_models)
# use uniform priors
cat(rlastan_models$uniform)
# compile model
rlastan <- rstan::stan_model
   (model_code = rlastan_models$uniform,
   model_name = "Removals Limit Algorithm"
   )
```

The `rstan` library (Stan Development Team, 2020) is required for the RLA but is not included among the dependencies of the RLA package so that the user must load the library themself. The model code currently uses uniform priors on both parameters (r, D_T). The prior for D_T is bounded between 0 and 1, and the prior for r is bounded below by 0 but requires the user to set the upper bound according to the species/population under study. This can be set using the function `standata()`, which also needs user input on values for IPL and w. The IWC uses IPL $= 0.54$ and $w = \frac{1}{16}$, and these are the default values of the function. This function `standata()` is



z	2.39
MNPL	0.6
K	250000
L	22
MNP	1.04
b_max	0.333
b_K	0.208
CV	0.3
CV_env	0
rho	0.5
seed_id	62615861

FIGURE 2 | Output from a call to `summary_plot() on a pellatomlinson_rla` output. Inputs are summarized on the left. Population dynamics are displayed in the middle, with either productivity and abundance (top) or birthrate and depletion (bottom). Simulated survey estimates (top) and removals are displayed on the right sub-figures.

primarily meant for MSE with simulations as it requires the output of a call to the function `pellatomlinson_rla()` which implements a stochastic and age-disaggregated version of a generalized logistic (Pella-Tomlison) model of population dynamics (**Algorithm 2**). The operating model presented in **Algorithm 2** assumes a balanced sex-ratio at birth, and the density-dependent birth rate is expressed as female calves per female by default (hence the factor 2 when simulating the number of calves). The output of `pellatomlinson_rla()` can be visualized with a call to `summary_plot()` to generate **Figure 2**. More specifically, the operating population dynamics model is conditioned on the species/population under study and requires knowledge of age-specific vital rates such as survival and fecundity.

In contrast to the previous example with the PBR control rule, which uses a rather generic (and deterministic) operating model for marine mammal population dynamics, the RLA control rule is used with an operating model conditioned on specific values for a population of a given species. The harbor porpoise in the North Sea is one of the most studied species of marine mammal in European waters. It is also protected in both national and union-level legislation such as the Habitats Directive. In particular, it is listed on both Annexes II and IV of the said directive which requires designation of protected area and strict protection for this species. In the OSPAR Intermediate Assessment 2017, an assessment of harbor porpoise bycatch could not be carried out due to the lack of an agreed upon removals limit and ongoing discussions on methods to set such a limit. In 2009, ICES (2009) advised the European Commission "that a CLA approach is the most appropriate method to set limits on the bycatch of harbor porpoises [...]." The use of the RLA control rule for setting removals limit to this species in the North Sea was agreed at OSPAR's biodiversity and ecosystems committee in 2021. For illustration, an RLA was tuned to the CO "the harbor porpoise population in the North Sea should be able to recover to or be maintained at 80% of carrying capacity, with probability 0.8, within a 100-year period."

Life-history parameters for the harbor porpoise (*Phocoena phocoena*) in the North Sea were taken from Hammond et al. (2019): they are available as data in the RLA package with `data("north_sea_hp")`. The frequency with which survey estimates were assumed to be collected was set to 6 years to match the MSFD reporting cycle. Carrying capacity during the simulations was set to $K = 500,000$. The IPL was set to 0.54, that is in a population estimated to be depleted to less than 54% of carrying capacity, the removals limit was automatically set to 0. The weight w was set to $\frac{1}{16}$. The upper bound for the uniform prior on parameter r was set to 0.1 given recent evidence on the maximum growth rate of harbor porpoise populations (Forney et al., 2021). Tuning was achieved by evaluating the same base case scenario as Hammond et al. (2019) and some "robustness" trials. For each scenario/trial, $1,200$ simulations were run, and the final depletion level of the population was monitored after 100 years of using equation (2) to set limits to anthropogenic removals. The 20th–80th quantile, by an increment of 10, were evaluated. The initial depletion was set between 0.3 and 0.9 of K, with 200 simulations in each bin $[0.3 : 0.4[, \ldots, [0.8 : 0.9[$. The MNPL was drawn from a normal distribution centered on 0.6, with an SD of 0.05 (**Figure 3**). For the base case scenario, changing the time horizon to 50 or 200 years with respect to the CO was also considered as part of a sensitivity analysis. All simulations were carried out on the supercomputer facilities of the "Mésocentre de calcul de Poitou Charentes (Université de Poitiers/ISAE-ENSMA/La Rochelle Université)."

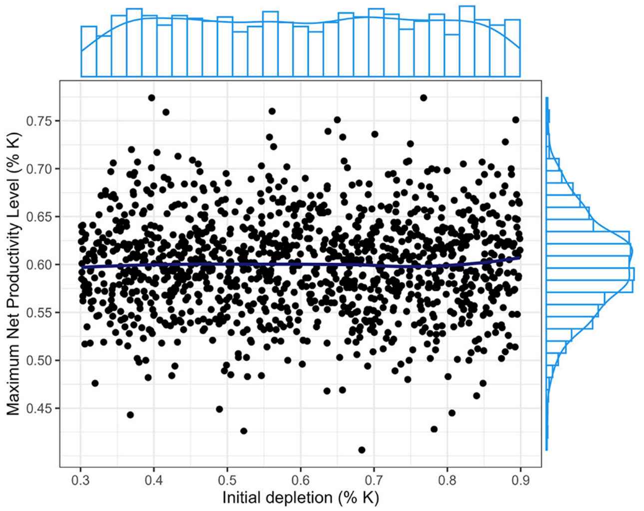


FIGURE 3 | Inputs for simulations with the Removals Limit Algorithm (RLA). A uniform distribution is induced on the initial depletion level and a normal distribution centered on 0.6 for the Maximum Net Productivity Level (MNPL).

RESULTS

Modified PBR

All results can be accessed *via* a shiny application (Chang et al., 2021), available at https://gitlab.univ-lr.fr/pelaverse/pbrfrtuning:

```
remotes::install_gitlab(
  host = "https://gitlab.univ-lr.fr",
  repo = "pelaverse/pbrfrtuning"
)
library(pbrFrTuning)
run_app()
```

Base Case Scenario

The CO was reached in the base case scenario with $F_r = 0.35$ and $F_r = 0.60$ assuming $CV = 0.2$ and 0.8, respectively (**Figure 4**). In other words, with the CO "a cetacean population should be able to recover to or be maintained at 80% of carrying capacity, with probability 0.8, within a 100-year period," the recovery factor F_r should not be set above 0.6 when abundance is imprecisely estimated, and not above 0.35 when it is precisely estimated. The recovery factor F_r could take a higher value when abundance $\hat{N}$ was imprecisely estimated (larger *cv*) because N_{min} is defined as the 20% quantile of a log-normal distribution. In computing

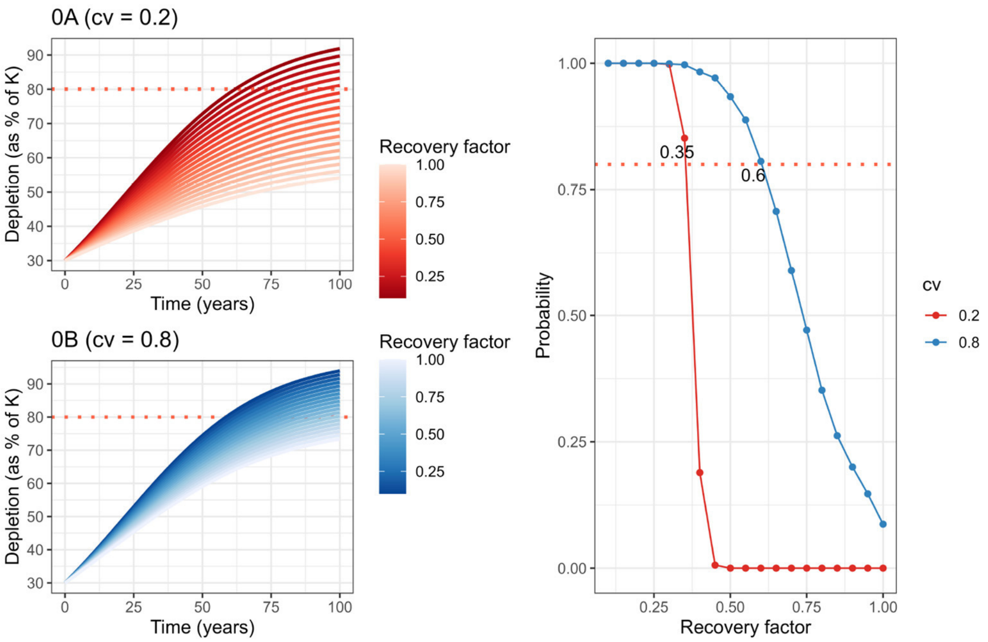


FIGURE 4 | Representation of recovery factor (F_r) impact on depletion level over time for the base case scenario (left) and probability of reaching Conservative Objective (CO) depending on the F_r values (right).

this quantile, the scale and location parameters for the log-normal distribution are, respectively $\sigma = \sqrt{\log\left(1 + cv^2\right)}$ and $\mu = \log(\hat{N}) - \frac{\log\left(1+cv^2\right)}{2}$. A larger cv results both in a larger value for the scale σ and in a lower value for location μ for the same estimated abundance $\hat{N}$: $N_{\min}$ is lowered as a result (and the skewness of the distribution, which is solely a function of σ, is increased). This behavior may be visualized with the `PBR()` function implemented in the package which returns a plot of the assumed log-normal distribution.

Robustness Trials

Tuning of the recovery factor F_r for the modified PBR is summarized in **Table 3**. F_r could vary from 0.15 to 1.0 across the different trials. In particular, scenarios in which bycatch was underestimated by a factor 2 or abundance was overestimated by a factor 2 led to selecting a value of $F_r = 0.15$. Scenarios 7A and 7B, corresponding to a lower MNPL than the one assumed, revealed a lack of robustness as no value of $F_r \geq 0.1$ allowed the CO to be reached.

Differences Between PBR and mPBR for Cetacean Species

Table 3 recapitulates possible choices for F_r depending on several biases or uncertainty. The 8 first scenarios are the same as those of Wade (1998) who found that the value of $F_r = 0.50$ was sufficient for cetaceans to meet the MMPA CO of reaching at least 50% of carrying capacity with a probability of 0.95 over 100 years. In contrast, with the CO of reaching at least 80% of carrying capacity with probability of 0.8 over 100 years, the sufficient value was $F_r = 0.15$. This illustrates the change induced by changing the CO between PBR and mPBR for cetacean species.

Removals Limit Algorithm

All results can be accessed *via* a shiny application (Chang et al., 2021), available at https://gitlab.univ-lr.fr/pelaverse/rlascenarioviz:

```
remotes::install_gitlab(
  host = "https://gitlab.univ-lr.fr",
  repo = "pelaverse/rlascenarioviz"
)
library(rlaScenarioViz)
run_app()
```

Base Case Scenario

The CO "the harbor porpoise population in the North Sea should be able to recover to or be maintained at 80% of carrying capacity, with probability 0.8, within a 100-year period" was reached in the base case scenario by selecting the 55th

TABLE 3 | Summary of parameters combination for each robustness trial tested and Fr associated.

Robustness trials	scen.	q	MNPL	K_{trend}	freq.	R_{max}	CV	b.byc.	b.ab.	b.R_{max}	byc.CV	cata.	F_r
Base case scenario	0A	0.2	0.50	1.0	6	0.04	0.2	1	1	1.0	0.3	0.0	0.35
	0B	0.2	0.50	1.0	6	0.04	0.8	1	1	1.0	0.3	0.0	0.60
Bycatch underestimation	1A	0.2	0.50	1.0	6	0.04	0.2	**2**	1	1.0	0.3	0.0	0.15
	1B	0.2	0.50	1.0	6	0.04	0.8	**2**	1	1.0	0.3	0.0	0.30
Abundance overestimation	2A	0.2	0.50	1.0	6	0.04	0.2	1	**2**	1.0	0.3	0.0	0.15
	2B	0.2	0.50	1.0	6	0.04	0.8	1	**2**	1.0	0.3	0.0	0.30
Maximum Productivity rate understimation	3A	0.2	0.50	1.0	6	0.04	0.2	1	1	**0.5**	0.3	0.0	0.70
	3B	0.2	0.50	1.0	6	0.04	0.8	1	1	**0.5**	0.3	0.0	1.00
Higher variation in N_{min}	4A	0.2	0.50	1.0	6	0.04	**0.8**	1	1	1.0	0.3	0.0	0.55
	4B	0.2	0.50	1.0	6	0.04	**1.6**	1	1	1.0	0.3	0.0	1.00
Higher bycatch coefficient of variation	5A	0.2	0.50	1.0	6	0.04	0.2	1	1	1.0	**1.2**	0.0	0.30
	5B	0.2	0.50	1.0	6	0.04	0.8	1	1	1.0	**1.2**	0.0	0.50
Lower survey frequency	6A	0.2	0.50	1.0	**10**	0.04	0.2	1	1	1.0	0.3	0.0	0.35
	6B	0.2	0.50	1.0	**10**	0.04	0.8	1	1	1.0	0.3	0.0	0.55
Lower MNPL	7A	0.2	**0.45**	1.0	6	0.04	0.2	1	1	1.0	0.3	0.0	NA
	7B	0.2	**0.45**	1.0	6	0.04	0.8	1	1	1.0	0.3	0.0	NA
Higher MNPL + bycatch underestimation	8A	0.2	**0.70**	1.0	6	0.04	0.2	**2**	1	1.0	0.3	0.0	0.70
	8B	0.2	**0.70**	1.0	6	0.04	0.8	**2**	1	1.0	0.3	0.0	1.00
Catastrophic events happening	9A	0.2	0.50	1.0	6	0.04	0.2	1	1	1.0	0.3	**0.1**	0.25
	9B	0.2	0.50	1.0	6	0.04	0.8	1	1	1.0	0.3	**0.1**	0.45
Carying capacity degradation	10A	0.2	0.50	**0.5**	6	0.04	0.2	1	1	1.0	0.3	0.0	0.30
	10B	0.2	0.50	**0.5**	6	0.04	0.8	1	1	1.0	0.3	0.0	0.55

Ten scenarios are tested, each with 2 cases assuming either a coefficient of variation (CV) for abundance estimates at 0.2 or 0.8 (except for scenarios 4A and 4B where figures for CV were doubled). scen., Scenario identifier; q, percentile of N_{min} kept for simulation testing; MNPL, Maximum Net Productivity Level; K_{trend}, assumed fraction of the initial K at the end of the simulation; freq., survey frequency; R_{max}, Maximum theoretical or estimated productivity rate; CV, Coefficient of variation associated with N_{min}; b.byc., Bias in bycatch estimates (2 means an underestimation by a factor 2); b.ab., Bias in abundance estimates (2 means an overestimation by a factor 2); b.R_{max}, Bias in R_{max} (0.5 means that the assumed R_{max} is one-half of the true value); byc.CV , Coefficient of Variation of bycatch estimates; cata., Magnitude (in % of K of a catastrophic event occurring randomly during simulation; F_r, Recovery Factor. All these parameters are inputs of function `forward_pbr()`*. Red values indicate values specific to each robustness trials.*

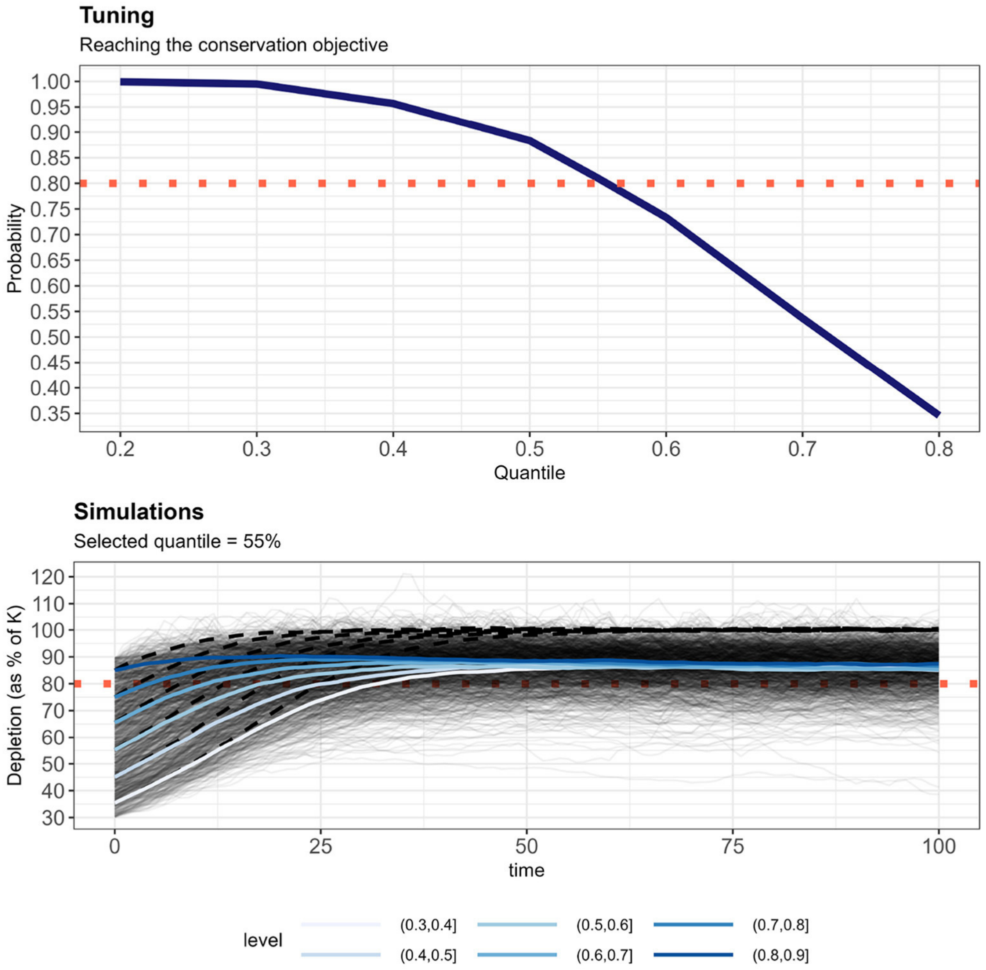


FIGURE 5 | Top panel: Probability to reach the conservation objective (CO) for setting the removals limit as a quantile of the posterior distribution of Equation (2). The 55th quantile is the largest one that allows reaching the CO with a probability of 0.8 after 100 years. Lower panel: All 1, 200 simulations (thick lines: average stratified by initial depletion level) after the implementation of the RLA and removals limit set by using the 55th quantile. The red dotted line shows the 80% of carrying capacity (*K*). The black hashed line shows average population trajectory if anthropogenic removals were eliminated.

quantile of the control rule given by Equation (2) (**Figure 5**). This quantile choice corresponded to an average (across all simulations) removals limit set to 1.3% of the best available abundance estimate, or some 5, 600 animals per year (assuming $K = 500,000$ animals in the simulations). No change in quantile selection was observed when the time horizon for the CO was lowered to 50 years; but for 200 years, the selected quantile was the 50th, resulting in a somewhat lower removals limit (see shiny application).

Robustness Trials

Tuning of the RLA is summarized in **Table 4**. The selected quantile could vary from the 30th to the 80th across the different trials. Trials *C* and *D*, corresponding to scenarios in which removals are underestimated by a factor 2, or abundance is overestimated and removals are underestimated both by a factor 1.5, were the most challenging ones to reach the CO. The 30th percentile choice corresponded to an average (across all simulations) removals limit set to 0.5% of the best available abundance estimate, or some 2, 500 animals per year (assuming $K = 500,000$ animals in the simulations). These results were obtained by averaging over the uncertainty in both the MNPL (on average occurring at 60% of *K*) and the initial depletion level (with an average of 60% of *K*, that is the population being at the MNPL when the RLA is implemented; **Figure 3**).

DISCUSSION

Approaches for setting threshold values for removals of protected cetacean species in the Northeast Atlantic have been extensively discussed (see ICES 2019, page 83), with a focus on three approaches in particular: fixed percentages of abundance, PBR, and the CLA developed by the IWC. Of these, the first is both the simplest and the crudest. Its simplicity translated as a direct,

TABLE 4 | Summary of parameter combination for each robustness trial tested and the resulting quantile.

Robustness trials	ID	Freq.	Horizon	b.ab.	b.byc.	K_{trend}	cata.	quantile
Base case scenario	A0	6	100	1.0	1.0	1.0	0.0	0.55
Lower survey frequency	A1	**12**	100	1.0	1.0	1.0	0.0	0.55
Lower projection horizon	A2	6	**50**	1.0	1.0	1.0	0.0	0.55
Higher projection horizon	A3	6	**200**	1.0	1.0	1.0	0.0	0.50
Abundance overestimation	B0	6	100	**2.0**	1.0	1.0	0.0	0.30
Bycatch underestimation	C0	6	100	1.0	**2.0**	1.0	0.0	0.30
Abundance overestimation + Bycatch underestimation	D0	6	100	**1.5**	**1.5**	1.0	0.0	0.30
Abundance overestimation + Bycatch underestimation + higher survey frequency	D1	**12**	100	**1.5**	**1.5**	1.0	0.0	0.30
Catastrophic events happening	E0	6	100	1.0	1.0	1.0	**0.2**	0.50
Carying capacity degradation	F0	6	100	1.0	1.0	**0.5**	0.0	0.80

ID, scenario identifier; freq., survey frequency; horizon, assumed time horizon for projected the population forward in time during the simulation; b.byc., Bias in bycatch estimates (2 means an underestimation by a factor 2); b.ab., Bias in abundance estimates (2 means an overestimation by a factor 2); K_{trend} *: assumed fraction of the initial K at the end of the simulation; cata., Magnitude (in % of K of a catastrophic event occurring randomly during simulation); quantile, minimum quantile allowing to reach CO. Red values indicate values specific to each trials.*

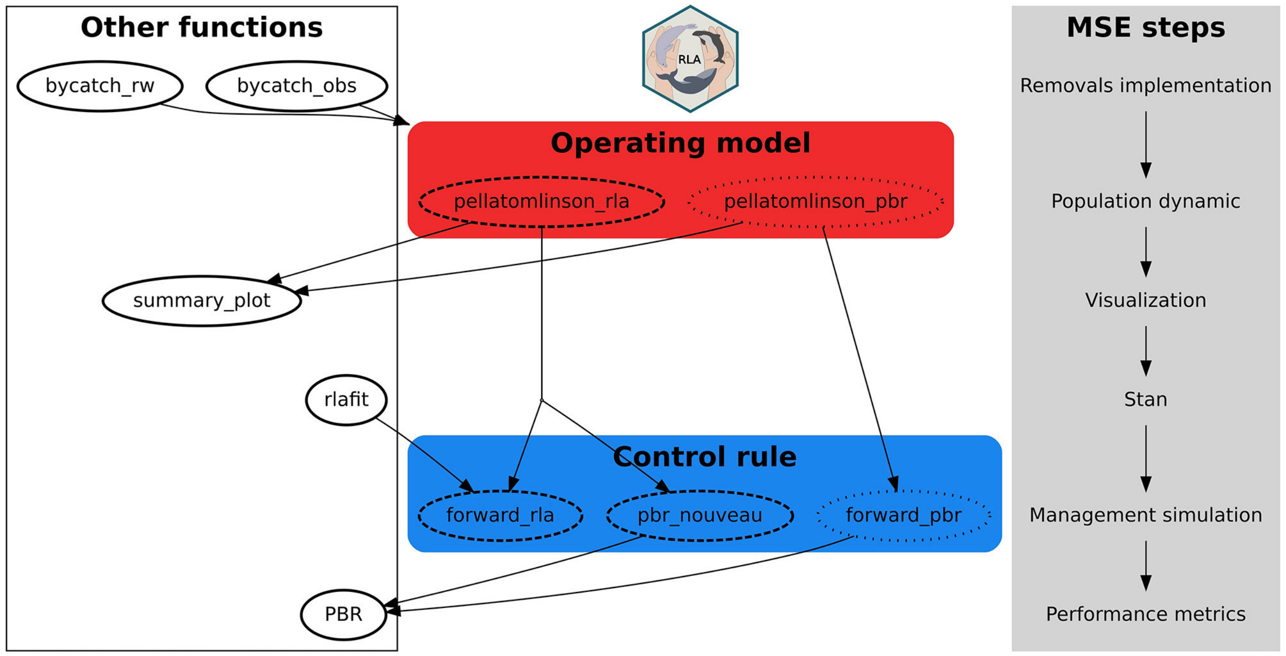


FIGURE 6 | Package structure with the core functions to carry out Management Strategy Evaluation (MSE).

off-the-shelf, availability that permitted its use in ASCOBANS resolutions. Yet, its crudity also resulted in the push-back against the approach from scientists and stakeholders alike (for probably different reasons though). In 2009, ICES advised the European Commission that "a Catch Limit Algorithm approach is the most appropriate method to set limits on the bycatch of harbor porpoises or common dolphins" (ICES, 2013). A practical hurdle to using either the PBR or RLA control rule was the lack of tools to carry out a MSE tailored to the European context where the ASCOBANS interim conservation objective is "to restore and/or maintain stocks/populations to 80% or more of the carrying capacity." We addressed this gap by building an R package to provide scientists with the means of carrying out MSE for setting thresholds on anthropogenic removals of marine mammals. A clear motivation was to remedy the situation seen in the OSPAR Intermediate Assessment 2017 where the recommended approach to setting removals limit could not be implemented.

The RLA package for software R implements utilities to perform the MSE of anthropogenic removals on marine mammal populations (**Figure 6**). The core functions are two data simulators, `pellatomlinson_pbr()` and `pellatomlinson_rla()`, coupled with functions implementing specific control rules: `forward_pbr()` and `forward_rla()` for projecting the population forward in time. Around these core functions gravitates a suite of additional functions (**Figure 6**). The population dynamics simulators are generalized logistic (Pella-Tomlinson) density-dependent models (Punt, 2016) (although other functional forms could be coded and added to the package). Age-aggregated or age-disaggregated versions of the generalized logistic operating model (data simulator) are available. We illustrated the use of these simulators in tuning a modified PBR for small cetaceans (**Figure 4** and **Table 3**) and an RLA for harbor porpoises in the North Sea (**Figure 5** and **Table 4**). In the former case, the simulator is very simple (**Algorithm 1**) and only allows a very coarse conditioning on species- or population-specific information (e.g., R_{max} or F_r; Wade 1998). The age-disaggregated simulator is more involved (**Algorithm 2**) and allows conditioning on species- or population-specific survival and fecundity when these are known. It is also possible to use the age-disaggregated simulator `pellatomlinson_rla()` with the PBR control rule with function `pbr_nouveau()` as in *e.g.* Brandon et al. (2017) or Punt et al. (2020a). This allows conditioning on species- or population-specific survival and fecundity data in the population dynamics model for increased realism, but relying on minimal data (abundance estimates only) to design a precautionary MSE. Both simulators as currently implemented assume a single population of a single species, which can be limiting. Extension to multiple populations of the same species, or multiple species (Punt et al., 2020b; Kanaji et al., 2021; Kaplan et al., 2021) are potential extensions of the operating models within the simulators. In particular, consideration of migration between populations would account for potential sink-source dynamics. This could lead to a more accurate and realistic MSE. This would also need to potentially consider different removals limits for different populations, either to reflect the sink-source distinction or to account for transboundary differences in management if populations are managed by different parties/states or subjected to several fisheries.

Albeit currently limited to single-population, single-species, and single-control rule, the RLA package provides a convenient implementation of population dynamics simulators, which can be leveraged to design a precautionary management strategy by means of robustness trials (Wade, 1998). These robustness trials include consideration of

- systematic bias in abundance or removal estimates;
- random errors in removal estimates, either before or after the implementation of a control rule;
- random catastrophic mortality events killing off a fraction of the population (e.g., epizootics; Aguilar and Raga, 1993);
- a decline in carrying capacity *K* over time;
- environmental stochasticity on population dynamics with a correlated, first order, random walk model;
- differential vulnerabilities to the removal with respect to age.

All these trials can currently be run with the RLA package as exemplified by the two case studies presented herein (**Tables 3**, **4**). They use different control rules, either PBR or RLA, reflecting a difference in the data assumed available for management. The output of simulations is, however, the same as the whole population trajectory under the chosen control rule is available to the user. From this output, additional work is required from the user to analyze the strategy with respect to performance metrics.

Performance metrics with respect to the CO are left to user discretion: there are no special functions in the RLA package to compute specific performance metrics. As the output from the core functions include the whole population trajectory, removals, *etc.* (**Figures 1**, **2**), there is user flexibility for computing performance metrics. In the two case studies, the only performance metric that was assessed was whether the CO was reached after 100 years of implementation of a control rule for computing the removals limit. The probability with which the CO was reached was computed as the frequency of simulations with final depletion $\geq$ 80% of *K* over all simulations. One straightforward additional performance metric is the delay in population recovery with the implementation of the removals limit compared to a counterfactual situation whereby anthropogenic removals were eliminated. This metric was computed for the RLA case study (as shown in Results and associated shiny applications) using the function `time2CO` which requires the user to specify a number of consecutive years for which the CO must have been reached to declare success. This function calls another function `get_streaks` which will identify streaks of 0 and 1 in a vector. The output of a simulation includes a vector named `depletion` which can be used to compute the time to reach the CO as follows:

```
time2CO(ifelse(depletion > 0.8, 1, 0))
```

As each simulation output reports an identifier, which is also a seed that the user can set, it is possible to match the result from a removals limit implementation with the counterfactual outcome under no anthropogenic removals (by using the same seed in both cases). This feature allows performance comparisons under counterfactuals as random number generation remains under user control.

One strength of the RLA package is the enhanced flexibility for users. In addition to defining CO and performance metrics, advanced users familiar with `Stan` syntax can code their own model, or modify the ones stored as text in a list available in the package, to implement different control rules to set a removals limit. The only requirement is to use a parameter or derived quantity from parameters with the name `removal_limit` for function `forward_rla()` to work. For example, the removals limit could change (by hard coding in `Stan` syntax; as shown in Methods above and mc-stan.org) from

```
generated quantities {
real removal_limit;
removal_limit =
 r * fmax(0.0, depletion - IPL);
}
```

to the IWC CLA (e.g., Aldrin et al. 2008; but note that the IWC CLA[5] differs also with respect to Equation 3)

---

[5] https://iwc.int/rmp2

```
generated quantities {
real removal_limit;
removal_limit =
 gamma * r * fmax(0.0, depletion - IPL);
}
```

where the scalar `gamma` must be defined by the user in the `transformed data` block.

As noted by a reviewer, the currently implemented statistical model for estimating current depletion and population growth rate in the RLA control rule (Equation 3) assumes the population is at carrying capacity before the time series of removals starts. Implicitly, this assumes that data on anthropogenic removals are available from the start of human impacts on marine mammal populations. This assumption is likely wrong in many instances, but convenient, as the available time series of removals may not extend back to pristine conditions. The assumption could be relaxed in using an alternative statistical model with an extra parameter on the initial depletion. The latter is not estimable from the abundance and removal data alone, and an informative prior would be needed to ensure identifiability. Alternatively, an additional robustness trial may be considered: in this trial, the removals data would be left-truncated (that is the start of the series would be unavailable to the investigator) and the performance of the currently implemented model assessed. Depending on the results, an alternative model specification may be needed.

The `Stan` engine for Bayesian inference using Hamiltonian Monte Carlo (Carpenter et al., 2017) is versatile and allows to fit a large set of models with efficient algorithms (Monnahan et al., 2017). This versatility may be leveraged by advanced users: function `forward_rla()` needs an object of class "stanmodel" to run. This object is a compiled model that will be repeatedly used within function `forward_rla()`, thus minimizing model compilation time for a faster run. For example, with a time horizon of 100 years and an assumed frequency of 6 years, function `forward_rla()` calls internally $\lceil \frac{100}{6} \rceil = 17$ times the function `sampling` from package `rstan` (Stan Development Team, 2020) in a single simulation. While a single simulation with `forward_rla()` takes a couple of minutes on a laptop, running a large number of simulations quickly becomes prohibitively long. However, computing clusters can be used and resulted in our case of $\approx$ 36 h to run 1,200 simulations (and hence $1,200 \times 17 = 20,400$ calls to `sampling`). Further gains in computation time may be leveraged by taking advantages of parallelization with `Stan` (interested readers can refer to the `Stan` manual: https://mc-stan.org/docs/2_27/stan-users-guide/parallelization-chapter.html).

The use of Hamiltonian Monte Carlo for inference on parameters needed for the RLA is justified as sampling from the joint posterior distribution of parameters of equation (3) can be difficult: parameters r and D_T are often positively correlated. Further work on priors other than independent uniform distribution may help. We plan to explore the use of a joint prior to model the correlation between r and D_T, using for example, a copula (dos Santos Silva and Freitas Lopez, 2008). Further work on the weight w should also be undertaken to assess the sensitivity of RLA to the current choice inherited from the IWC's CLA. The RLA as currently implemented in the package RLA differs from the IWC's CLA: these differences may have consequences that deserve more scrutiny. In particular, we found that increasing the time horizon from 100 to 200 years actually decreased the removals limit, while the reverse was found with the CLA (Aldrin et al., 2008). While surprising and requiring a more in-depth investigation with respect to its cause, this result may currently provide a disincentive to unambitious CO with a long-time horizon to address the issue of unsustainable anthropogenic impacts on marine mammals.

The RLA package is primarily geared toward MSE for setting precautionary limits to anthropogenic removals in marine mammal conservation. The population dynamics simulator provided by **Algorithm 2** may however be harnessed for other uses such as Approximate Bayesian Computation (ABC; Beaumont, 2010; Csilléry et al., 2010). In practice, the generalized logistic model may be difficult to fit directly, and one may resort to likelihood-free methods to carry out inferences on a subset of parameters of interest such as survival ($\boldsymbol{\phi}$), maturity ($\mathbf{m}$), density-dependence (z), or historic removals (assuming, for example, a simple generative model such as Poisson with constant rate). In this case, summary statistics would be the abundance estimates: a rejection algorithm (as available for example in package `abc`; Csillery et al., 2012) can be run using the observed abundance estimates and the simulated ones to infer parameters of interest. Not all parameters may be realistically inferred and some may need to be fixed, or highly informative priors may be needed.

FUTURE APPLICATIONS

We have described the RLA package, which provides a set of functions to carry out the MSE of anthropogenic removals on marine mammals. The two case studies presented were initially carried out under the remit of OMMEG to tune the PBR control rule to the ASCOBANS CO and to continue developing an RLA for harbor porpoise in the North Sea (Hammond et al., 2019). While documentation on MSE was abundant, OMMEG was faced with a dearth of applied tools, which motivated the development of the package. Results obtained and presented need to move through the OSPAR policy process but suggest new default values for the recovery factor F_r with the PBR control rule for small cetaceans and using the 30th quantile with the RLA (Equations 2 and 3) control rule for harbor porpoise to set removals limit in the North Sea. The results for mPBR reported in **Table 3** provide first results on values of the recovery factor F_r for setting removals limit to cetacean bycatch in accordance with the OMMEG interpretation of the ASCOBANS CO. A very conservative choice is $F_r = 0.1$ because of a lack of robustness of mPBR against an MNPL lesser than 0.5. However, such low values of MNPL are implausible for marine mammal populations (Taylor and DeMaster, 1993). More plausible scenarios are those wherein bycatch is underestimated (ICES, 2020a), although assuming an underestimation by a factor 2 maybe be extreme in some cases. Further work on mPBR conditioned on specific contexts within European waters is necessary, especially in considering realistic robustness trials for optimal realism and plausibility.

In the case of the harbor porpoise in the North Sea, the results presented in **Table 4** averaged over a large range of initial depletion. Using the 30th quantile is a very conservative default which can be relaxed in practice when more evidence and information on specific species and areas of interest are available (for example, to narrow down the plausible range of initial depletion). The actual removals limit to be used for example in the next OSPAR assessment for the 2023 Quality Status Report (https://www.ospar.org/work-areas/cross-cutting-issues/qsr2023) needs to be calculated on the best available evidence, including the latest SCANS-III survey abundance estimates (Hammond et al., 2021) and bycatch estimates in the North Sea. This illustrates that implementation of management of bycatch based on removal limits derived from PBR and RLA is dependent on the continuation of cetacean population monitoring programs on a scale commensurate with biological meaningful assessment units (see for example North Atlantic Marine Mammal Commission and the Norwegian Institute of Marine Research 2019 page 13 for assessment units of harbor porpoises). A modified PBR tuned to OMMEG's interpretation of the ASCOBANS CO, namely "a population should be able to recover to or be maintained at 80% of carrying capacity, with probability 0.8, within a 100-year period" will address a current misalignment between management and conservation objectives in the salient context of small cetacean conservation in the Northeast Atlantic (ICES, 2020c). It is our hope that the RLA package will enable easier MSE, in particular in the current EU MSFD context of achieving "Good Environmental Status." This hope crucially hinges on users' feedback and involvement in further developing and expanding the package to the benefit of improved management of the impact of human activities on marine mammals.

DATA AVAILABILITY STATEMENT

All codes and datasets used in this study are available at https://gitlab.univ-lr.fr/pelaverse/rla_paper.

AUTHOR CONTRIBUTIONS

MG and MA led the analyses, the conception, and writing of the paper and package. All authors supported in analyses, paper conception, writing and contributed to the article and approved the submitted version.

FUNDING

ADERA provided support in the form of salaries for MA but did not have any additional role in the study design, data collection and analysis, decision to publish, or preparation of the manuscript.

ACKNOWLEDGMENTS

AG thanks the German Federal Agency for Nature Conservation for funding her time as the lead of the OSPAR expert group on marine mammals. MA and MG thank Mikaël Guichard and the dedicated staff at the Mésocentre de Calcul de Poitou Charentes. We thank reviewers, Finn Larsen, Lotte Kindt-Larsen, and OMMEG members for discussion and critical comments.

REFERENCES

Aguilar, A., and Raga, J. A. (1993). The stripped dolphin epizootic in the Mediterranean sea. *Ambio* 22, 524–528.

Aldrin, M., Husevy, R. B., and Schweder, T. (2008). A note on tuning the catch limit algorithm for commercial baleen whaling. *J. Cetacean Manag. Res*. 10, 191–194.

Anonymous (2008). Directive 2008/56/EC of the European Parliament and of the Council of 17 June 2008 establishing a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive) (Text with EEA relevance). Official Journal of the European Union. 2008/56/EC.

Anonymous (2010). Letter from the european commission to ices, dated 29 july 2010. ref. ares(2010)830555 - 18/11/2010. Letter.

ASCOBANS (2000). Annex O - Report of the IWC-ASCOBANS working group on harbour porpoises. *J. Cetacean Res. Manag*. 2(Suppl.):297–305.

Authier, M., Galatius, A., Gilles, A., and Spitz, J. (2020). Of power and despair in cetacean conservation: estimation and detection of trend in abundance with noisy and short time-series. *PeerJ* 8:e9436. doi: 10.7717/peerj.9436

Avila, I. C., Kaschner, K., and Dormann, C. F. (2018). Current global risks to marine mammals: taking stock of the threats. *Biol. Conserv*. 221, 44–58. doi: 10.1016/j.biocon.2018.02.021

Beaumont, M. (2010). Approximate bayesian computation in evolution and ecology. *Ann. Rev. Ecol. Evolut. Syst*. 41, 379–406. doi: 10.1146/annurev-ecolsys-102209-144621

Boyce, M. S. (2000). *Quantitative Methods for Conservation Biology, Chapter Whaling Models for Cetacean Conservation*. New York, NY: Springer, 109–126.

Brandon, J. R., Punt, A. E., Moreno, P., and Reeves, R. R. (2017). Toward a tier system approach for calculating limits on human-caused mortality of marine mammals. *ICES J. Mar. Sci*. 74, 877–887. doi: 10.1093/icesjms/fsw202

Brownell Jr, R., Reeves, R., Read, A., Smith, B., Thomas, P., Ralls, K., et al. (2019). Bycatch in gillnet fisheries threatens critically endangered small cetaceans and other aquatic megafauna. *Endanger. Species Res*. 40, 285–296. doi: 10.3354/esr00994

Bunnefeld, N., Hoshino, E., and Milner-Gulland, E. J. (2011). Management strategy evaluation: a powerful tool for conservation? *Trends Ecol. Evol*. 26, 441–447. doi: 10.1016/j.tree.2011.05.003

Carpenter, B., Gelman, A., Hoffman, M. D., Lee, D., Goodrich, B., Betancourt, M., et al. (2017). *Stan*: a probabilistic programming language. *J. Stat. Softw*. 76, 1–32. doi: 10.18637/jss.v076.i01

Chang, W., Cheng, J., Allaire, J., Sievert, C., Schloerke, B., Xie, Y., et al. (2021). shiny: Web Application Framework for R. *R package version 1.6.0*.

Cooke, J. (1994). The management of whaling. *Aquatic Mamm*. 20, 129–135.

Cooke, J. G. (1999). Improvement of fishery-management advice through simulation testing of harvest algorithms. *ICES J. Mar. Sci*. 797–810. doi: 10.1006/jmsc.1999.0552

Csilléry, K., Blum, M., Gaggiotti, O., and François, F. (2010). Approximate bayesian computation (ABC) in practice. *Trends Ecol. Evol*. 25, 410–418. doi: 10.1016/j.tree.2010.04.001

Csillery, K., Francois, O., and Blum, M. G. B. (2012). abc: an r package for approximate bayesian computation (abc). *Methods Ecol. Evolut*. 3, 475–479. doi: 10.1111/j.2041-210X.2011.00179.x

dos Santos Silva, R., and Freitas Lopez, H. (2008). Copula, marginal distributions and model selection: a bayesian note. *Stat. Comput*. 18, 313–320. doi: 10.1007/s11222-008-9058-y

Forney, K. A., Moore, J. E., Barlow, J., Carretta, J. V., and Benson, S. R. (2021). A multidecadal bayesian trend analysis of harbor porpoise (*Phocoena phocoena*) populations off california relative to past fishery bycatch. *Mar. Mamm. Sci*. 37, 546–560. doi: 10.1111/mms.12764

Gerrodette, T. (1987). A power analysis for detecting trends. *Ecology* 68, 1364–1372. doi: 10.2307/1939220

Gray, C. A., and Kennelly, S. J. (2018). Bycatches of endangered, threatened and protected species in marine fisheries. *Rev. Fish. Biol. Fish*. 28, 521–541. doi: 10.1007/s11160-018-9520-7

Hammond, P. S., Lacey, C., Gilles, A., Viquerat, S., Börjesson, P., Herr, H., et al. (2021). *Estimates of Cetacean Abundance in European Atlantic Waters in Summer 2016 from the SCANS-III Aerial and Shipboard Surveys*. Technical report, Sea Mammal Research Unit, University of St Andrews, UK.

Hammond, P. S., Paradinas, I., and Smout, S. C. (2019). *Development of a Removals Limit Algorithm (RLA) to Set Limits to Anthropogenic Mortality of Small Cetaceans to Meet Specified Conservation Objectives. JNCC Report No. 628*. (Peterborough: JNCC). Available online at: https://data.jncc.gov.uk/data/8ac9a424-eda5-4062-957e-63d82d3e39cc/JNCC-Report-628-FINAL-WEB.pdf

Hilborn, R., and Mangel, M. (1997). *The Ecological Detective. Confronting Models With Data, 1st Edn*. Princeton, NJ: Princeton University Press.

ICES (2009). *Report of the Study Group for Bycatch of Protected Species (SGBYC)*. Technical Report, ICES CM 2009/ACOM:22. 19-22 January 2009, Copenhagen, Denmark, 117.

ICES (Ed.). (2010). *General Advice EC Request on Cetacean Bycatch Regulation 812/2004, Item 3*. Copenhagen: International Council for the Exploration of the Sea.

ICES (2013). *Report from the Working Group on Bycatch of Protected Species*, (*WGBYC)*. Technical Report No. ICES CM 2013/ACOM:27, International Council for the Exploration of the Sea, 4–8 February, Copenhaguen, 73.

ICES (Ed.). (2019). *Report of the Working Group on Marine Mammal Ecology (WGMME)*, volume 11–14 February 2019, ICES Scientific Reports. 1:22, Büsum, International Council for the Exploration of the Sea, 131.

ICES (2020a). *Report from the Working Group on Bycatch of Protected Species*, (*WGBYC)*. Tech report 2:81, International Council for the Exploration of the Sea, 209.

ICES (Ed.). (2020b). *Report of the Working Group on Marine Mammal Ecology (WGMME)*, volume 10-14 February 2020, Barcelona, ICES Scientific Reports. 2:39. International Council for the Exploration of the Sea, 85.

ICES (2020c). *Workshop on fisheries Emergency Measures to minimize BYCatch of short-beaked common dolphins in the Bay of Biscay and harbour porpoise in the Baltic Sea (WKEMBYC)*. Resreport 2:43, International Council for the Exploration of the Sea. 354.

Johnson, A. F., Caillat, M., Verutes, G. M., Peter, C., Junchompoo, C., Long, V., et al. (2016). Poor Fisheries Struggle with U.S. *Import Rule Sci*. 355, 1031–1032. doi: 10.1126/science.aam9153

Kanaji, Y., Maeda, H., Okamura, H., Punt, A. E., and Branch, T. (2021). Multiple-model stock assessment frameworks for precautionary management and conservation on fishery-targeted coastal dolphin populations off Japan. *J. Appl. Ecol*. 58, 2479–2492. doi: 10.1111/1365-2664.13982

Kaplan, I. C., Gaichas, S. K., Stawitz, C. C., Lynch, P. D., Marshall, K. N., Deroba, J. J., et al. (2021). Management strategy evaluation: allowing the light on the hill to illuminate more than one species. *Front. Mar. Sci*. 8:664355. doi: 10.3389/fmars.2021.624355

Korpinen, S., Laamanen, L., Bergström, L., Nurmi, M., Andersen, J. H., Haapaniemi, J., et al. (2021). Combined effects of human pressures on EuropeâĂŹs marine ecosystems. *Ambio* 50, 1325–1336. doi: 10.1007/s13280-020-01482-x

Mangel, M., Talbot, L. M., Meffe, G. K., Agardy, M. T., Alverson, D. L., Barlow, J., et al. (1996). Principles for the conservation of wild living resources. *Ecol. Appl*. 6, 338–362. doi: 10.2307/2269369

Monnahan, C. C., Thorson, J. T., and Branch, T. A. (2017). Faster estimation of bayesian models in ecology using hamiltonian monte carlo. *Methods Ecol. Evolut*. 8, 339–348. doi: 10.1111/2041-210X.12681

North Atlantic Marine Mammal Commission and the Norwegian InstituteNorth Atlantic Marine Mammal Commission and the Norwegian Institute of Marine Research (2019). *Report of Joint IMR/NAMMCO International Workshop on the Status of Harbour Porpoises in the North Atlantic*. Technical report, Tromsøo.

Pielke R. A. Jr. (2007). *The Honest Broker - Making Sense of Science in Policy and Politics, 1st Edn*. Cambridge: Cambridge University Press, 198 ISBN: 9780521873208.

Punt, A. E. (2006). The FAO precautionary approach after almost 10 years: have we progressed towards implementing simulation-tested feedback-control management systems for fisheries management? *Natural Resour. Model*. 19, 441–464. doi: 10.1111/j.1939-7445.2006.tb00189.x

Punt, A. E. (2016). *Review of Contemporary Cetacean Stock Assessment Models*. Technical report, School of Aquatic and Fishery Sciences, University of Washington, Seattle, WA.

Punt, A. E., Moreno, P., Brandon, J. R., and Mathews, M. A. (2018). Conserving and recovering vulnerable marine species: a comprehensive evaluation of the US approach for marine mammals. *ICES J. Mar. Sci*. 75, 1813–1831. doi: 10.1093/icesjms/fsy049

Punt, A. E., Siple, M., Francis, T. B., Hammond, P. S., Heinemann, D., Long, K. J., et al. (2020a). Robustness of potential biological removal to monitoring, environmental, and management uncertainties. *ICES J. Mar. Sci*. 77, 2491–2507. doi: 10.1093/icesjms/fsaa096

Punt, A. E., Siple, M., Francis, T. B., Hammond, P. S., Heinemann, D., Long, K. J., et al. (2021). Can we manage marine mammal bycatch effectively in low-data environments? *J. Appl. Ecol*. 58, 596–607. doi: 10.1111/1365-2664.13816

Punt, A. E., Siple, M., Sigurðsson, G. M., Víkingsson, G., Francis, T. B., Granquist, S. M., et al. (2020b). Evaluating management strategies for marine mammal populations: an example for multiple species and multiple fishing sectors in Iceland. *Can. J. Fish. Aquatic Sci*. 77, 1316–1331. doi: 10.1139/cjfas-2019-0386

R Core Team (2020). *R: A Language and Environment for Statistical Computing*. Vienna: R Foundation for Statistical Computing.

Rayner, S. (2012). Uncomfortable knowledge: the social construction of ignorance in science and environmental policy discourses. *Econ. Soc*. 41, 107–125. doi: 10.1080/03085147.2011.637335

Reeves, R. R., McClellan, K., and Werner, T. B. (2013). Marine mammal bycatch in gillnet and other entangling net fisheries, 1990 to 2011. *Endanger. Species Res*. 20, 71–97. doi: 10.3354/esr00481

Rogan, E., Read, A. J., and Berggren, P. (2021). Empty promises: the european union is failing to protect dolphins and porpoises from fisheries by-catch. *Fish Fish*. 22, 865–869. doi: 10.1111/faf.12556

Siple, M. C., Punt, A. E., Francis, T. B., Hammond, P. S., Heinemann, D., Long, K. J., et al. (2021). mmrefpoints: Projecting long-term marine mammal abundance with bycatch. *R package version 1.0.0*.

Stan Development Team (2020). RStan: the R interface to Stan. *R package version 2.21.22*.

Taylor, B., Martinez, M., Gerrodette, T., Barlow, J., and Hrovat, Y. (2007). Lessons from monitoring trends in abundance of marine mammals. *Mar. Mamm. Sci*. 23, 157–175. doi: 10.1111/j.1748-7692.2006.00092.x

Taylor, B. L., and DeMaster, D. P. (1993). Implications of non-linear density dependence. *Mar. Mamm. Sci*. 9, 360–371. doi: 10.1111/j.1748-7692.1993.tb00469.x

Wade, P. R. (1998). Calculating limits to the total allowable human-caused mortality of cetaceans and pinnipeds. *Mar. Mamm. Sci*. 14, 1–37. doi: 10.1111/j.1748-7692.1998.tb00688.x

Wade, P. R., Long, K. J., Francis, T. B., Punt, A. E., Hammond, P. S., Heinemann, D., et al. (2021). Best practices for assessing and managing bycatch of marine mammals. *Front. Mar. Sci*. 8:1566. doi: 10.3389/fmars.2021.757330

Williams, R., Burgess, M. G., Ashe, E., Galnes, S. D., and Reeves, R. (2016). U.S. Seafood import restriction presents opportunity and risk. *Science* 354, 1372–1374. doi: 10.1126/science.aai8222

Williams, R., Hall, A., and Winship, A. (2008). Potential limits to anthropogenic mortality of small cetaceans in coastal waters of British

Columbia. *Can. J. Fish. Aquatic Sci*. 65, 1867–1878. doi: 10.1139/F08-098

Winship, A. J. (2009). *Estimating the Impact of Bycatch and Calculating Bycatch Limits to Achieve Conservation Objectives as Applied to Harbour Porpoises in the North Sea* (Ph.D. thesis). School of Biology, University of St Andrews.

Conflict of Interest: MA is employed by the commercial company ADERA which did not play any role in this study beyond that of employer. AW is employed by the commercial company CSS, Inc. KM is employed by the commercial company HiDef.

The remaining authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

frontiers
in Marine Science

ORIGINAL RESEARCH
published: 04 January 2022
doi: 10.3389/fmars.2021.791484

Precautionary Principle or Evidence-Based Conservation? Assessing the Information Content of Threat Data for the Yangtze Finless Porpoise

Lisa M. W. Mogensen[1,2], Zhigang Mei[3], Yujiang Hao[3], Xavier A. Harrison[4], Ding Wang[3] and Samuel T. Turvey[1]*

[1] Institute of Zoology, Zoological Society of London, London, United Kingdom, [2] Department of Genetics, Evolution and Environment, University College London, London, United Kingdom, [3] Key Laboratory of Aquatic Biodiversity and Conservation, Institute of Hydrobiology, Chinese Academy of Sciences, Wuhan, China, [4] Biosciences, College of Life and Environmental Sciences, University of Exeter, Exeter, United Kingdom

Conservation management requires evidence, but robust data on key parameters such as threats are often unavailable. Conservation-relevant insights might be available within datasets collected for other reasons, making it important to determine the information content of available data for threatened species and identify remaining data-gaps before investing time and resources in novel data collection. The Yangtze finless porpoise (*Neophocaena asiaeorientalis asiaeorientalis*) has declined severely across the middle-lower Yangtze, but multiple threats exist in this system and the relative impact of different anthropogenic activities is unclear, preventing identification of appropriate mitigation strategies. Several datasets containing information on porpoises or potential threats are available from past boat-based and fishing community surveys, which might provide novel insights into causes of porpoise mortality and decline. We employed multiple analytical approaches to investigate spatial relationships between live and dead porpoises and different threats, reproductive trends over time, and sustainable offtake levels, to assess whether evidence-based conservation is feasible under current data availability. Our combined analyses provide new evidence that mortality is spatially associated with increased cargo traffic; observed mortality levels (probably a substantial underestimate of true levels) are unsustainable; and population recruitment is decreasing, although multiple factors could be responsible (pollutants, declining fish stocks, anthropogenic noise, reduced genetic diversity). Available data show little correlation between patterns of mortality and fishing activity even when analyzed across multiple spatial scales; however, interview data can be affected by multiple biases that potentially complicate attempts to reconstruct levels of bycatch, and new data are required to understand dynamics and sustainability of porpoise-fisheries interactions. This critical assessment of existing data thus suggests that *in situ* porpoise conservation management must target multiple co-occurring threats. Even limited available datasets can provide new insights for understanding declines, and

Edited by:
Jeff Moore,
Southwest Fisheries Science Center
(NOAA), United States

Reviewed by:
Xuming Zhou,
Institute of Zoology, Chinese
Academy of Sciences (CAS), China
Tom Jefferson,
Clymene Enterprises, United States

***Correspondence:**
Samuel T. Turvey
samuel.turvey@ioz.ac.uk

Specialty section:
This article was submitted to
Marine Megafauna,
a section of the journal
Frontiers in Marine Science

Received: 08 October 2021
Accepted: 08 December 2021
Published: 04 January 2022

Citation:
Mogensen LMW, Mei Z, Hao Y,
Harrison XA, Wang D and Turvey ST
(2022) Precautionary Principle or
Evidence-Based Conservation?
Assessing the Information Content
of Threat Data for the Yangtze Finless
Porpoise. Front. Mar. Sci. 8:791484.
doi: 10.3389/fmars.2021.791484

we demonstrate the importance of an integrative approach for investigating complex conservation problems and maximizing evidence in conservation planning for poorly known taxa.

Keywords: cetacean, conservation effectiveness, evidence-based conservation, interview survey, precautionary principle, sustainable offtake, uncertainty

INTRODUCTION

Conservation practitioners recognize the need for evidence-based conservation, where robust data are used to understand the dynamics of decline and guide best-practice management (Sutherland et al., 2004; Bower et al., 2018). However, data for many threatened species are limited in quantity and/or quality, hindering informed decision-making (Catullo et al., 2008; McDonald-Madden et al., 2008; Lindenmayer et al., 2013). In such cases, precautionary conservation is often applied (Carr and Raimondi, 1999; Pan and Huntington, 2015; Sampaio et al., 2015). This approach deals with uncertainty using defensive intervention, and typically advocates action even in data-poor contexts (Cooney, 2004). However, precautionary conservation can increase the risk of suboptimal outcomes and inefficient use of time and resources, as interventions are more poorly informed and might not target key problems effectively (VanderWerf et al., 2006). Further research can overcome data limitation, but requires investment in resources that could be allocated to more practical activities, can yield diminishing investment returns or risk replicating collection of existing data (Grantham et al., 2008), and can generate delays that reduce the guarantee of improved conservation prospects (Jaramillo-Legorreta et al., 2007). Alternatively, conservation-relevant insights might be provided through analysis of existing datasets originally collected for other reasons (McDonald-Madden et al., 2008; Zhang and Vincent, 2017). It is thus important to determine the information content of available data on threatened species before investing in novel data collection. This may involve multiple analyses of limited data to extract maximally useful conservation baselines and identify remaining data-gaps (MacMillan and Marshall, 2006; Thieme et al., 2007; Rodrigues, 2011).

The Yangtze finless porpoise (*Neophocaena asiaeorientalis asiaeorientalis*) is a freshwater cetacean endemic to the middle-lower Yangtze drainage in eastern China, occurring in the 1,700 km river mainstem between Yichang-Shanghai and in the appended Dongting and Poyang lakes. It is usually considered a freshwater subspecies of narrow-ridged finless porpoise, but has recently been proposed as a recently diverged distinct species on the basis of its genetic adaptation to freshwater conditions and complete genetic isolation from marine finless porpoise populations (Zhou et al., 2018). The Yangtze River experienced the extinction of the Yangtze River dolphin or baiji (*Lipotes vexillifer*) in the early 2000s (Turvey et al., 2007), and the porpoise population has also declined severely, falling in the mainstem from ∼2,700 in the 1990s (Zhang et al., 1993) to ∼500 in 2012 (Mei et al., 2014). It is now one of the few cetaceans listed as Critically Endangered by IUCN (2021). The middle-lower Yangtze drainage is heavily industrialized, supports extremely large human populations, and is impacted by numerous anthropogenic pressures including intensive legal and illegal fishing activities (free-floating nets, hook-based gears such as rolling-hook long-lines, fixed nets, and electrofishing), high vessel traffic, pollution, resource depletion, and habitat loss/degradation. These activities are all proposed as potential drivers of porpoise decline through elevated direct mortality, reduced survivorship, or reduced carrying capacity (Zhao et al., 2008; Wang, 2009; Mei et al., 2014; Huang et al., 2017).

The relative impact of different potential porpoise threats has been the subject of limited research. Previous studies consist largely of investigating local-scale porpoise distributions in relation to available resources (Kimura et al., 2012; Wang et al., 2014), and opportunistically-observed porpoise deaths associated with specific threats including fisheries bycatch, propeller collisions, and pollutants (Zhou and Wang, 1994; Wang et al., 2000, 2015; Dong et al., 2006; Yang et al., 2008; Turvey et al., 2013; Xiong et al., 2019). Two population viability analyses have been conducted to model future decline (Zhang and Wang, 1999; Huang et al., 2017); however, neither attempted to quantify sustainable levels of offtake (the number of individuals removed from the environment through hunting, harvesting, or other killing by humans; cf. Ingram et al., 2015), or incorporated mortality data associated with different threats. The only population-level study into drivers of decline used a mortality dataset derived from interviews with artisanal fishers (Turvey et al., 2013). This study suggested that propeller-related mortality has increased over time and might be driving decline, whereas bycatch mortality has decreased over time and might therefore merely be tracking decline. However, the relative significance of only these two threats could be assessed, and most deaths could not be attributed to known causes.

Rapid, effective action is needed to prevent Yangtze finless porpoise extinction. However, mitigation of different threats requires different approaches, but in the absence of robust data on primary driver(s) of decline, it is difficult to identify priority management targets. Indeed, in ecosystems where cetaceans face multiple threats, each threat might have reduced individual risk, resulting in reduced power to identify drivers of decline (Thompson et al., 2000). A precautionary approach has been generally recommended for cetacean conservation, especially for threatened species lacking robust data on causes of decline (Thompson et al., 2000), and precautionary measures have been initiated for Yangtze finless porpoise conservation. Porpoises are awarded national-level protection under China's 1989 Wildlife Protection Act, several *in situ* and *ex situ* reserves have been established (with regulation of boat traffic within some reserves), and seasonal fishing bans have been implemented across the middle-lower Yangtze since 2003 to mitigate against bycatch and fish stock depletion (Wang, 2009), based on the assumption that fisheries impacts are an important driver of decline (Zhou and Wang, 1994; Wang et al., 1998; Zhao et al., 2008). However, existing *in situ* management potentially risks further declines if other factors are primarily responsible for regulating porpoise population dynamics (cf. Rojas-Bracho et al., 2019).

Several datasets containing information on porpoises and/or potential threats are available for the Yangtze system, including from boat-based surveys in 2006 and 2012 that followed identical survey protocols across the same area and same time of year (November–December) (Turvey et al., 2007; Zhao et al., 2008; Mei et al., 2014), and interview surveys in artisanal fishing communities in 2008 and 2011–2012 (Turvey et al., 2013, 2015a). These datasets date from a period when the porpoise population was experiencing a rapid decline (Mei et al., 2014), and so might be able to provide important insights into the drivers of this decline. We investigated the information content of this existing evidence-base using multiple analytical approaches, to determine whether available datasets can strengthen our baseline for understanding primary causes of porpoise mortality, investigate relative impacts of different possible threats, identify remaining data-gaps, and assess whether evidence-based porpoise conservation is feasible under current data availability or whether a more precautionary approach is justified. Our study reveals novel insights into the dynamics and probable drivers of porpoise population depletion, and illustrates an approach to maximize evidence in conservation planning and prioritization for highly threatened taxa.

MATERIALS AND METHODS

Spatial Mortality and Threat Patterns

100 km-Section Analysis

Spatial data on counts of live porpoise sightings, cargo vessels and fishing vessels were collected along the Yangtze mainstem (Yichang-Shanghai) during the 2006 boat-based survey (Turvey et al., 2007; Zhao et al., 2008; **Figure 1**). Spatial data on reported porpoise deaths and locally used fishing gear types were collected across the same area during the 2008 interview survey of 599 respondents in 27 fishing communities (see Turvey et al., 2013 for survey methods and questionnaire design). Associated information reported by respondents on dates and locations of specific dead porpoise observations indicates that these all represented independent mortality events, rather than multiple reports of the same animals (Turvey et al., 2013). To investigate relationships between reported mortality levels and spatial distributions of different threats on a Yangtze-wide scale, data from both surveys were grouped into 17,100 km river sections, to control for spatially uneven distribution of fishing communities and match study design in Turvey et al. (2013) (**Supplementary Table 1**). The furthest downstream section was excluded from analysis because respondents here fished partly at sea, so reports likely also refer to marine narrow-ridged finless porpoises (*Neophocaena asiaeorientalis sunameri*) and marine-type gears.

The relationship between mortality levels and threats in the remaining 16,100 km sections was investigated using a binomial generalized linear model (GLM) framework in R v3.4.3 (R Development Core Team, 2017). Proportion of respondents who had seen a dead porpoise during the previous 12 months was used as the response variable; data on numbers of dead porpoise sightings per respondent were too imprecise to analyze, as most responses were qualitative (e.g., "a few") rather than quantitative. Predictor variables included counts of cargo vessels, fishing vessels and live porpoises, and proportions of respondents who reported using different functional fishing gear types (free-floating nets, fixed nets, hook-type gears), and who thought electrofishing was locally a problem. Vessel counts had much larger values and ranges than other variables, so were individually rescaled to a mean of 0 and standard deviation of 1. An indirect electrofishing metric was necessary because this activity is illegal and heavily penalized, so direct reporting data are prone to bias by omission.

Data were over-dispersed and so were fitted in a beta-binomial framework, which takes sample size information into account rather than simply using raw proportion data (Harrison, 2015). Models were ranked by Akaike Information Criterion corrected for small sample size (AICc). We considered all models within $\Delta 6$ units of the top model as competitive, but also applied the nesting rule (Richards, 2008) to remove models that were more complex versions of those with better AICc support and so likely to contain uninformative parameters. We applied model averaging to this best-model set. As this dataset is relatively small ($n = 16$ replicates), data were also fitted into a logit Gaussian framework to check robustness and directionality of significant predictors from the beta-binomial model. Influence of spatial autocorrelation on final models was discounted (**Supplementary Material**).

County-Level Analysis

Additional data on reported porpoise deaths and fishing activities were collected from all mainstem riverside counties between Huangshi-Dongzhi (∼270 km, 13 counties, 205 respondents) and all Poyang Lake counties within the distribution of the lake's porpoise population (6 counties, 201 respondents) during the 2011–2012 interview survey (**Figure 1**; see Turvey et al., 2015a for survey methods and questionnaire design), allowing investigation into predictors of mortality at finer-scale resolution (**Supplementary Table 2**).

County-level mortality and fishing activity patterns were again investigated within a binomial GLM framework with models ranked by AICc, using proportion of respondents who had seen a dead porpoise during the previous 12 months as the response variable, and the same four functional gear categories analyzed above as predictors. The indirect electrofishing metric used here was mean percentage/county based on the question "What percentage of this village practices electrofishing?" with random values calculated from reported ranges (e.g., "10–30%"), and excluding vague responses. County-level count data of cargo and fishing vessels and live porpoises were unavailable for Poyang Lake, so could not be included as predictors. The model was over-dispersed within a binomial framework and

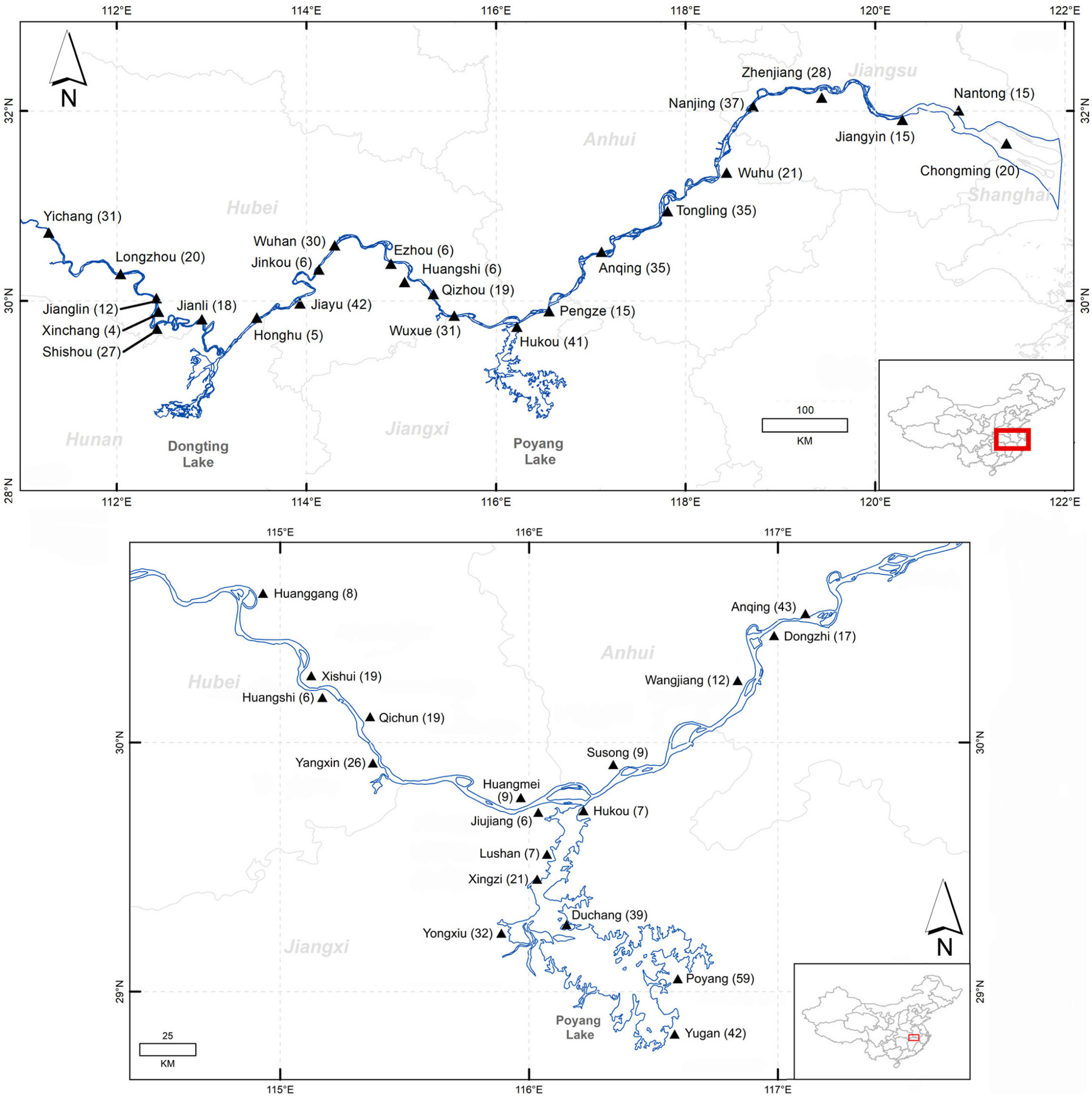


FIGURE 1 | (Top) Middle-lower Yangtze drainage (Yichang-Shanghai), showing 2008 fisher survey localities and number of interviews. **(Bottom)** 2011–2012 fisher survey localities and number of interviews.

was not successfully corrected with a quasibinomial model, so a logit Gaussian model that successfully fitted the data was also used, with logit transformation of response variable. Spatial autocorrelation was not detected in any model parameters (**Supplementary Material**).

Across-River Overlap of Porpoises and Fishing Gear

Along-river analyses were complemented by investigation of spatial correlation of porpoises and threats over an across-river transect. Porpoise sightings from both boat-based surveys between Ezhou-Zhenjiang, the ~650 km mainstem section containing ~80% of all sightings (Zhao et al., 2008; Mei et al., 2014; **Figure 1**), have been categorized by distance from bank into four distance bins (0–100, 101–300, 301–500 m, >500 m) (unpublished data). Fishers interviewed in 2011–2012 between Huangshi-Dongzhi (all located between Ezhou-Zhenjiang) were asked the distance from bank they used different gears; these data were categorized into the same distance bins used for porpoise data. Individual gear-use responses were assigned to all bins within a reported range (e.g., "100–150 m" was assigned to 0–100 and 101–300 m bins). Only free-floating nets and hook-type gears were analyzed, as reported use of other gears was very low. Chi-squared goodness-of-fit tests and Spearman rank correlation were used to investigate whether porpoise observations and gears showed similar distributions.

Reproductive Patterns Over Time

Boat-based surveys recorded whether observations were adults or calves (2006, 348 adults, 90 calves; 2012, 318 adults, 23 calves; unpublished data). To investigate whether reproduction has been uniform over time, we used a binary GLM to quantify differences in the proportion of calves in each survey year, where individuals were coded as 1 if calf and 0 if adult. We fitted year as the sole predictor in a GLM with a binomial error structure, and assessed significance by likelihood ratio test against a model containing only the intercept.

Estimating Sustainable Offtake

Levels of offtake that will maintain or achieve a sustainable Yangtze finless porpoise population were estimated using a Potential Biological Removal (PBR) model and a logistic population growth model, two population models commonly used to predict cetacean population change (Wade, 1998; Monnahan et al., 2015). Theoretical removal limits were then compared with numbers of dead porpoises reported during the 2008 interview survey, to assess whether direct mortality and known causes of mortality could have driven population decline.

Potential Biological Removal Model

PBR values were calculated using the formula in Wade (1998):

$$PBR = N_{min}\, 0.5 R_{max}\, F_r$$

where N_{min} = 20th percentile of estimated population size, R_{max} = maximum annual population growth rate, and F_r = recovery factor.

N_{min} is calculated by:

$$N_{min} = O_{abs} \exp\left[Z \sqrt{\log(1 + CV_{abs}^2}\right]$$

where O_{abs} = survey estimate of absolute abundance, Z = standard normal deviate, and CV_{abs} = population estimate's coefficient of variation.

Values were calculated across several populations and parameters: (1) using 2006 and 2012 population estimates, with a coefficient of variation (CV_{abs}) of 0.133 for 2006 and 0.159 for 2012 (Zhao et al., 2008; Mei et al., 2014); (2) using maximum annual population growth rates (R_{max}) of 0.04 for Indo-Pacific finless porpoise (*Neophocaena phocaenoides*; Taylor et al., 2007), and 0.035 for a semi-wild translocated Yangtze finless porpoise population (Huang et al., 2017); and (3) using mainstem-only and mainstem+lakes population estimates. A default recovery factor (F_r) value of 0.1 was used (Wade and Angliss, 1997), and the standard normal deviate Z was fixed at -0.842 (Wade, 1998). N_{min} (20th percentile of estimated population size) was calculated specifically for each estimate.

Logistic Growth Model

Logistic population growth for 2006–2012 was modeled in an Ordinary Differential Equation framework in R v3.4.3 with the DeSolve package (Soetaert et al., 2010), using:

$$N_{t+1} = N_t \left[1 + R_{max}\left(1 - \frac{N_t}{K}\right)\right] - C_t$$

where N = population estimate in any given year, t = year, K = population size at carrying capacity, and C_t = removal rate in year t, and with annual additions (A) to the population determined by:

$$A = \frac{R_{max}\, N}{K}\,(K - N)$$

The values of state, r, K, and R_{max} were modified for each specific model. Identical R_{max} values were used as for PBR analysis. In the absence of known carrying capacity values, highest recorded population estimates were used as proxies for population size at carrying capacity (K): mainstem = 2,546, Dongting Lake = 50, Poyang Lake = 450, total = 3,146 (Zhang et al., 1993; Zhao et al., 2008; Mei et al., 2014). Range-wide population estimates were 1,800 individuals in 2006 (Zhao et al., 2008) and 1,040 individuals in 2012 (Mei et al., 2014). Models were re-run using multiple values for starting population size and K: (1) range-wide model with $N_{(2006)}$ = 1,800 and K = 3,146; (2) mainstem-only with $N_{(2006)}$ = 1,225 and K = 2,546; (3–4) models 1 and 2 re-run to estimate maximum sustainable removal rates to maintain 2006 and 2012 range-wide and mainstem-only population estimates; (5–6) models 1 and 2 re-run to estimate actual removal rates required for observed range-wide and mainstem-only declines.

RESULTS

For 100 km-section mainstem analysis, beta-binomial best-fit models and logit Gaussian model-averaged estimates both indicate that cargo vessel and live porpoise counts are significant positive predictors of reported mortality (**Tables 1**, **2**). Fishing vessel density is also included in the final beta-binomial and logit Gaussian model sets as a negative predictor, but 95% confidence intervals include 0 in logit Gaussian model-averaged estimates, providing weaker support for a possible relationship. Best-fit models do not include other fishing activity indices. The null (intercept-only) model is included in both beta-binomial best-fit models and logit Gaussian model-averaged estimates.

For county-level analysis, model outputs indicate that mortality is not predicted by any fishing-based predictors. Final model selection includes only the null model, with all other model structures nested and discounted (**Supplementary Table 3**).

Neither porpoises nor gear categories are distributed evenly across nearshore-offshore bins (porpoises, 2006: X^2 = 62.19, $p < 0.0001$; porpoises, 2012: X^2 = 40.46, $p < 0.0001$; free-floating nets, X^2 = 106.95, $p < 0.0001$; hook-type gears, X^2 = 40.48, $p < 0.0001$; df = 3 for all analyses; **Figure 2**). Spearman rank tests show no correlation in porpoise or gear relative abundance across bins [porpoises vs. free-floating nets (2006), r_s = 0.80, p = 0.333; porpoises vs. free-floating nets (2012), r_s = −0.20, p = 0.917; porpoises vs. hook-type gears (2006), r_s = 0.74, p = 0.262; porpoises vs. hook-type gears (2012), r_s = −0.32, p = 0.684]. Gear use is biased toward near-bank habitats, being predominantly used ≤ 100 m from shore (hook-type gears, 53.9%; free-floating nets, 50.2%), whereas porpoises were observed most frequently 101–300 m from shore (2006, 36.4%; 2012, 43.6%). Across-river porpoise distribution also differs significantly between survey

TABLE 1 | Beta-binomial model selection showing Δ6 set for Yangtze-wide analysis, reporting Akaike Information Criterion corrected for small sample sizes (AICc), AICc scores (ΔAICc) and Akaike's weight (ωi).

Model structure	AICc	ΔAICc	ωi
pd ∼ cargo + fishvess	107.1	0.00	0.192
pd ∼ cargo + fishvess + livep	107.8	0.68	0.137
pd ∼ cargo	108.4	1.29	0.101
pd ∼ livep	108.9	1.84	0.067
pd ∼ cargo + livep	109.2	2.17	0.065
pd ∼ cargo + fishvess + hook	109.8	2.74	0.049
pd ∼ cargo + fishvess + livep + fixed	110.2	3.11	0.041
pd ∼ cargo + fishvess + livep + hook	110.6	3.54	0.033
pd ∼ fishvess + livep	110.6	3.56	0.032
pd ∼ cargo + fishvess + net	111.1	4.05	0.025
pd ∼ intercept only model	111.3	4.24	0.023
pd ∼ elec + livep	111.3	4.27	0.023
pd ∼ cargo + fishvess +elec	111.4	4.31	0.022
pd ∼ cargo + fishvess + fixed	111.4	4.34	0.022
pd ∼ cargo + hook	111.6	4.52	0.020
pd ∼ cargo + fishvess + elec + livep	111.7	4.64	0.019
pd ∼ cargo + net	111.9	4.85	0.017
pd ∼ cargo + fixed	111.9	4.87	0.017
pd ∼ cargo + elec	112.0	4.91	0.016
pd ∼ livep + hook	112.5	5.40	0.013
pd ∼ livep + fixed	112.5	5.46	0.013
pd ∼ livep + net	112.5	5.47	0.012
pd ∼ net	112.7	5.66	0.011
pd ∼ cargo + elec + livep	112.8	5.72	0.011
pd ∼ cargo + fishvess + livep + net	113.1	5.98	0.010

Cargo, cargo vessels; elec, proportion of fishers who think electrofishing is local problem; fishvess, fishing vessels; fixed, proportion of fishers using fixed nets; hook, proportion of fishers using hook-based gears; livep, live porpoises observed in 2006; net, proportion of fishers using free-floating nets; pd, proportion of respondents/section reporting dead porpoise observation in previous 12 months.

TABLE 2 | Beta-binomial and logit Gaussian model-averaged parameter estimates for Yangtze-wide analysis.

Parameter	Model-averaged estimate	95% CI lower	95% CI upper
Beta-binomial			
Intercept	−0.477	−1.379	0.425
Live porpoise sightings	0.013	0.001	0.024
Cargo vessels	0.982	0.182	1.780
Fishing vessels	−0.618	−1.193	−0.043
Logit Gaussian			
Intercept	−0.942	−2.237	0.353
Live porpoise sightings	0.014	−0.001	0.030
Cargo vessels	1.036	0.250	1.821
Fishing vessels	−0.693	−1.396	0.010

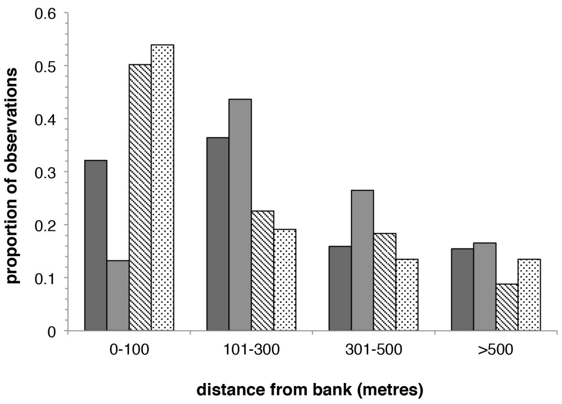


FIGURE 2 | Proportions of porpoise observations and reported use of fishing gears in four distance-from-bank bins across Yangtze mainstem. Dark gray, 2006 porpoise data (*n* = 439); mid-gray, 2012 porpoise data (*n* = 181); hashed bar, free-floating nets (*n* = 283); dotted bar, hook-type gears (*n* = 89).

TABLE 3 | Annual PBR values calculated for 2006 and 2012 porpoise population estimates.

R_{max} value	2006 estimate		2012 estimate	
	Mainstem-only (N_{min} = 1138.47)	Total population (N_{min} = 1672.85)	Mainstem-only (N_{min} = 462.71)	Total population (N_{min} = 952.91)
0.04	2.28	3.35	0.93	1.91
0.0352	2.00	2.94	0.81	1.68

years ($X^2 = 24.48$, $p < 0.0001$), with 32.1% observed ≤ 100 m from shore in 2006 but only 13.3% in 2012.

There was a significant decrease between survey years in the proportion of calves: in 2006, mean proportion of calves was 0.20 (95% confidence intervals, 0.17–0.24), whereas in 2012, mean proportion was 0.067 (0.045–0.10) ($X^2 = 31.7$, df = 1, $p < 0.001$).

Using the PBR model, maximum annual removal from the total population to allow an optimum sustainable population is always ≤ 3.35 individuals (**Table 3**). The logistic model estimates removal values that are nearly ten times higher, and predicts a maximum annual removal of 30 individuals using 2006 population size to allow an optimum sustainable population (**Table 4**). To generate the observed 2006–2012 decline, the logistic model always requires annual removal of ≥ 140 individuals. Thirty mainstem deaths were reported by respondents as having been observed during the 12 months before the 2008 interview survey (fishery-based trauma, *n* = 6; vessel collisions, *n* = 7; unknown cause, *n* = 17; Turvey et al., 2013). Using this baseline minimum removal rate, the 2006 logistic model predicts a decline, but not severe enough to produce the observed 2012 mainstem-only estimate of 505 individuals; observed deaths account for 21.0–21.5% of total predicted annual mortality (**Supplementary Table 4**).

DISCUSSION

Our combined analyses of different potential threats to the Critically Endangered Yangtze finless porpoise demonstrate the importance of an integrative approach for investigating complex

TABLE 4 | Estimated annual removals in logistic model from mainstem-only and total porpoise populations required to maintain stable 2006 population size (maximum sustainable removal) or result in observed 2012 population size (predicted actual removal).

R_{max}	Estimated annual removal (2006–2012)	Mainstem-only (2006 = 1,225, 2012 = 505)	Total population (2006 = 1,800, 2012 = 1,040)
0.04	Max sustainable removal	25	30
	Predicted actual removal	142–143	157–158
0.0352	Max sustainable removal	22	27
	Predicted actual removal	140–141	153–154

conservation problems, rather than only considering single anthropogenic threats and their impacts within multi-threat landscapes. By assessing the information content of existing conservation data, we demonstrate that critical consideration of even limited available datasets can provide important new insights to help understand population decline and guide future work. These findings support the possibility of evidence-based conservation management even for data-poor species.

Although our along-river analyses were conducted at a relatively coarse spatial scale, this spatial framework has been sufficient to provide useful conservation-relevant insights in previous studies (Turvey et al., 2010, 2013), and best-model sets contained better models than the null model. This approach identified a correlation between high cargo vessel densities and increased dead porpoise observations, supporting the suggestion that direct and/or indirect impacts of ship traffic (e.g., collisions, disturbance, anthropogenic noise) might be responsible for reducing porpoise viability and causing mortality (Li et al., 2008; Wang, 2009; Turvey et al., 2013). These findings highlight the urgent need for improved monitoring of vessel traffic and its environmental impact in the Yangtze, and further research into the dynamics and sustainability of interactions between porpoises and cargo vessels, which remain difficult to detect or infer in the absence of specific targeted studies.

Conversely, whereas fisheries bycatch is a key cause of mortality in small cetaceans (Reeves et al., 2013) and several Yangtze finless porpoise bycatch events are documented (Zhou and Wang, 1994; Wang et al., 2000, 2015; Turvey et al., 2013), available data show limited correlation between porpoise deaths and fishing activity even when analyzed across multiple spatial scales. Indeed, the only statistically significant correlation is an unexpected negative relationship between fishing vessel density and reported porpoise mortality in along-river analysis, although we note that this pattern is weak or questionable because 95% confidence intervals include 0 in some model-averaged estimates; any such relationship may potentially reflect a correlation with other landscape-level factors (e.g., greater localized amounts of fish resources for both subsistence fishers and porpoises in more anthropologically undisturbed river sections). If fisheries interactions are genuinely not involved in driving porpoise decline, our across-river analyses provide a potential reason why: porpoises and fishers tend not to use the same parts of the river, with porpoises observed most frequently 101–300 m from the bank, whereas free-floating nets and hook-type gears are used most frequently $\leq$ 100 m from the bank. Indeed, whereas hook-type gears account for 45.2% of observed porpoise bycatch events (Turvey et al., 2013), this gear category shows less overlap than free-floating nets with porpoise distribution.

However, we note that respondents reported similar levels of porpoises killed by fishery-based trauma and vessel collisions, suggesting that fisheries bycatch may also be an important driver of porpoise decline in this system. If this is indeed the case, various factors might explain the limited relationship we observed in our analyses between porpoise mortality and fishing activity. Although protocols have been developed to reduce reporting inaccuracy by respondents and detect ecological signals in interview datasets (e.g., Newing, 2011; Turvey et al., 2015b), these data can be affected by multiple biases (e.g., declining recall accuracy, misremembering) that potentially complicate our attempt to reconstruct porpoise mortality patterns. In particular, levels of bycatch reporting are often affected by respondent reticence in discussing sensitive or illegal behaviors, with fishers reluctant to report porpoise deaths (Lien et al., 1994; Slooten and Dawson, 2016). Furthermore, not all deaths will be observed, and carcasses might also drift downstream (Peltier et al., 2012; Moore et al., 2020); although we note that the coarse spatial scales used in along-river analyses should reduce mismatches between locations of porpoise deaths and subsequent carcass detections, and the fact that live porpoise counts are a significant positive predictor of dead porpoise counts in 100 km-section analysis suggests a local origin for observed dead individuals. However, our spatial data resolution may not detect finer-scale bycatch mortality patterns and correlates (especially for electrofishing, a potential threat for which we could only use indirect indices).

Porpoise distribution is influenced at local scales by hydrology, water quality, substrate and fish abundance (Wei et al., 2003; Zhang et al., 2015, 2018), so regional variation in habitat use by porpoises and fishers might be associated with higher overlap and bycatch risk. We therefore recommend additional field-based research into porpoise-fisheries interactions (including spatial mapping of areas of high fishing activity, and establishment of fisher observer programs) to identify possible bycatch-risk hotspots. Data on Yangtze finless porpoise prey species are still limited (Chen et al., 1980; Yang et al., 2021); future research into the identity and diversity of porpoise prey, and their local distribution, population trends and importance in regional fisheries, is also a priority. Furthermore, 2006 and 2012 survey data show marked differences in across-river porpoise distribution, with fewer porpoises observed near the bank in 2012. If this represents a genuine change in habitat use (versus undercounting of porpoises near the bank, and furthest from mid-channel survey vessels, in 2012), porpoise-gear overlap has decreased. Ecological reasons for this shift are unclear, but it could indicate increasing anthropogenic degradation of near-shore habitats, or elevated mortality from near-shore fishing.

Expected proportions of calves in a healthy Yangtze finless porpoise population are unknown; no pre-decline baseline exists, and data for other porpoise taxa vary between ∼2-27% of the population (Kasuya and Kureha, 1979; Siebert et al., 2006;

Thomsen et al., 2007). Although our analysis of between-year reproductive variation is relatively simple, it indicates that significantly fewer calves were recorded in 2012 compared to 2006. We note that reproduction is an annually variable trait in many cetacean populations (e.g., Manlik et al., 2016), and several uncontrolled factors can influence cetacean count data (Dawson et al., 2008); however, both surveys were conducted using the same methods and across the same area (Zhao et al., 2008; Mei et al., 2014). We thus interpret these results as suggesting reproductive success has decreased between the two survey years. A comparable pattern was observed (although not tested statistically) during baiji decline, with proportion of observed immature individuals decreasing from 31 to 17% in 1985–1999 (Zhang et al., 2003). Multiple mechanisms could reduce porpoise reproductive success and/or survivorship, hindering identification of cause(s). High concentrations of pollutants that affect cetacean reproduction and calf survival (heavy metals, persistent organic pollutants) have been found in Yangtze finless porpoise carcasses (Dong et al., 2006; Yang et al., 2008; Xiong et al., 2019), Yangtze fish (Xian et al., 2008), and the Yangtze River (Müller et al., 2008), with highest mercury concentrations and polychlorinated biphenyl hazard quotients found in porpoise calves (Dong et al., 2006; Yang et al., 2008). Although few data exist on specific Yangtze fish decreases (including porpoise prey species), fish stocks have declined substantially through habitat loss/degradation and damming (Ye et al., 2013), and decreased prey availability is associated with reduced reproductive success in several cetaceans (Mann, 2000; Ford et al., 2009). Anthropogenic noise from vessels and industrial activities is a source of environmental stress for Yangtze cetaceans (Zhao et al., 2008; Wang, 2009), and is implicated in reduced porpoise reproductive success through physiological damage, hormonal stress, behavioral alteration, and ecological effects on prey species (Nabi et al., 2018). Reproductive fitness may also be affected by reduced genetic diversity seen in the surviving porpoise population (Chen et al., 2017).

Predicted sustainable offtake rates differ between our population models by an order of ∼10 due to differences in model structure and aims: whereas logistic modeling only considers demographic and ecological parameters (e.g., point estimates of N) and does not explicitly consider uncertainty, PBR uses N_{min} and incorporates a precautionary recovery factor to identify whether sustainable removal thresholds have been exceeded in post-depletion populations (Robards et al., 2009). Conservation target-setting using the PBR approach can be affected by some sources of data uncertainty (e.g., bias in abundance estimates, catastrophic events, trends in natural mortality; Punt et al., 2020), but there is no evidence that these constitute significant concerns in this system. However, as PBR is a conservative management technique, higher removal rates might still be sustainable in the longer-term than predicted in our model. Furthermore, mainstem-only models might be invalidated if this section does not contain a closed population (Taylor, 1997), but instead experiences movement of individuals between the appended lakes, which is suggested although not demonstrated (Li et al., 2010; Huang et al., 2017). We have also assumed that modeled mortality levels represent human-caused deaths, because observed porpoise mortality has increased as the population declined over recent decades (Turvey et al., 2013); the number of deaths would instead have decreased with decreasing porpoise population size if these events were mostly due to natural causes, but we acknowledge that some observed deaths may be natural. Model outputs therefore represent guidelines only under certain population scenarios.

However, all models show that observed mortality levels are sufficient to drive decline, indicating that current direct mortality is likely to be unsustainable even with stable reproduction and carrying capacity. A precautionary approach suggests the lowest predicted sustainable removal threshold, the PBR estimates based upon 2012 porpoise population data, should constitute a management guideline (Milner-Gulland and Akçakaya, 2001). These predicted PBR sustainable removal thresholds are comparable to values estimated for other highly threatened cetaceans (e.g., Maui's dolphin *Cephalorhynchus hectori maui*; Slooten and Dawson, 2008), and suggest that reported mortality from both fishing and vessel strikes exceed sustainable levels; even if either threat was mitigated effectively, direct mortality would remain too high to maintain a stable population. Using these PBR thresholds to guide management would make maximum allowable annual mortality less than one porpoise in the mainstem, and less than two porpoises for the entire Yangtze.

Conversely, reported deaths account for only ∼21% of predicted annual mortality required to drive observed mainstem decline in the logistic model. This might suggest that reduction in survivorship and/or carrying capacity have also been important in driving decline, as demonstrated in population modeling comparing relative effects of anthropogenic removal vs. reducing prey availability in other cetaceans (Williams et al., 2016). We note that available mortality records were derived from a small respondent sample; for example, only 27 fishers were interviewed between Huangshi-Dongzhi in 2008, whereas this section contained 1,982 licensed fishing families in 2013 (Turvey et al., 2015a). Additional interviews might therefore yield considerably more records. However, as discussed above, respondent data contain multiple sources of uncertainty, making it challenging to infer the level of underestimation in our data. Although actual mortality levels could be substantially greater than estimated, we caution against extrapolating mortality data from small respondent samples across much larger respondent populations.

CONCLUSION

Our combined analyses of available datasets highlight what we can currently infer about threats associated with Yangtze finless porpoise decline, and also highlight what we still do not know. We provide evidence of a spatial association between mortality and cargo traffic; observed mortality levels (probably underestimates of true levels) are unsustainable; and recruitment is decreasing, although multiple factors could be driving this trend. Conversely, although we demonstrate that even a small annual mortality rate from bycatch is enough to drive porpoise decline, we cannot clarify a relationship between porpoise mortality and fishing activities using existing data, and we emphasize the urgent need for further applied research into the significance and spatial dynamics of this potential threat. These findings also suggest that existing precautionary management efforts that focus upon mitigating potential fisheries impacts within the Yangtze system may be insufficient to prevent further porpoise declines, and highlight the importance of adopting an evidence-based approach to investigate the information content of available datasets.

Uncertainty persists over impacts of different anthropogenic activities on porpoises across the Yangtze, and we recognize that porpoise decline might represent a cumulative effect of multiple stressors, a widely recognized problem in many aquatic systems facing complex threats; we encourage further research to investigate combined effects and potential interactions (additive, synergistic or antagonistic) between stressors in the Yangtze system (Crain et al., 2008; Côté et al., 2016). In particular, we recognize that the inevitable biases inherent to interview datasets are likely to represent a key limitation to our current understanding of patterns, levels and drivers of porpoise mortality in the Yangtze, especially with regard to fishers reporting sensitive information to outsiders on porpoise-fisheries interactions. To reduce such data limitations, it is crucial to conduct more systematic investigation of porpoise mortality to obtain direct data on deaths caused by different anthropogenic factors, ideally through a strandings investigation program that includes post-mortem examinations of all dead individuals using internationally established protocols (Wang et al., 2015; IJsseldijk et al., 2019).

However, our new baseline can help guide future research priorities, highlighting the need for targeted monitoring, research and modeling of potential threats across priority habitats. The current precautionary approach to reducing fishing pressure may provide beneficial impacts on wider biodiversity, but *in situ* management should also mitigate additional threats. More widely, critical assessment of available data for other threatened cetaceans can hopefully be used to understand sustainability or otherwise of different human activities in other poorly understood freshwater and marine systems, as a tool for reducing bycatch associated with commercial fishing operations (e.g., to meet the newly established Import Provisions of the United States Marine Mammal Protection Act; NOAA Fisheries, 2021). We encourage greater collaboration and sharing of existing datasets for threatened species, to maximize understanding of conservation-relevant data content and prevent time and resources from being wasted in biodiversity conservation (Mace et al., 2000; Haddaway, 2015).

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/**Supplementary Material**, further inquiries can be directed to the corresponding author/s.

ETHICS STATEMENT

The studies involving human participants were reviewed and approved by the Zoological Society of London Ethics Committee. Written informed consent for participation was not required for this study in accordance with national legislation and institutional requirements.

AUTHOR CONTRIBUTIONS

LM and ST conceived ideas, designed methodology, and led writing of manuscript. ST, ZM, YH, and DW collected the data. LM, XH, and ST analyzed the data. All authors contributed critically to drafts and gave final approval for publication.

FUNDING

Funding was provided by a Ph.D. studentship from the Natural Environment Research Council doctoral training partnership.

ACKNOWLEDGMENTS

We thank Helen Chatterjee for logistical support.

SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2021.791484/full#supplementary-material

REFERENCES

Bower, S. D., Brownscombe, J. W., Birnie-Gauvin, K., Ford, M. I., Moraga, A. D., Pusiak, R. J. P., et al. (2018). Making tough choices: picking the appropriate conservation decision-making tool. *Conserv. Lett.* 11:e12418. doi: 10.1111/conl.12418

Carr, M. H., and Raimondi, P. T. (1999). Marine protected areas as a precautionary approach to management. *CalCOFI Rep.* 40, 71–76.

Catullo, G., Masi, M., Falcucci, A., Maiorano, L., Rondinini, C., and Boitani, L. (2008). A gap analysis of southeast Asian mammals based on habitat suitability models. *Biol. Conserv.* 141, 2730–2744. doi: 10.1016/j.biocon.2008.08.019

Chen, M., Fontaine, M. C., Chehida, Y. B., Zheng, J., Labbé, F., Mei, Z., et al. (2017). Genetic footprint of population fragmentation and contemporary collapse in a freshwater cetacean. *Sci. Rep.* 7:14449. doi: 10.1038/s41598-017-14812-z

Chen, P., Liu, P., Liu, R., Lin, K., and Pilleri, G. (1980). Distribution, ecology, behaviour and conservation of the dolphins of the middle reaches of Changjiang (Yangtze) River (Wuhan-Yueyang). *Investig. Cetacea* 10, 87–103.

Cooney, R. (2004). *The Precautionary Principle in Biodiversity Conservation and Natural Resource Management: An Issues Paper for Policy-Makers, Researchers and Practitioners*. Gland: IUCN.

Côté, I. M., Darling, E. S., and Brown, C. J. (2016). Interactions among ecosystem stressors and their importance in conservation. *Proc. R. Soc. B* 283:20152592. doi: 10.1098/rspb.2015.2592

Crain, C. M., Kroeker, K., and Halpern, B. S. (2008). Interactive and cumulative effects of multiple human stressors in marine systems. *Ecol. Lett.* 11, 1304–1315.

Dawson, S., Wade, P., Slooten, E., and Barlow, J. (2008). Design and field methods for sighting surveys of cetaceans in coastal and riverine habitats. *Mamm. Rev.* 38, 19–49. doi: 10.1111/j.1365-2907.2008.00119.x

Dong, W., Xu, Y., Wang, D., and Hao, Y. (2006). Mercury concentrations in Yangtze finless porpoises (*Neophocaena phocaenoides asiaeorientalis*) from eastern Dongting Lake, China. *Fresenius Environ. Bull.* 15, 441–447.

Ford, J. K. B., Ellis, G. M., Olesiuk, P. F., and Balcomb, K. C. (2009). Linking killer whale survival and prey abundance: food limitation in the oceans' apex predator? *Biol. Lett.* 6, 139–142. doi: 10.1098/rsbl.2009.0468

Grantham, H. S., Moilanen, A., Wilson, K. A., Pressey, R. L., Rebelo, T. G., and Possingham, H. P. (2008). Diminishing return on investment for biodiversity data in conservation planning. *Conserv. Lett.* 1, 190–198. doi: 10.1038/s41598-019-52241-2

Haddaway, N. R. (2015). A call for better reporting of conservation research data for use in meta-analyses. *Conserv. Biol.* 29, 1242–1245. doi: 10.1111/cobi.12449

Harrison, X. A. (2015). A comparison of observation-level random effect and beta-binomial models for modelling overdispersion in binomial data in ecology & evolution. *PeerJ* 3:e1114. doi: 10.7717/peerj.1114

Huang, S. L., Mei, Z., Hao, Y., Zheng, J., Wang, K., and Wang, D. (2017). Saving the Yangtze finless porpoise: time is rapidly running out. *Biol. Conserv.* 210, 40–46. doi: 10.1016/j.biocon.2016.05.021

IJsseldijk, L. L., Brownlow, A. C., and Mazzariol, S. (2019). *Best Practice on Cetacean Post Mortem Investigation and Tissue Sampling. ACCOBAMS-MOP7/2019/Doc 33*. Rotterdam: ACCOBAMS/ASCOBANS.

Ingram, D. J., Coad, L., Collen, B., Kümpel, N. F., Breuer, T., Fa, J. E., et al. (2015). Indicators for wild animal offtake: methods and case study for African mammals and birds. *Ecol. Soc.* 20:40.

IUCN (2021). *The IUCN Red List of Threatened Species. Version 2021-1*. Available online at: http://www.iucnredlist.org (accessed August 9, 2021).

Jaramillo-Legorreta, A., Rojas-Bracho, L., Brownell, R. L. Jr., Read, A. J., Reeves, R. R., Ralls, K., et al. (2007). Saving the vaquita: immediate action, not more data. *Conserv. Biol.* 21, 1653–1655. doi: 10.1111/j.1523-1739.2007.00825.x

Kasuya, T., and Kureha, K. (1979). The population of finless porpoise in the Inland Sea of Japan. *Sci. Rep. Whales Res. Inst.* 31, 1–44. doi: 10.5181/cetology.0.23_1

Kimura, S., Akamatsu, T., Li, S., Dong, L., Wang, K., Wang, D., et al. (2012). Seasonal changes in the local distribution of Yangtze finless porpoises related to fish presence. *Mar. Mamm. Sci.* 28, 308–324.

Li, S., Akamatsu, T., Wang, D., Wang, K., Dong, S., Zhao, X., et al. (2008). Indirect evidence of boat avoidance behavior of Yangtze finless porpoises. *Bioacoustics* 17, 174–176. doi: 10.1080/09524622.2008.9753806

Li, S., Dong, S., Kimura, S., Akamatsu, T., Wang, K., and Wang, D. (2010). "Detection of Yangtze finless porpoises in the Poyang Lake mouth area via passive acoustic data-loggers," in *Biology, Evolution and Conservation of River Dolphins Within South America and Asia*, eds J. M. Shostell and M. Ruiz-Garcia (New York, NY: Nova Science Publishers), 341–356.

Lien, J., Stenson, G. B., Carver, S., and Chardine, J. (1994). "How many did you catch? The effects of methodology on bycatch reports obtained from fishermen," in *Gillnets and Cetaceans*, eds W. F. Perrin, G. P. Donovan, and J. Barlow (Cambridge: International Whaling Commission), 535–540.

Lindenmayer, D. B., Piggott, M. P., and Wintle, B. A. (2013). Counting the books while the library burns: why conservation monitoring programs need a plan for action. *Front. Ecol. Environ.* 11:549–555. doi: 10.1890/120220

Mace, G. M., Balmford, A., Boitani, L., Cowlishaw, G., Dobson, A. P., Faith, D. P., et al. (2000). It's time to work together and stop duplicating conservation efforts. *Nature* 405:393. doi: 10.1038/35013247

MacMillan, D. C., and Marshall, K. (2006). The Delphi process—an expert-based approach to ecological modelling in data-poor environments. *Anim. Conserv.* 9, 11–19. doi: 10.1111/j.1469-1795.2005.00001.x

Manlik, O., McDonald, J. A., Mann, J., Raudino, H. C., Bejder, L., Krützen, M., et al. (2016). The relative importance of reproduction and survival for the conservation of two dolphin populations. *Ecol. Evol.* 6, 3496–3512. doi: 10.1002/ece3.2130

Mann, J. (2000). Female reproductive success in bottlenose dolphins (*Tursiops* sp.): life history, habitat, provisioning, and group-size effects. *Behav. Ecol.* 11, 210–219. doi: 10.1093/beheco/11.2.210

McDonald-Madden, E., Baxter, P. W. J., and Possingham, H. P. (2008). Making robust decisions for conservation with restricted money and knowledge. *J. Appl. Ecol.* 45, 1630–1638. doi: 10.1111/j.1365-2664.2008.01553.x

Mei, Z., Zhang, X., Huang, S.-L., Zhao, X., Hao, Y., Zhang, L., et al. (2014). The Yangtze finless porpoise: on an accelerating path to extinction? *Biol. Conserv.* 172, 117–123. doi: 10.1016/j.biocon.2014.02.033

Milner-Gulland, E. J., and Akçakaya, H. R. (2001). Sustainability indices for exploited populations. *Trends Ecol. Evol.* 16, 686–692. doi: 10.1016/s0169-5347(01)02278-9

Monnahan, C. C., Branch, T. A., and Punt, A. E. (2015). Do ship strikes threaten the recovery of endangered eastern North Pacific blue whales? *Mar. Mamm. Sci.* 31, 279–297. doi: 10.1111/mms.12157

Moore, M. J., Mitchell, G. H., Rowles, T. K., and Early, G. (2020). Dead cetacean? Beach, bloat, float, sink. *Front. Mar. Sci.* 7:333. doi: 10.3389/fmars.2020.00333

Müller, B., Berg, M., Yao, Z., Zhang, X., Wang, D., and Pfluger, A. (2008). How polluted is the Yangtze river? Water quality downstream from the Three Gorges Dam. *Sci. Total Environ.* 402, 232–247. doi: 10.1016/j.scitotenv.2008.04.049

Nabi, G., McLaughlin, R. W., Hao, Y., Wang, K., Zeng, X., Khan, S., et al. (2018). The possible effects of anthropogenic acoustic pollution on marine mammals' reproduction: an emerging threat to animal extinction. *Environ. Sci. Pollut. Res.* 25, 19338–19345. doi: 10.1007/s11356-018-2208-7

Newing, H. (2011). *Conducting Research in Conservation: A Social Science Perspective*. Abingdon: Routledge. doi: 10.4324/9780203846452

NOAA Fisheries (2021). *NOAA Fisheries Establishes International Marine Mammal Bycatch Criteria for U.S. Imports*. Available online at: https://www.fisheries.noaa.gov/foreign/marine-mammal-protection/noaa-fisheries-establishes-international-marine-mammal-bycatch-criteria-us-imports (accessed December 1, 2021).

Pan, M., and Huntington, H. P. (2015). A precautionary approach to fisheries in the central Arctic ocean: policy, science, and China. *Mar. Policy* 63, 153–157. doi: 10.1016/j.marpol.2015.10.015

Peltier, H., Dabin, W., Daniel, P., Van Canneyt, O., Dorémus, G., Huon, M., et al. (2012). The significance of stranding data as indicators of cetacean populations at sea: modelling the drift of cetacean carcasses. *Ecol. Indic.* 18, 278–290. doi: 10.1371/journal.pone.0062180

Punt, A. E., Siple, M., Francis, T. B., Hammond, P. S., Heinemann, D., Long, K. J., et al. (2020). Robustness of potential biological removal to monitoring, environmental, and management uncertainties. *ICES J. Mar. Sci.* 77, 2491–2507. doi: 10.1093/icesjms/fsaa096

R Development Core Team (2017). *R: A Language and Environment for Statistical Computing*. Vienna: R Foundation for Statistical Computing.

Reeves, R. R., McClellan, K., and Werner, T. B. (2013). Marine mammal bycatch in gillnet and other entangling net fisheries, 1990 to 2011. *Endanger. Species Res.* 20, 71–97. doi: 10.3354/esr00481

Richards, S. A. (2008). Dealing with overdispersed count data in applied ecology. *J. Appl. Ecol.* 45, 218–227. doi: 10.1111/j.1365-2664.2007.01377.x

Robards, M. D., Burns, J. J., Meek, C. L., and Watson, A. (2009). Limitations of an optimum sustainable population or potential biological removal approach for conserving marine mammals: Pacific walrus case study. *J. Environ. Manage.* 91, 57–66. doi: 10.1016/j.jenvman.2009.08.016

Rodrigues, A. S. L. (2011). Improving coarse species distribution data for conservation planning in biodiversity-rich, data-poor, regions: no easy shortcuts. *Anim. Conserv.* 14, 108–110. doi: 10.1111/j.1469-1795.2011.00451.x

Rojas-Bracho, L., Brusca, R. C., Álvarez-Borrego, S., Brownell, R. L. Jr., Camacho-Ibar, V., Ceballos, G., et al. (2019). Unsubstantiated claims can lead to tragic conservation outcomes. *Bioscience* 69, 12–14. doi: 10.1093/biosci/biy138

Sampaio, F. D. F., Freire, C. A., Sampaio, T. V. M., Vitule, J. R. S., and Fávaro, L. F. (2015). The precautionary principle and its approach to risk analysis and quarantine related to the trade of marine ornamental fishes in Brazil. *Mar. Policy* 51, 163–168. doi: 10.1016/j.marpol.2014.08.003

Siebert, U., Gilles, A., Lucke, K., Ludwig, M., Benke, H., Kock, K. H., et al. (2006). A decade of harbour porpoise occurrence in German waters—analyses of aerial surveys, incidental sightings and strandings. *J. Sea Res.* 56, 65–80. doi: 10.1016/j.seares.2006.01.003

Slooten, E., and Dawson, S. M. (2008). Sustainable levels of human impact for Hector's dolphin. *Open Conserv. Biol. J.* 2, 37–43. doi: 10.2174/1874839200802010037

Slooten, E., and Dawson, S. M. (2016). *Hector's and Maui Dolphin Bycatch 1985-2015. International Whaling Commission Report SC/66b/SM/15*. Cambridge: International Whaling Commission.

Soetaert, K., Petzoldt, T., and Setzer, W. R. (2010). Solving differential equations in R: package deSolve. *J. Stat. Softw.* 33, 1–25. doi: 10.18637/jss.v033.i09

Sutherland, W. J., Pullin, A. S., Dolman, P. M., and Knight, T. M. (2004). The need for evidence based conservation. *Trends Ecol. Evol.* 19, 305–308. doi: 10.1016/j.tree.2004.03.018

Taylor, B. L. (1997). Defining 'population' to meet management objectives for marine mammals. *Mol. Genet. Mar. Mamm.* 3, 49–65. doi: 10.1111/gcb.14296

Taylor, B. L., Chivers, S. J., Larese, J., and Perrin, W. F. (2007). *Generation Length and Percent Mature Estimates for IUCN Assessments of Cetaceans. National*

Marine Fisheries Administrative Report LJ-07-01. La Jolla, CA: Southwest Fisheries Science Center, 1–24.

Thieme, M., Lehner, B., Abell, R., Hamilton, S. K., Kellndorfer, J., Powell, G., et al. (2007). Freshwater conservation planning in data-poor areas: an example from a remote Amazonian basin (Madre de Dios River, Peru and Bolivia). *Biol. Conserv.* 135, 500–517.

Thompson, P. M., Wilson, B., Grellier, K., and Hammond, P. S. (2000). Combining power analysis and population viability analysis to compare traditional and precautionary approaches to conservation of coastal cetaceans. *Conserv. Biol.* 14, 1253–1263. doi: 10.1046/j.1523-1739.2000.00099-410.x

Thomsen, F., Laczny, M., and Piper, W. (2007). The harbour porpoise (*Phocoena phocoena*) in the central German Bight: phenology, abundance and distribution in 2002-2004. *Helgol. Mar. Res.* 61, 283–289. doi: 10.1007/s10152-007-0075-6

Turvey, S. T., Barrett, L. A., Hart, T., Collen, B., Hao, Y., Zhang, L., et al. (2010). Spatial and temporal extinction dynamics in a freshwater cetacean. *Proc. R. Soc. B* 277, 3139–3147. doi: 10.1098/rspb.2010.0584

Turvey, S. T., Hao, Y., and Wang, D. (2015a). *Identifying Conservation Solutions for the Yangtze Finless Porpoise (Neophocaena asiaeorientalis asiaeorientalis) Through Community Research*. International Whaling Commission. Available online at: http://www.iucn-csg.org/wp-content/uploads/2010/03/Turvey_IWC_final_report.pdf (accessed December 1, 2021).

Turvey, S. T., Trung, C. T., Quyet, V. D., Nhu, H. V., Thoai, D. V., Tuan, V. C. A., et al. (2015b). Interview-based sighting histories can inform regional conservation prioritization for highly threatened cryptic species. *J. Appl. Ecol.* 52, 422–433. doi: 10.1111/1365-2664.12382

Turvey, S. T., Pitman, R. L., Taylor, B. L., Barlow, J., Akamatsu, T., Barrett, L. A., et al. (2007). First human-caused extinction of a cetacean species? *Biol. Lett.* 3, 537–540. doi: 10.1098/rsbl.2007.0292

Turvey, S. T., Risley, C. L., Moore, J. E., Barrett, L. A., Hao, Y., Zhao, X., et al. (2013). Can local ecological knowledge be used to assess status and extinction drivers in a threatened freshwater cetacean? *Biol. Conserv.* 157, 352–360. doi: 10.1016/j.biocon.2012.07.016

VanderWerf, E. A., Groombridge, J. J., Fretz, J. S., and Swinnerton, K. J. (2006). Decision analysis to guide recovery of the po'ouli, a critically endangered Hawaiian honeycreeper. *Biol. Conserv.* 129, 383–392. doi: 10.1016/j.biocon.2005.11.005

Wade, P. R. (1998). Calculating limits to the allowable human-caused mortality of cetaceans and pinnipeds. *Mar. Mamm. Sci.* 14, 1–37. doi: 10.1111/j.1748-7692.1998.tb00688.x

Wade, P. R., and Angliss, R. (1997). *Guidelines for Assessing Marine Mammal Stocks: Report of the GAMMS Workshop. NOAA Technical Memorandum NMFS-OPR-12*. Seattle, WA: U.S. Department of Commerce, 1–93.

Wang, D. (2009). Population status, threats and conservation of the Yangtze finless porpoise. *Bull. Chin. Acad. Sci.* 27, 3473–3484.

Wang, D., Liu, R., Zhang, X., Yang, J., Wei, Z., Zhao, Q., et al. (2000). "Status and conservation of the Yangtze finless porpoise," in *Biology and Conservation of Freshwater Cetaceans in Asia*, eds R. R. Reeves, B. D. Smith, and T. Kasuya (Gland: IUCN), 81–85.

Wang, D., Zhang, X., and Liu, R. (1998). "Conservation status and its future of baiji and Yangtze finless porpoise in China," in *Ecology and Environmental Protection of Large Irrigation Projects in Yangtze River in 21st Century*, eds Z. Hua, B. Fu, and Y. Yang (Beijing: Environmental Science Press), 218–226. doi: 10.1065/espr2006.10.350

Wang, Y., Li, W., and van Waerebeek, K. (2015). Strandings, bycatches and injuries of aquatic mammals in China, 2000-2006, as reviewed from official documents: a compelling argument for a nationwide strandings programme. *Mar. Policy* 51, 242–250.

Wang, Z., Akamatsu, T., Wang, K., and Wang, D. (2014). The diel rhythms of biosonar behavior in the Yangtze finless porpoise (*Neophocaena asiaeorientalis asiaeorientalis*) in the port of the Yangtze River: the correlation between prey availability and boat traffic. *PLoS One* 9:e97907. doi: 10.1371/journal.pone.0097907

Wei, Z., Wang, D., Zhang, X., Zhao, Q., Kuang, X., Wang, X., et al. (2003). Habitat use and preliminary evaluation of the habitat status of the Yangtze finless porpoise (*Neophocaena phocaenoides asiaeorientalis*) in the Balijiang section of the Yangtze River, China. *Acta Zool. Sin.* 49, 163–170.

Williams, R., Thomas, L., Ashe, E., Clark, C. W., and Hammond, P. S. (2016). Gauging allowable harm limits to cumulative, sub-lethal effects of human activities on wildlife: a case-study approach using two whale populations. *Mar. Policy* 70, 58–64. doi: 10.1016/j.marpol.2016.04.023

Xian, Q., Ramu, K., Isobe, T., Sudaryanto, A., Liu, X., Gao, Z., et al. (2008). Levels and body distribution of polybrominated diphenyl ethers (PBDEs) and hexabromocyclododecanes (HBCDs) in freshwater fishes from the Yangtze River, China. *Chemosphere* 71, 268–276. doi: 10.1016/j.chemosphere.2007.09.032

Xiong, X., Qian, Z., Mei, Z., Wu, J., Hao, Y., Wang, K., et al. (2019). Trace elements accumulation in the Yangtze finless porpoise (*Neophocaena asiaeorientalis asiaeorientalis*)—a threat to the endangered freshwater cetacean. *Sci. Total Environ.* 686, 797–804. doi: 10.1016/j.scitotenv.2019.06.031

Yang, F., Zhang, Q., Xu, Y., Jiang, G., Wang, Y., and Wang, D. (2008). Preliminary hazard assessment of polychlorinated biphenyls, polybrominated diphenyl ethers, and polychlorinated dibenzo-p-dioxins and dibenzofurans to Yangtze finless porpoise in Dongting Lake, China. *Environ. Toxicol. Chem.* 27, 991–996. doi: 10.1897/07-381.1

Yang, J., Wang, K., Mei, Z., Xu, J., Zheng, J., Wan, X., et al. (2021). Temporal variation in the diet of Yangtze finless porpoise calls for conservation of semi-migratory fish. *Freshw. Biol.* 66, 992–1001. doi: 10.1111/fwb.13692

Ye, S., Li, Z., Zhang, T., Liu, J., and Xie, S. (2013). Assessing fish distribution and threats to fish biodiversity in the Yangtze River Basin, China. *Ichthyol. Res.* 61, 183–188. doi: 10.3897/BDJ.9.e76669

Zhang, X., Liu, R., Zhao, Q., Zhang, G., Wei, Z., Wang, X., et al. (1993). The population of finless porpoise in the middle and lower reaches of Yangtze River. *Acta Theriol. Sin.* 13, 260–270.

Zhang, X., and Vincent, A. C. J. (2017). Integrating multiple datasets with species distribution models to inform conservation of the poorly-recorded Chinese seahorses. *Biol. Conserv.* 211, 161–171. doi: 10.1016/j.biocon.2017.05.020

Zhang, X., Wang, D., Liu, R., Wei, Z., Hua, Y., Wang, Y., et al. (2003). The Yangtze River dolphin or baiji (*Lipotes vexillifer*): population status and conservation issues in the Yangtze River, China. *Aquat. Conserv.* 13, 51–64. doi: 10.1002/aqc.547

Zhang, X., Wang, H., Wan, A., Chen, M., Wang, K., and Yu, D. (2018). Effects of macrozoobenthos and zooplankton on the occurrence of Yangtze finless porpoise via the bottom-up force at a confluence of rivers. *Nat. Environ. Pollut. Technol.* 17, 175–182.

Zhang, X., and Wang, K. (1999). Population viability analysis for the Yangtze finless porpoise. *Acta Ecol. Sin.* 19, 529–533. doi: 10.1007/s11626-011-9448-x

Zhang, X., Yu, D., Wang, H., Wan, A., Chen, M., Tao, F., et al. (2015). Effects of fish community on occurrences of Yangtze finless porpoise in confluence of the Yangtze and Wanhe rivers. *Environ. Sci. Pollut. Res. Int.* 22, 9524–9533. doi: 10.1007/s11356-015-4102-x

Zhao, X., Barlow, J., Taylor, B. L., Pitman, R. L., Wang, K., Wei, Z., et al. (2008). Abundance and conservation status of the Yangtze finless porpoise in the Yangtze River, China. *Biol. Conserv.* 141, 3006–3018. doi: 10.1065/espr2006.10.350

Zhou, K., and Wang, X. (1994). Brief review of passive fishing gear and incidental catches of small cetaceans in Chinese waters. *Rep. Int. Whal. Comm. Spec. Issue* 15, 347–354.

Zhou, X., Guang, X., Sun, D., Xu, S., Li, M., Seim, I., et al. (2018). Population genomics of finless porpoises reveal an incipient cetacean species adapted to freshwater. *Nat. Commun.* 9:1276. doi: 10.1038/s41467-018-03722-x

Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

frontiers in Marine Science

ORIGINAL RESEARCH
published: 13 January 2022
doi: 10.3389/fmars.2021.795942

Estimating Bycatch From Non-representative Samples (II): A Case Study of Pair Trawlers and Common Dolphins in the Bay of Biscay

Etienne Rouby[1,2,3], *Laurent Dubroca*[4], *Thomas Cloâtre*[5], *Sebastien Demanèche*[5], *Mathieu Genu*[1,3], *Kelly Macleod*[6†], *Hélène Peltier*[3,7], *Vincent Ridoux*[1,2,3] and *Matthieu Authier*[1,3,7*]

[1] Université de La Rochelle, La Rochelle, France, [2] UMR7372 Centre d'Etudes Biologiques de Chizé (CEBC), Villiers En Bois, France, [3] Observatoire PELAGIS, UMS3462, CNRS-La Rochelle Université, La Rochelle, France, [4] Ifremer, Laboratoire Ressources Halieutiques de Port en Bessin, Port-en-Bessin-Huppain, France, [5] Ifremer, Laboratoire de Biologie Halieutique de Brest, Brest, France, [6] Joint Nature Conservation Committee, Aberdeen, United Kingdom, [7] ADERA, Pessac, France

Edited by:
Jeff Moore,
Southwest Fisheries Science Center
(NOAA), United States

Reviewed by:
Stephen Stohs,
Southwest Fisheries Science Center
(NOAA), United States
Tomo Eguchi,
National Marine Fisheries Service
(NOAA), United States
Doug Sigourney,
Integrated Statistics, United States

***Correspondence:**
Matthieu Authier
matthieu.authier@univ-lr.fr

†Present address:
Kelly Macleod,
HiDef Aerial Surveying Limited,
Lillyhall, United Kingdom

Specialty section:
This article was submitted to
Marine Megafauna,
a section of the journal
Frontiers in Marine Science

Received: 15 October 2021
Accepted: 02 December 2021
Published: 13 January 2022

Citation:
Rouby E, Dubroca L, Cloâtre T,
Demanèche S, Genu M, Macleod K,
Peltier H, Ridoux V and Authier M
(2022) Estimating Bycatch From
Non-representative Samples (II):
A Case Study of Pair Trawlers and
Common Dolphins in the Bay of
Biscay. Front. Mar. Sci. 8:795942.
doi: 10.3389/fmars.2021.795942

Marine megafauna plays an important functional role in marine ecosystems as top predators but are threatened by a wide range of anthropogenic activities. Bycatch, the incidental capture of non-targeted species in commercial and recreational fisheries, is of particular concern for small cetacean species, such as dolphins and porpoises. In the North-East Atlantic, common dolphin (*Delphinus delphis*, Linné 1758) bycatch has been increasing and associated with large numbers of animals stranding during winter on the French Atlantic seashore since at least 2017. However, uncertainties around the true magnitude of common dolphin bycatch and the fisheries involved have led to delays in the implementation of mitigation measures. Current data collection on dolphin bycatch in France is with non-dedicated observers deployed on vessels for the purpose of national fisheries sampling programmes. These data cannot be assumed representative of the whole fisheries' bycatch events. This feature makes it difficult to use classic ratio estimators since they require a truly randomised sample of the fishery by dedicated observers. We applied a newly developed approach, regularised multilevel regression with post-stratification, to estimate total bycatch from unrepresentative samples and total fishing effort. The latter is needed for post-stratification and the former is analysed in a Bayesian framework with multilevel regression to regularise and better predict bycatch risk. We estimated the number of bycaught dolphins for each week and 10 International Council for the Exploration of the Sea (ICES) divisions from 2004 to 2020 by estimating jointly bycatch risk, haul duration, and the number of hauls per days at sea (DaS). Bycatch risk in pair trawlers flying the French flag was the highest in winter 2017 and 2019 and was associated with the longest haul durations. ICES divisions 8.a and 8.b (shelf part of the Bay of Biscay) were estimated to have the highest common dolphin bycatch. Our results were consistent with independent estimates of common dolphin bycatch from strandings. Our method show cases how non-representative observer data can nevertheless be analysed to estimate fishing duration, bycatch risk and, ultimately, the number of bycaught dolphins. These weekly-estimates improve upon current knowledge of the nature of common dolphin bycatch and can be used to inform management and policy decisions at a finer spatio-temporal scale than has been possible to date. Our results suggest that limiting haul duration, especially in winter, could serve as an effective mitigation strategy.

Keywords: additional mortality, anthropogenic activities, modelling, non-representative samples, conservation, small cetaceans, fisheries, post-stratification

1. INTRODUCTION

Over the last 50 years, the conservation status of cetaceans has been deteriorating (Brownell et al., 2019). Over 80 species of cetaceans occur worldwide and bycatch, the non-intentional capture or killing of non-target species in commercial or recreational fisheries (Hall, 1996; Davies et al., 2009), remains a threat, especially to small-sized species (Scarff, 1977; Read et al., 2006; Avila et al., 2018; Anderson et al., 2020). Success stories in small cetacean conservation are the exception rather than the rule (e.g., Bessesen, 2018). Both Rogan et al. (2021) and Bearzi and Reeves (2021) opined of institutional failures to conserve cetaceans in European Waters in spite of current legislation (for example, the Habitats Directive, the Marine Strategy Framework Directive) or regional agreements such as the Agreement on the Conservation of Small Cetaceans of the Baltic and North Seas (ASCOBANS, see **Table 1** for acronyms; ICES, 2020c). Over 20 species of small cetaceans have been registered in the North-East Atlantic, with roughly half of which occurring regularly (Course, 2021). Because of their slow life histories and their limited potential rates of increase, small cetaceans are particularly at risk of decline when anthropogenic activities induce additional mortality on populations (Read, 2008). Anthropogenic activities and their cumulative impacts can take a heavy toll on populations. Common species may disappear, such as short-beaked common dolphins (*Delphinus delphis*, hereafter called common dolphins) in the Adriatic Sea (Bearzi and Reeves, 2021), or are under many threats, e.g., in the Bay of Biscay (García-Baron et al., 2019; Murphy et al., 2021).

In 2013, the common dolphin's conservation status in the European Marine Atlantic, as assessed under Article 17 of the Habitats Directive, was "Unfavourable–Inadequate" because of fishery bycatch (Murphy et al., 2021). Common dolphin bycatch in the Bay of Biscay, in particular, has attracted a lot of media coverage since 2017 in international outlets[1] and motivated (with bycatch of Harbour porpoise *Phocoena phocoena* in the Baltic Sea) a special request of Non-Governmental Organisations to the European Commission in 2019. The International Council for the Exploration of the Sea (ICES) advised in 2020, for the common dolphin in the Bay of Biscay, a combination of temporal closures of all métiers (*i.e* the combination of gear, target species, and fishing area) of concern and application of pingers on pair trawlers to mitigate bycatch outside of the period of closure (ICES, 2020b). Temporal closures, restricted to winter months in which strandings of common dolphins with evidence of bycatch have increased in recent years (ICES, 2020d), could have been implemented as emergency measures under the provisions of the Common Fisheries Policy. For 2021, France instead required the mandatory use of acoustic repulsive devices (pingers) on all pair trawlers flying the French Flag (code métier Pair trawlers and hereafter referred to as PTM) operating in the Bay of Biscay[2], a technical mitigation measure whose efficiency was found wanting (Ulrich and Doerner, 2021). This decision against the advice of ICES was motivated by a lack of knowledge on common dolphins, including its abundance at the level of the whole North-East Atlantic (the currently recognised management unit: Murphy et al., 2013) and the extent of bycatch. The issue of managing uncomfortable knowledge through interpretation of scientific uncertainty can be raised (Schweder, 2000; Rayner, 2012); yet it should not eclipse that there are genuine difficulties in estimating accurately the true magnitude and the extent of bycatch of small cetaceans (Moore et al., 2021).

Several types of fishing gear are known to cause cetacean bycatch: drift nets, set gill, trammel nets, both pair and single midwater trawls, and some demersal trawls (Rogan and Mackey,

TABLE 1 | List of acronyms.

Acronym	Meaning
ASCOBANS	Agreement on the Conservation of Small Cetaceans of the Baltic, North East Atlantic, Irish and North Seas
DCF	Data collection framework
DPMA	"Direction des pêches maritimes et de l'aquaculture"
GNS	Gillnetters
GTR	Gill trammel netters
ICES	International Council for the Exploration of the Sea
Ifremer	Institut Français de Recherche pour l'Exploitation de la Mer
ObsMer	Observation des captures en Mer (French national observer scheme for monitoring fisheries)
PBR	Potential Biological Removal
PTM	Pair trawlers
PTB	Bottom pair trawlers
VAST	Vector-Autoregressive Spatio-Temporal
WGBYC	ICES Working Group on Bycatch of Protected Species
WKEMBYC	ICES Workshop on fisheries Emergency Measures to minimise BYCatch of short-beaked common dolphins in the Bay of Biscay and harbor porpoise in the Baltic Sea

---

[1] https://www.nytimes.com/2019/05/02/world/europe/france-dolphins-fishing.html

[2] https://www.mer.gouv.fr/protecting-cetaceans-annick-giradin-presents-7-commitments-made-french-state-fishermen-and

2007; Fernández-Contreras et al., 2010; Peltier et al., 2016). Accurate quantification of bycatch rates by fishing gears or métiers remains a challenging endeavour (Babcock et al., 2003; ICES, 2019). Traditionally, bycatch data are collected by onboard observers monitoring fishing operations and recording the unwanted catch of non-commercial species (Course, 2021). Ratio estimators, based on the number of observed hauls with bycatch over the total number of monitored hauls, are used (Alverson et al., 1994; page 18) but are plagued by large uncertainties due to low coverage and the usual small number of hauls with small cetacean bycatch (Babcock et al., 2003; Authier et al., 2021; Course, 2021). It may also happen that some bycatch events may not be reported by non-dedicated observers since they may drive observations for other purposes than report bycatch (e.g., commercial discards or stock assessments). A critical assumption behind the use of such ratio-estimators is that of a representative sample: this assumption is difficult to sustain unless monitoring is dedicated to marine mammals, and allocation of observers to fishing vessels is truly randomised (that is, not at the discretion of skippers). Even if we are willing to assume representative sampling, if coverage is low, the main challenge remains to extrapolate from sample to the whole fisheries. In France, monitoring of cetacean bycatch in fisheries is non-dedicated (Cornou et al., 2018), and the collected data are described as non-representative of the bycatch events, preventing the use of ratio-estimators (Anonymous, 2016; page 24).

This non-dedicated nature and the sparseness of the bycatch data complicates the use of state-of-the-art spatio-temporal models such as Vector-Autoregressive Spatio-Temporal (VAST) (Thorson, 2019). This framework accommodates density-dependence, spatial and temporal scales to estimate biomass or abundance or presence of a species (Thorson et al., 2015). Spatio-temporal models are also used to model the co-occurrence of commercial and bycaught species, allowing the estimate of bycatch risk with time-varying spatial effects (Ward et al., 2015). These types of model-based approaches methodologies allow modelling spatial and temporal auto-correlation through the use of Gaussian process priors. It is difficult to transfer a priori the same model-based structure to analyse small cetacean bycatch. Models such as VAST capitalise on the availability of catch data that are collected as part of fisheries monitoring. In contrast, bycatch monitoring is not as developed or efficiently enforced in many fisheries in Europe (ICES, 2019, 2020a; Sala et al., 2019), and bycatch data are typical of low quality and unrepresentative (Authier et al., 2021). In Europe, fisheries monitoring is carried out under the Data Collection Framework (DCF) but "remains not well-suited for the dedicated monitoring of rare and protected bycatch in high-risk fisheries since its main focus is the statistically-sound random sampling of all commercial fisheries" (Ulrich and Doerner, 2021). Because of these data quality issues, Authier et al. (2021) conducted a simulation study to gauge the potential of investigating recent methods for the analysis of non-representative samples (for a recent example of a model-based approach to estimate bycatch, refer to Luck et al., 2020) in the context of small cetacean bycatch: they concluded the potential of regularised multilevel regression with post-stratification to infer more accurately bycatch rates (although uncertainties remained large). The approach of Authier et al. (2021) also makes use of Gaussian process priors but does not necessarily assume that a large dataset has been collected.

We analysed historical bycatch monitoring data collected by onboard observers (from 2004 to 2020) on PTM, a métier historically associated with high levels of dolphin bycatch in the Bay of Biscay (ICES, 2019; Murphy et al., 2021). Leveraging recent modelling developments (see companion article; Authier et al., 2021), we jointly estimated bycatch risk, haul duration, and number of hauls per days at sea (DaS) from an updated and revised observer dataset on common dolphin bycatch. The modelling procedure accounts for the sparseness of the bycatch incident dataset and the low observer coverage through constraints. This type of constraint (which can be viewed as some sort of penalisation) is also called regularisation. We used structured priors, such as Gaussian processes, to achieve regularisation and leverage the within-year information at the weekly scale (inducing correlation between some weeks). Structured priors allow inducing some spatial- or temporal-dependency between so called random-effects whereas unstructured priors do not induce such dependency (but both assume exchangeability). Importantly, we used this model-based approach to disaggregate bycatch risk at the level of calendar weeks in order to document within-year variations. Estimates were summed over a whole year to investigate between-year variations in the number of bycaught dolphins. We compared these model-based estimates with strandings, both within- and between-years. Finally, we concluded with recommendations on conservation and mitigation.

2. MATERIALS AND METHODS

2.1. Materials

2.1.1. Study Division

The study area (**Figure 1**) encompasses 10 ICES divisions within area 27: it includes the Bay of Biscay, the English Channel, and part of the Celtic seas. These zones are associated with submesoscale and mesoscale oceanographic processes, such as eddies and upwelling, that enhance ecosystem productivity and result in high availability of fishes, including commercial species (e.g., European seabass *Dicentrarchus labrax*, Sardine *Sardina pilchardus* or Anchovy *Engraulis encrasicolus*). Each division can roughly be classified as oceanic or neritic: divisions 7.d, 7.e, 7.f, 7.g, 7.h, 8.a, 8.b, and 8.c are related to neritic ecosystems while divisions 7.j, and 8.d are related to oceanic ecosystems.

2.1.2. Data Sources

Two main sources of data were used. The first dataset, called ObsMer[3] ("Observation des captures en Mer"), is collected as part of an onboard observer program set up within the Data Collection Framework of the Common Fisheries Policy. The ObsMer program is carried out by Ifremer ("Institut Français de Recherche pour l'Exploitation de la Mer"), under the supervision of the Directorate of Fisheries and Aquaculture ("Direction

---

[3] https://sih.Ifremer.fr/Ressources/ObsMer

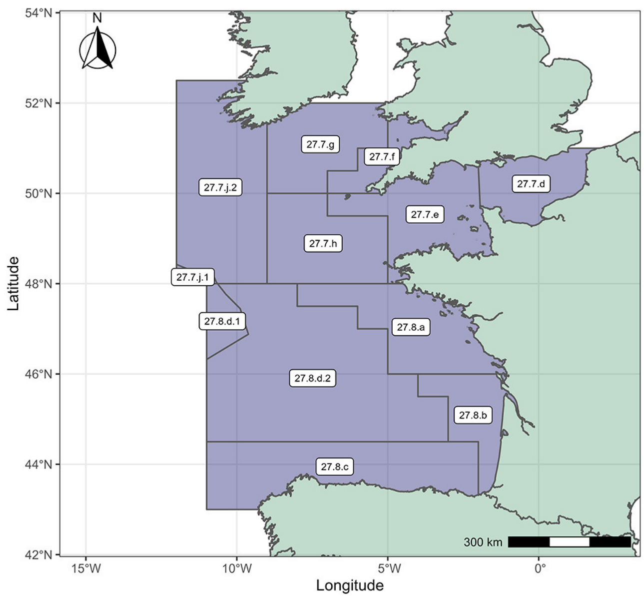


FIGURE 1 | Study area in the North-East Atlantic ocean, with ICES divisions overlayed.

des pêches maritimes et de l'aquaculture," DPMA). ObsMer observers' primary duty is to register the length and weight composition of catches. Still, they have to report any bycatch event if they witness such events. ObsMer data on PTM cover 4, 484 hauls between 2004 and 2021, of which 82 were associated with a bycatch event of at least 1 and up to 50 common dolphins. ObsMer provides, among other information, the geographic position, timing, and duration of hauls. Although ObsMer is aiming at a coverage of 10 and 5% of fishing effort for (level-3 métier) PTM for vessels of more than 15 m and less than 15 m, respectively, these figures are rarely, if ever, reached in practice: accepting onboard observers remains entirely at the discretion of skippers. The effort is quite low overall, ranging from 0 to 11% of Days at Sea (DaS) (**Table 2**). A DaS is any continuous period of 24 h (or part thereof) during which a vessel is present within an area and absent from the port (Anonymous, 2019). The number of observed hauls with at least one bycatch record is very small because the yearly percentage of observed hauls with a bycatch event never exceeded 4.5% and was 0 in nearly half of the surveyed years. ObsMer data on pair-trawlers are an unrepresentative sample of hauls, largely because allowing an observer remains largely at the discretion of skippers (Babcock et al., 2003; Benoît and Allard, 2009).

The second dataset provides monthly estimates of total fishing effort in each division. This dataset is generated from the algorithm SACROIS developed by Ifremer and integrates data from Vessel Monitoring System, log-books, and landing statistics (for boats longer than 18 m from January 1, 2004, and longer than 15 m from January 1, 2005; Système d'Information Halieutique, 2017). SACROIS aims at (1) correcting errors that could exists in the integrated dataset due to recording or collecting errors and (2) reconstitute métiers during the fishing trip as they are not recorded in logbooks or fish market data (Cornou et al., 2018). The SACROIS dataset provides the best available estimates of total effort, in DaS, between 2004 and 2020 (**Table 2**). There are also refusals from skippers due to administrative and security reasons. Skippers must file an application for authorisation to embark observers and even if they decide to file, the authorisation may be declined due to security reasons (e.g., not enough room or rails not high enough).

These two datasets are complementary for our purposes: ObsMer provides micro-level data on marine mammal bycatch at the resolution of hauls. From these data, bycatch risk may be estimated (Luck et al., 2020). Fishing trips effort data, on the other hand, are macro-level: they provide spatialised effort data at the scale of a whole fishing fleet. These population-level data on effort allows the post-stratification of bycatch risk estimated from observer data to obtain the number of bycaught dolphins (Authier et al., 2021). Descriptive statistics of both datasets are displayed in **Table 2**. Used in tandem, both datasets allow using regularised multilevel regression with post-stratification to estimate cetacean bycatch from non-representative samples (Authier et al., 2021).

Estimates were finally compared to strandings along the French Atlantic seaboard. The French Stranding Network, founded in the 1970s, is dedicated to the monitoring of marine mammal strandings along the shores of France (mainland and overseas). Around 400 trained volunteers are currently taking an active part in the network. These volunteers make the complete coverage of French coastlines possible. Standardised training of volunteers by permanent Observatoire Pelagis staff, which takes place two times a year, ensure the homogeneity, comparability, and standardisation of data collection procedures in the field. Observatoire Pelagis is mandated by the French Ministry of Ecology to train and deliver authorisation to handle carcasses of marine mammals (which are all protected species under national law). It also collates the data and analyse it to inform on the status of marine mammal populations. Stranding data for the period 2004–2020 were used. Only common dolphins found with lesions diagnostic of bycatch in fishing gear were considered (Kuiken, 1994) as well as those stranded during multiple stranding events, or "unusual mortality events" related to lesions diagnostic of bycatch. Multiple stranding events were defined as high numbers of strandings occurring in a restricted area with a common cause of death. The threshold was defined at 30 cetaceans over 10 consecutive days recorded along a maximal distance of 200 km in the Bay of Biscay, and 10 individuals per 10 days per 200 km of coastline along the coast of the western Channel (Peltier et al., 2014). Reverse drift modelling uses a deterministic drift model developed by Meteo France (Peltier et al., 2012) to reconstruct the trajectory of every stranded common dolphin from its stranding location to its likely area of death at sea. The number of dead stranded animals in each cell is then corrected by the cell-specific probability of being stranded (Peltier et al., 2016). These probabilities were estimated by numerical experiment in which the drift of carcasses in the study area was simulated in order to assess with which frequency they would reach a coastline (Peltier and Ridoux, 2015).

TABLE 2 | Descriptive statistics for Observation des captures en Mer (ObsMer) and SACROIS data displayed for each year.

Dataset	ObsMer						SACROIS
Year	**Hauls**	**Average Duration (hours)**	**Bycatch events**	**Median nb of dolphins**	**Max. nb of dolphins**	**DaS (Coverage %)**	**Total Effort (DaS)**
2004	4	2.80	0	-	-	4 (0.0)	8 530
2005	5	4.26	0	-	-	4 (0.0)	8 790
2006	122	4.62	0	-	-	90 (1.1)	7 853
2007	727	3.89	6	1.5	5	401 (6.4)	6 305
2008	554	4.81	6	1.5	4	328 (10.9)	3 011
2009	464	5.50	20	2	50	326 (7.4)	4 413
2010	305	3.52	1	4	4	159 (3.5)	4 486
2011	173	3.99	2	3	3	86 (2.1)	4 001
2012	210	3.58	4	4	8	96 (2.4)	4 005
2013	128	3.81	2	5.5	9	75 (1.8)	4 192
2014	114	4.44	0	-	-	78 (1.9)	4 136
2015	136	2.77	1	2	2	78 (1.7)	4 597
2016	156	4.75	5	3	10	106 (2.3)	4 603
2017	196	5.23	12	2	20	124 (2.6)	4 835
2018	184	3.85	1	1	1	102 (2.8)	3 613
2019	438	5.45	11	2	8	289 (7.4)	3 139
2020	123	3.69	2	2	3	70 (4.0)	1 686

2.2. Methods

2.2.1. Modelling Bycatch Risk and Duration of Hauls

Observation des captures en Mer data allow both bycatch risk and haul duration to be modelled. The two may be correlated as a longer towing time may result in an increased likelihood of bycatch, all else being equal. Bycatch risk is defined at the level of a haul. Hauls can differ in duration as skippers may target different commercial species at different times of the year. However, the population-level data on effort is aggregated and available as DaS, the metric currently used in international fora (e.g., ICES Working Group on BYCatch, WGBYC). The number of hauls per DaS was also modelled from the ObsMer dataset in order to scale up bycatch risk per haul by the number of hauls per DaS. We modelled jointly bycatch risk, fishing duration of hauls, and the number of hauls per DaS of pair-trawlers flying the French flag at the week-level for each year between 2004 and 2020 (**Table 2**) and each ICES division (**Figure 1**). The goal of the approach is to model bycatch rates at the weekly scale for each year within each ICES division using a simple autoregressive model. To smooth the fluctuations of estimated bycatch rates in weekly estimates we constrained estimation using Gaussian Process structured priors. These priors allow (i) to estimate an average bycatch risk profile at the weekly scale and from this weekly average, (ii) to estimate year- and division-level deviations.

2.2.2. Notations

Let $\mathcal{N}(d, s)$ denote a normal distribution of location parameter d and scale parameter s. Let $\mathcal{G}(a, b)$ denote a gamma distribution of scale parameter a and rate parameter b. Let $\mathcal{LN}(d, s)$ denote a log-normal distribution of location parameter d and scale parameter s. The gamma and the log-normal distribution are used and compared to model the likelihood of the haul duration since they assume a positive continuous distribution. These distribution laws are appropriate modelling choices for positively skewed data with a constant coefficient of variation. Let $\mathcal{GP}(m, c)$ denote a Gaussian process of mean function m and covariance function c. A Gaussian Process is a prior distribution on a function f in which, for any vector $\boldsymbol{x} = (x_1, \ldots, x_n)$, $f(\boldsymbol{x})$ is drawn from a $n-$dimensional normal distribution with mean $m(\boldsymbol{x})$ and covariance matrix depending only on the distances of the point x from each other (Gelman et al., 2021, page 465). In the following, we will drop the $\boldsymbol{x}$ and write in a shorthand manner $\boldsymbol{\theta} \sim \mathcal{GP}(\boldsymbol{m}, \boldsymbol{S})$ to mean that the vector $\boldsymbol{\theta}$ of n parameters has a Gaussian process prior and follows a multivariate normal distribution whose mean vector $\boldsymbol{m}$ is equal to $m(\boldsymbol{x})$ and whose covariance matrix $\boldsymbol{S}$ is defined for any pairs (x, x') as $S\left(x, x'\right) = c(x, x')$, where c is the covariance function of the Gaussian process prior.

2.2.3. Joint Modelling Approach

Let i denotes the i^{th} haul (fishing operation) happening in ICES statistical division j in week t of year k. Let y_{ijkt}, d_{ijkt}, and n_{jkt} denote, respectively, bycatch event (0 or 1), fishing duration (in hours, $d_{ijkt} > 0$), and the number of hauls per DaS_{jkt}. Bycatch risk p_{jkt} is estimated from

$$y_{ijkt} \sim \text{Bernoulli}\left(\mathrm{p}_{jkt} = \text{logit}^{-1}\left(\alpha^{1}_{jkt}\right)\right) \qquad (1)$$

To account for strict positivity, fishing duration is modelled either with a Gamma or a log-normal likelihood:

$$d_{ijkt} \sim \mathcal{G}\left(\beta, \frac{\beta}{\bar{d}_{jkt}}\right) \tag{2a}$$

$$d_{ijkt} \sim \mathcal{LN}\left(\bar{d}_{jkt}, \sigma\right) \tag{2b}$$

The number of hauls per DaS is modelled assuming a zero-truncated Poisson likelihood:

$$n_{jkt} \sim \mathcal{P}^{+}\left(\mathrm{DaS}_{jkt} \times \lambda_{jkt}\right) \tag{3}$$

Parameters $\bar{d}_{jkt} = e^{\alpha^{2}_{jkt}}$ and $\lambda_{jkt} = e^{\alpha^{3}_{jkt}}$ are rates. The linear predictors $\boldsymbol{\alpha}_{jk}$ are vectors of week-level parameters related to ICES division j and year k (dropping the superscript for convenience):

$$\begin{cases} \boldsymbol{\alpha}_{jk} \sim \mathcal{GP}\left(\boldsymbol{\delta}_k, \boldsymbol{\Sigma}_{\text{division}}\right) & \\ \boldsymbol{\delta}_k \sim \mathcal{GP}\left(\boldsymbol{\epsilon}, \boldsymbol{\Sigma}_{\text{year}}\right) & \\ \epsilon_t = \mu & t = 1 \\ \epsilon_{t+1} \sim \mathcal{N}(\varepsilon_t, \sigma_{\text{week}}) & t > 1 \end{cases} \tag{4}$$

Parameter μ is the intercept. The vector $\boldsymbol{\epsilon}$ aggregates the mean weekly effects (on the linear predictor scale) which are modelled with a first-order random walk to ensure some smoothness in between-week variations (Authier et al., 2021). The vector $\boldsymbol{\delta}_k$ are year-specific deviations from the mean weekly pattern $\boldsymbol{\epsilon}$. The vector $\boldsymbol{\alpha}_{jk}$ are division-specific deviations from the mean yearly pattern $\boldsymbol{\delta}_k$. Smoothness in $\boldsymbol{\alpha}_{jk}$ and $\boldsymbol{\delta}_k$ is controlled via the covariance matrices $\Sigma_{\text{division}} = \Delta_{\text{division}}\Omega\Delta_{\text{division}}$ and $\Sigma_{\text{year}} = \Delta_{\text{year}}\Omega\Delta_{\text{year}}$.Matrices $\Sigma_{.}$ have dimensions $\mathrm{n}_{\text{week}} \times \mathrm{n}_{\text{week}}$ (53×53). These covariance matrices are decomposed into a product of a diagonal matrix $\Delta_{.}$ (of dimension 53×53) with the common scale parameter on the diagonal, and a correlation matrix Ω (of dimension 53×53; Chen and Dunson, 2003):

$$\Delta_{.} = \begin{bmatrix} \sigma_{.} & 0 & \dots & 0 & 0 \\ 0 & \sigma_{.} & \dots & 0 & 0 \\ \vdots & \vdots & \ddots & \vdots & \vdots \\ 0 & 0 & \dots & \sigma_{.} & 0 \\ 0 & 0 & \dots & 0 & \sigma_{.} \end{bmatrix} \tag{5}$$

$\Omega = \Omega\left(t, t^{'}\right)$ is a matrix with the correlation between week t and week $t^{'}$ of dimensions $\mathrm{n}_{\text{week}} \times \mathrm{n}_{\text{week}}$ (53×53). A Matérn correlation function of order $\nu = \frac{3}{2}$ and range parameter fixed to $\rho = \frac{3}{2}$ was assumed: $\Omega\left(t, t^{'}\right) = \left(1 + \frac{2\sqrt{3} \times d(t-t^{'})}{3}\right) \times \exp -\frac{2\sqrt{3} \times d(t-t^{'})}{2}$ where $d(t - t^{'}) = |t - t^{'}|$ is the temporal distance (in weeks) between weeks t and $t^{'}$. The choice of the range parameter induces a temporal correlation of 0.05 after 4 weeks (that is, temporal independence after a month; Authier et al., 2021). The correlation matrix Ω is assumed known and is depicted in **Figure 2**. Equations 4 and 5 allow modelling an interaction between week, year, and division. The joint model defined in Equations (1), (2a), and (3) includes a time-varying component at the week-scale with interaction with year and division.

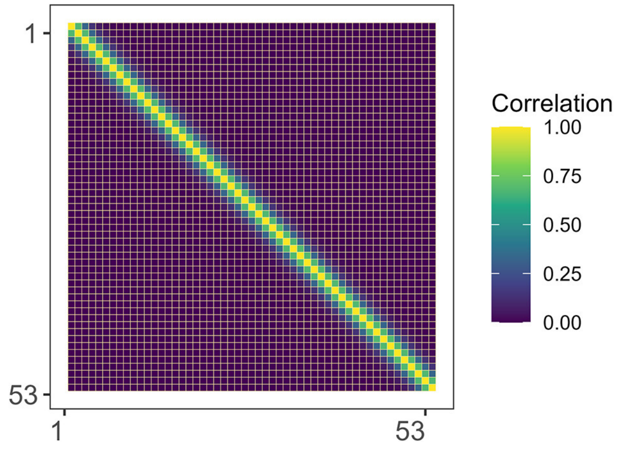


FIGURE 2 | Graphical representation of the assumed correlation matrix Ω.

Simpler models without such interactions, and with only additive effects, were also fitted to the data. The simplest model included only additive random (unstructured) effects (dropping the superscript for convenience):

$$\begin{cases} \alpha_{jkt} = \epsilon_t + \delta^{*}_{k} + \alpha^{*}_{j} & \\ \alpha^{*}_{j} \sim \mathcal{N}\left(0, \sigma_{\text{division}}\right) & \forall\, j \\ \delta^{*}_{k} \sim \mathcal{N}\left(0, \sigma_{\text{year}}\right) & \forall\, k \\ \epsilon_t = \mu & t = 1 \\ \epsilon_{t+1} \sim \mathcal{N}(\varepsilon_t, \sigma_{\text{week}}) & t > 1 \end{cases} \tag{6}$$

Models are multilevel, accommodating week-, year-, and division-level variations. They also use structured priors such as Gaussian processes or random walks to regularise estimation (Gao et al., 2019). More information on these models, and on applying (regularised) multilevel regression with post-stratification in the context of estimating bycatch, are detailed by Authier et al. (2021). Estimation was carried out in a Bayesian framework using programming language `Stan` (Carpenter et al., 2017) called from `R` v.4.0.1 (R Core Team, 2020) with library `Rstan` (Stan Development Team, 2020). `Stan` uses Hamiltonian dynamics in Markov chain Monte Carlo (MCMC) to sample values from the joint posterior distribution (Carpenter et al., 2017). Four chains were initialised from diffuse random starting points and run for a total of 2,000 iterations, discarding the first 1,000 as a warm-up. Default settings for the No-U-Turn Sampler (NUTS) were changed to 0.99 for `adapt delta` and 15 for `max treedepth` (Hoffman and Gelman, 2014). Priors are reported in **Table 3**. We fitted a total of 6 models of differing complexity (**Table 4**): we compared models assuming either gamma or a log-normal likelihood for haul duration, and models assuming additive effects vs. interactive effects of the week, year, and divisions. Model fitting was carried out

TABLE 3 | Prior specifications.

Parameter	Specification	Response variable	Meaning
μ	$\sim \mathcal{N}(0, \frac{1}{2})$	Bycatch risk	Intercept (on linear predictor scale).
prop	$\sim \mathcal{D}(1, 1, 1)$		Variance partitioning proportions
σ_{total}	$\sim \mathcal{GG}(\frac{1}{2}, \frac{1}{2}, \frac{\log 10}{2})$		Total variability (on linear predictor scale)
σ_{week}	$= \sigma_{\text{total}} \sqrt{\text{prop}_1}$		Week-level variability
σ_{year}	$= \sigma_{\text{total}} \sqrt{\text{prop}_2}$		Year-level variability
σ_{division}	$= \sigma_{\text{total}} \sqrt{\text{prop}_3}$		Division-level variability
μ	$\sim \mathcal{N}(0, 5)$	Fishing duration	Intercept (on linear predictor scale).
prop	$\sim \mathcal{D}(1, 1, 1)$		Variance partitioning proportions
σ_{total}	$\sim \mathcal{GG}(\frac{1}{2}, \frac{1}{2}, \frac{\log 2}{3})$		Total variability (on linear predictor scale)
σ_{week}	$= \sigma_{\text{total}} \sqrt{\text{prop}_1}$		Week-level variability
σ_{year}	$= \sigma_{\text{total}} \sqrt{\text{prop}_2}$		Year-level variability
σ_{division}	$= \sigma_{\text{total}} \sqrt{\text{prop}_3}$		Division-level variability
μ	$\sim \mathcal{N}(0, 5)$	Haul numbers	Intercept (on linear predictor scale).
prop	$\sim \mathcal{D}(1, 1, 1)$		Variance partitioning proportions
σ_{total}	$\sim \mathcal{GG}(\frac{1}{2}, \frac{1}{2}, \frac{\log 2}{2})$	per Days	Total variability (on linear predictor scale)
σ_{week}	$= \sigma_{\text{total}} \sqrt{\text{prop}_1}$		Week-level variability
σ_{year}	$= \sigma_{\text{total}} \sqrt{\text{prop}_2}$	at Sea	Year-level variability
σ_{division}	$= \sigma_{\text{total}} \sqrt{\text{prop}_3}$		Division-level variability
ρ	$\frac{3}{2}$	All	Range of Matérn correlation function
ν	$\frac{3}{2}$		Smoothness of Matérn correlation function

$\mathcal{D}()$ denotes the Dirichlet distribution for modelling proportions (such that $\sum_{l=1}^{3} \boldsymbol{prop}_l = 1$) and $\mathcal{GG}()$ the Gamma-Gamma distribution for scale parameters (Griffin and Brown, 2017; Pérez et al., 2017).

on the supercomputer facilities of the "Mésocentre de calcul de Poitou Charentes (Université de Poitiers/ISAE-ENSMA/La Rochelle Université)." Codes are available at https://gitlab.univ-lr.fr/mauthier/cdptmbycatch. For confidentiality reasons, the actual dataset cannot be shared: a synthetic dataset, generated by predicting from the posterior distribution, is provided instead.

2.2.4. Estimating the Total Number of Hauls and Bycatch Events

The number of unobserved hauls N_{jkt} that happened in ICES statistical division j in week t of year k can be estimated from the number of observed DaS in ObsMer ($\text{DaS}_{jkt}^{\text{ObsMer}}$) and from total effort $\text{DaS}_{jkt}^{\text{tot}}$ (and accounting for zero-truncation):

$$\hat{\text{N}}_{jkt} = \frac{\hat{\lambda_{jkt}}}{1 - e^{-\hat{\lambda_{jkt}}}} \times \left(\text{DaS}_{jkt}^{\text{tot}} - \text{DaS}_{jkt}^{\text{ObsMer}}\right) \quad (7)$$

The total number of bycatch events in ICES statistical division j in week t of year k is estimated as the sum of events observed in ObsMer ($\text{Bycatch}_{jkt}^{\text{ObsMer}}$) and the number of unobserved hauls multiplied by bycatch risk ($\hat{\text{p}_{jkt}}$):

$$\text{Byc}\hat{\text{a}}\text{tch}_{jkt} = \text{Bycatch}_{jkt}^{\text{ObsMer}} + \hat{\text{N}_{jkt}} \times \hat{\text{p}_{jkt}} \quad (8)$$

Similarly, for each year, the number of common dolphins bycaught in pair-trawlers can be estimated using the observed number of bycaught dolphins in ObsMer, the estimated number of unobserved hauls (Equation 7), bycatch risk, and either the median number of dolphins involved in a bycatch event (**Table 2**, or the grand median of $m = 2$ for years with no observed bycatch event). We used the median to attenuate the influence of some bycatch events involving up to 50 dolphins (**Table 2**). These estimates are thereafter referred to as model-based estimates.

2.3. Comparing Model-Based Estimates With Strandings

The sample provided by ObsMer, a non-dedicated observer scheme of marine mammal bycatch, may not be representative of all bycatch. In addition, it provides very sparse data, with less than 100 observed events over 17 years (**Table 4**) when strandings have reached several hundred per week in recent years (ICES, 2020d) (for all causes of death). Despite this, the weekly pattern of bycatch risk provided by ObsMer roughly matches that of strandings, with an increase in winter (**Figure 3**). Despite this rough match, the ObsMer data also suggest a heightened risk in summer, especially in the 2000s, whereas strandings suggest such an increased risk in very recent years (Peltier et al., 2021).

The number of stranded common dolphins with evidence of bycatch can be used to estimate the total bycatch mortality with reverse drift modelling (Peltier et al., 2016). These stranding-based estimates are now used in international working groups (ICES, 2020d). Reverse drift modelling corrects for at-sea drifting conditions, but cannot inform on which fishing gears were responsible for bycatch. Hence, strandings-based estimates are total estimates of bycatch and can be compared to model and observation based estimates of bycatch by French pair-trawlers. These model-based estimates use data independent from strandings, but they should not exceed stranding-based estimates. Second, whether model-estimates correlate with strandings-based ones is of interest to shed light on the increased mortality witnessed in the Bay of Biscay (Peltier et al., 2021). For each year, we checked the magnitude of model-based estimates against stranding-based ones and computed Pearson's correlation coefficient between the two time-series at the month level. To account for drift, these correlations were computed with and without a lag of 2 weeks when aggregating model-based estimates at the month level.

3. RESULTS

We built and compared six models (**Table 4**). Convergence was reached for all parameters with all $\hat{R} < 1.05$. Model $\mathcal{M}_6$ had the lowest WAIC and was selected as the best model for

TABLE 4 | Model selection.

Model	Likelihood for duration	Specification	$\widehat{WAIC}_{se}$	$\Delta_{\widehat{WAIC}}$	Computation time (h)
$\mathcal{M}_6$	Gamma	ICES division × week × year	$18,265_{169}$	0	50
$\mathcal{M}_5$	Log-normal	ICES division × week × year	$18,746_{185}$	481	47
$\mathcal{M}_4$	Gamma	ICES division + week × year	$19,065_{151}$	800	10
$\mathcal{M}_3$	Log-normal	ICES division + week × year	$19,475_{167}$	1,210	11
$\mathcal{M}_2$	Gamma	ICES division + week + year	$21,553_{133}$	3,288	4
$\mathcal{M}_1$	Log-normal	ICES division + week + year	$21,886_{148}$	3,621	3

Models are ordered in increasing order of WÂIC (the smaller, the better the fit). se stands for "standard error".

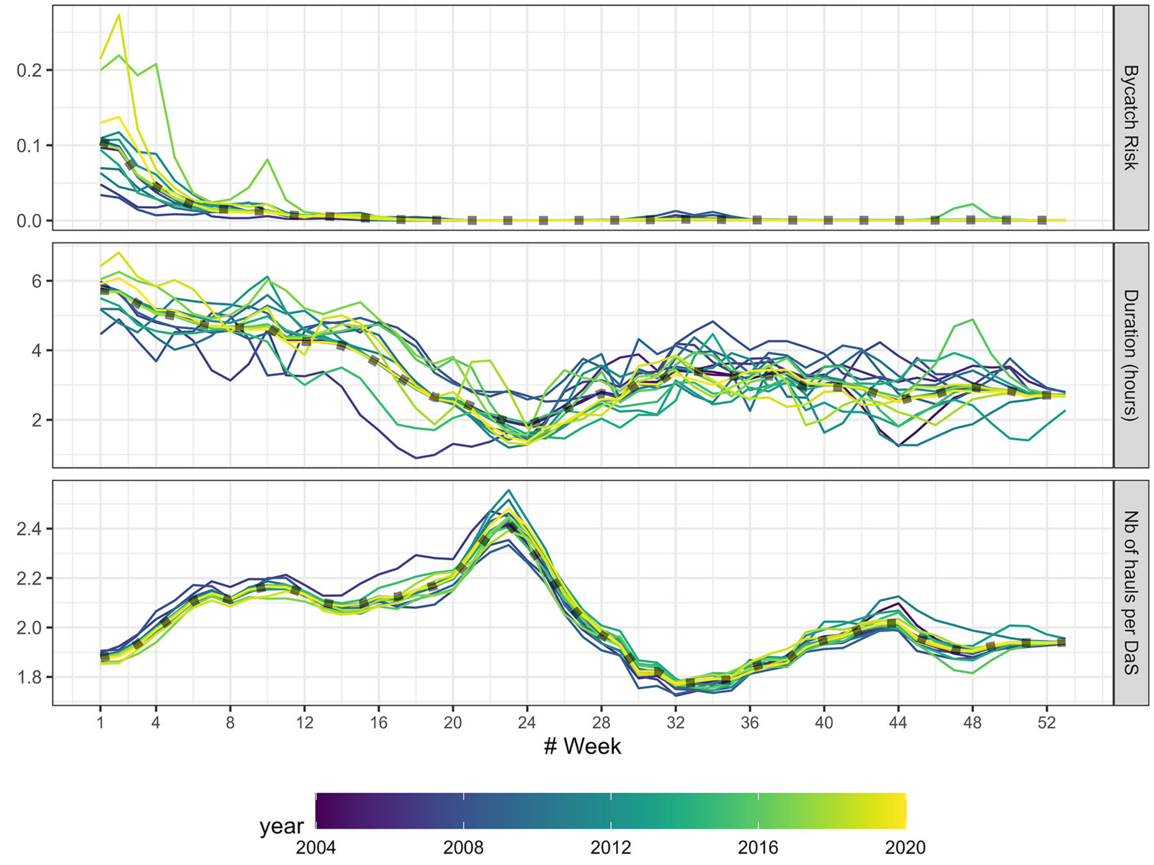


FIGURE 3 | Model-based estimates (posterior medians) of bycatch risk, haul duration, and number of hauls per Days at Sea (DaS) of pair-trawlers (PTM) flying the French flag operating in the study area. Each colour represents a different year (parameters α_{jkt} in Equation 4) and the dotted black line the yearly average (parameters ϵ_t in Equation 4).

further inferences. Model $\mathcal{M}_6$ included an interaction between week, year, and ICES division (Equations 4 and 5). All codes to fit models are available at https://gitlab.univ-lr.fr/mauthier/cdptmbycatch.

3.1. Bycatch Risk, Haul Duration, and Haul Number Per DaS

Haul duration, hauls per DaS, and bycatch risk per haul (Equations 7 and 8) were jointly estimated. Their temporal variations are displayed in **Figure 3** for each week between 2004 and 2020. Haul duration was the highest in week 1 with a posterior median estimate of 5.8 h that decreased to 4.0 h in week 16, before dropping to 2 h in week 24. Haul duration increased up to 3 h in week 32 and plateaued until the end of the year. Remarkable years were 2017, 2019, and 2020 with the longest haul durations estimated from week 1 to 10. From week 10 onwards, years before 2012 displayed some variations in haul duration. In particular, duration was consistently smaller in 2004. In 2016, an increase in haul duration was estimated in week 48 (5 vs. 3 h on average across years).

Bycatch risk was maximum in week 1 (around 0.1) and decreased to almost 0 from week 8 onwards. 2017, 2019, and 2020 were the years with the highest estimated bycatch risk in the first 8 weeks. In particular, the risk was as high as 0.20 in 2017 for the first four consecutive years of the year. Two years prior to 2012 were associated with an increased risk between weeks 30 and 36. The year 2016 showed a rise in bycatch risk in week 48. Bycatch risk and haul duration were positively correlated with weeks in years associated with the highest risk and also having the longest haul duration. Numbers of hauls per DaS were negatively correlated with weeks with longer haul duration. There was little variation across years in numbers of hauls per DaS, but substantial within year variations.

Spatial variations in bycatch risk and haul duration are available as supplementary information. There were noteworthy differences between divisions regarding bycatch risk (see supplementary information). The overall signal was similar to the one observed in **Figure 3** with the highest risk values estimated between weeks 1 and 8. Risk in 2017 and 2019 was higher by a factor of 5 in week 1 compared to other years. After week 8, this difference disappeared. With respect to divisions, division 8.a. was the one with the highest bycatch risk, with an estimate as high as 0.50 in winter 2017 and 2019.

3.2. Number of Bycaught Dolphins

The estimated total number of bycaught dolphins for each year is reported in **Table 5**. The study area was further divided into three strata: a neretic stratum in ICES subarea 7 (divisions 7.defgh) and another in subarea 8 (divisions 8.abc); and an oceanic stratum spanning subareas 7 and 8 (divisions 7.j and 8.d). Estimates were the lowest in the oceanic stratum of the study area and the largest in the neretic stratum spanning ICES subarea 8. The largest bycatch estimate was in 2017, with a posterior median of > 600 common dolphins bycaught in PTM operating in the neretic stratum spanning ICES subarea 8. There were large between-year variations in estimates, ranging from less than a hundred (in 2018) to more than one thousand (in 2017). Uncertainties around model-based estimates were also large.

3.3. Comparison and Correlations With Strandings

Strandings data were used to estimate common dolphins mortality due to fisheries following method described in Peltier et al. (2016) for each month from 1990 to 2020. Stranding-based estimates aggregate mortality due to all fisheries and do not distinguish between gears or métiers. Nevertheless, we correlated stranding-based estimates with our model-based estimates of mortality from PTM flying the French flag both between years (**Figure 4**) and within each year (**Figure 4**). For yearly estimates, correlations were computed on raw and standardised (mean centered and unit variance) values (**Figure 4**). Model-based estimates of bycatch by PTM were always below stranding-based estimates (which do not allow to disaggregate by métiers) save for 2010 (**Figure 4**). In 2010, model-based and stranding based estimates were 465 and 343, respectively, with a large overlap in credibility interval. At the year level, the Pearson correlation between stranding-based and model-based estimates was 0.25. Yearly variations between the two time series were more in phase from 2015 onwards (**Figure 4**). At the within year (between month) level, correlations between the two time-series were always positive. These within year correlations generally increased by 47% (median) when model-based estimates were aggregated by month with a lag

TABLE 5 | Model-based estimates of common dolphin bycaught in PTM in the study area.

Year	Neretic 7	Neretic 8	Oceanic	Total
2004	${}_{0}48_{248}$	${}_{0}177_{876}$	${}_{0}1_{10}$	${}_{0}227_{1134}$
2005	${}_{0}56_{302}$	${}_{0}235_{1101}$	${}_{0}2_{15}$	${}_{0}293_{1417}$
2006	${}_{0}77_{378}$	${}_{0}208_{923}$	${}_{0}0_{3}$	${}_{0}286_{1303}$
2007	${}_{15}45_{102}$	${}_{0}29_{111}$	${}_{1}2_{8}$	${}_{16}77_{219}$
2008	${}_{1}18_{63}$	${}_{11}46_{125}$	${}_{0}0_{4}$	${}_{12}65_{190}$
2009	${}_{10}94_{248}$	${}_{172}315_{568}$	${}_{0}1_{6}$	${}_{183}412_{820}$
2010	${}_{0}119_{537}$	${}_{4}112_{454}$	${}_{0}0_{3}$	${}_{4}232_{994}$
2011	${}_{9}128_{359}$	${}_{0}61_{270}$	${}_{0}1_{7}$	${}_{9}191_{635}$
2012	${}_{22}233_{667}$	${}_{0}129_{511}$	${}_{0}3_{13}$	${}_{23}366_{1190}$
2013	${}_{13}315_{1086}$	${}_{0}105_{442}$	${}_{0}5_{25}$	${}_{13}426_{1552}$
2014	${}_{0}33_{158}$	${}_{0}50_{224}$	${}_{0}0_{3}$	${}_{0}84_{384}$
2015	${}_{0}14_{71}$	${}_{2}78_{368}$	${}_{0}1_{8}$	${}_{2}94_{446}$
2016	${}_{0}15_{76}$	${}_{55}255_{852}$	${}_{0}0_{3}$	${}_{55}270_{929}$
2017	${}_{0}18_{61}$	${}_{156}600_{1355}$	${}_{0}0_{1}$	${}_{156}618_{1415}$
2018	${}_{0}2_{15}$	${}_{1}31_{147}$	${}_{0}0_{2}$	${}_{1}35_{163}$
2019	${}_{0}12_{40}$	${}_{59}203_{391}$	${}_{0}1_{6}$	${}_{59}216_{441}$
2020	${}_{0}6_{27}$	${}_{4}50_{159}$	${}_{0}0_{5}$	${}_{4}57_{190}$

Divisions 7.j and 8.d are labelled "Oceanic," divisions 7.defgh are labelled "Neritic 7," and divisions 8.abc are labelled "Neritic 8." Estimates (posterior median) are reported with the lower and upper bound of a 80% credibility interval (Louis and Zeger, 2009).

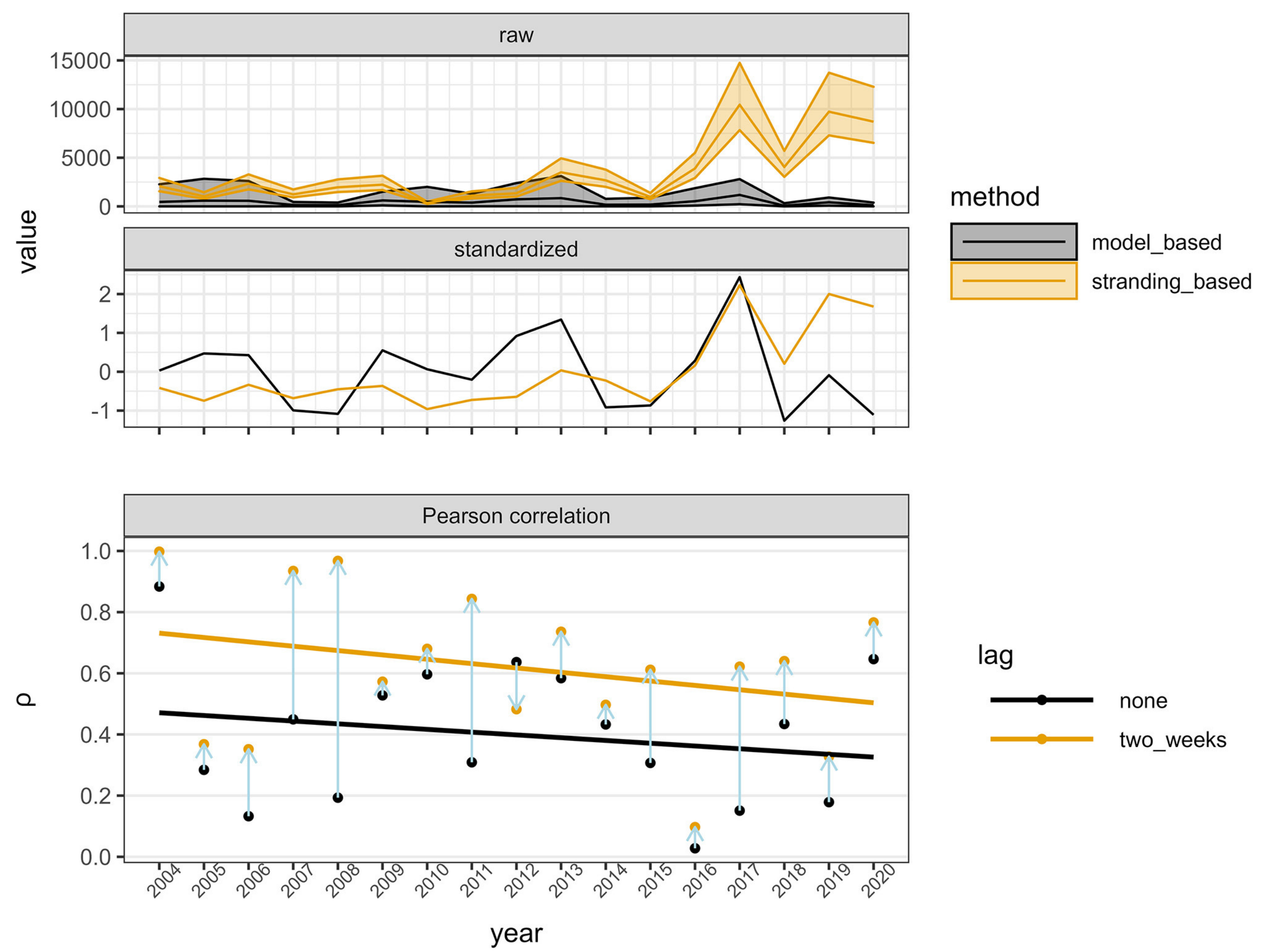


FIGURE 4 | Comparison of model-based (orange solid line) and stranding-based estimates (dark violet solid line) of common dolphin bycatch. **Upper**: Raw estimates with uncertainty intervals (80% for model-based estimates and 95% for stranding-based estimates); **Middle**: standardised (mean centred and unit variance) estimates. **Bottom**: Year-level and month-level (within each year) correlations were computed. At the within year (between month level), correlations were computed with and without 2-weeks lag. Change in the magnitude of the Pearson correlation coefficient is shown with an arrow.

of 2 weeks to account for drift (**Figure 4**). The temporal trend in within year correlation was negative over the study period.

4. DISCUSSION

From a non-representative sample of bycatch events of common dolphins collected over more than 15 years, we estimated bycatch risk and number of dolphins bycaught in PTM. Leveraging recent methodological developments in the analysis of non-representative samples (Gao et al., 2019; Authier et al., 2021), we built a joint model of bycatch risk, haul duration, and haul number per DaS to investigate changes within and between years in common dolphin bycatch. The years 2017 and 2019 were associated with the highest bycatch risk and the longest haul duration in winter.

4.1. Within-Year Variations in Bycatch Risk

We uncovered the within-year pattern in bycatch risk of common dolphins. Bycatch risk is the highest in winter, during the first weeks of a calendar year. This pattern is largely congruent with the pattern seen in strandings of common dolphins in the Bay of Biscay (Gilbert et al., 2021). Both stranding and observer data, which are independent, identified 2017 and 2019 as years with the highest risk of bycatch (Gilbert et al., 2021; Peltier et al., 2021). A limitation of stranding data is how the location of bycatch events must be inferred with reverse drift modelling (Peltier and Ridoux, 2015). The ObsMer data in contrast included geolocalised bycatch events, with a spatial resolution at the level of ICES divisions kept for analysis. Despite this coarse resolution, we could identify divisions 8.a and 8.b as the ones with the highest risk of bycatch by PTM.

The ICES Working Group on Bycatch (WGBYC) estimate bycatch of protected species, including common dolphins, in the

North East Atlantic. Using data collected by onboard observers collected between 2005 and 2017, bycatch rates for ICES divisions on the continental shelf of the Bay of Biscay were estimated with ratio estimators (ICES, 2019). These estimates are not produced at the week level, but ICES (2019) also identified divisions 8.a and 8.b as the ones with the highest of bycatch in midwater trawls for common dolphins over the period 2005–2017 (p. 61). ICES (2019) estimated yearly rates ranging between 0.285 and 0.372 dolphins per DaS and warned against extrapolation given the low observer coverage (p. 61). Our model-based approach overcomes this limitation (Authier et al., 2021) and was able to identify, within each year, that weeks 3 to 5 were the ones with the highest bycatch numbers for both divisions 8.a and 8.b. These results were concomitant with the seasonal stranding pattern observed each year on the French seashore (that is, winter strandings; Gilbert et al., 2021): around 80% of all common dolphin strandings on the French Atlantic seashore is observed between the end of January and the beginning of April.

A key feature of our model-based approach is how it leverages correlations between bycatch risk, haul duration, and number of hauls per DaS (**Figure 3**). Some of the correlations are expected, such as the negative correlation between haul duration and the number of hauls per DaS. However, average haul duration is not constant within a year, with the variations reflecting the change in the commercial fish species targeted by PTM at different time of the year. These variations at the week-level were quite substantial and were taken into account when estimating bycatch in our model. There was a positive correlation between haul duration and a bycatch risk, with at least a two-fold increase in the later when haul duration exceeds 5 h (**Figure 3**). This was particularly evident in weeks 1 to 5 in 2017 and 2019 and week 48 in 2016. The latter was due to a single fishing trip with 5 hauls that lasted > 10 h, each of which resulted in a bycatch event. We recommend, in light of the within-year pattern in haul duration (**Figure 3**), to investigate management actions and mitigation measures on limiting haul duration in winter to assess whether bycatch may also be reduced.

Another possible mitigation measure is to manage common dolphin interactions with PTM with spatio-temporal closures (and acoustic repulsive devices such as pingers) during the first week of a year, when bycatch is the highest. Such measures were explored by WKEMBYC (ICES, 2020d) to reduce bycatch mortality across several scenarios. The performance of each scenario was assessed with the Potential Biological Removal (Wade, 1998), bycatch reduction rate, and fishing effort reduction rate. WKEMBYC (ICES, 2020d) defined an efficiency score by the ratio between the latter two rates. This efficiency score is a trade-off between the expected bycatch reduction and the cost for the fishing industry (without direct economic consideration). WKEMBYC (ICES, 2020d) identified one scenario (scenario L) wherein 2 months closure from mid January to mid March for all fishing métier (and the use of pingers for "Bottom pair trawlers" (PTB) and PTM the rest of the year) was efficient. This scenario appears as a good compromise between bycatch reduction and a reduced cost for the industry. Another efficient scenario (scenario N) involves a 3-month closure from January to March and another 1 month from mid July to mid August for all métier (and the use of pingers for PTB and PTM the rest of year). This scenario can achieve the highest level of bycatch reduction but incurs a high cost to the industry. However, scenarios considered by WKEMBYC are emergency measures meant to reduce punctually common dolphin bycatch. Systematic spatio-temporal closures, which are usually not favoured by the fisheries, were not considered and remained to be explored. In contrast, mitigation measures relying on the large scale deployment of acoustic repulsive devices and the development of new such devices are underway (e.g., in the CetAMBICion project[4]).

4.2. Between-Year Variations in Bycatch Risk

There were large between-year variations in model-based estimates of common dolphin bycatch in the study area. To some extent, these variations were explained by other factors than bycatch risk. For example, the (posterior median) estimate is >600 dolphins in 2017 down to <100 in 2018. The total effort in DaS in the Bay of Biscay (divisions 8.a and 8.b) in the first 10 weeks of 2017, when bycatch risk was highest, is two times the value of total effort in 2018. The median number of dolphins involved in a bycatch event in 2017 was also two times the number in 2018 (2 and 1, respectively, **Table 2**). All else being equal, the estimate for 2017 is expected to be at least four times that of 2018. A further improvement of the model-based approach is to jointly model the number of dolphins involved in a bycatch event. This improvement will require accomodating a large overdispersion, but there were however less than 100 such events in the dataset and we chose to use the median. This is a cautionary choice since the median is less sensitive to the few events for more than 10 dolphins. The uncertainty in the median number of dolphins involved in a bycatch event is currently ignored: incorporating it in future development will further widen credibility intervals (which are already large; Authier et al., 2021). Thus, the model-based estimates are conservative estimates of bycatch by PTM.

Bycatch risk was also very variable between years: the large between-year variations may be due to ecological factors. Bycatch risk results from both fisheries activity within a particular division at a particular time and dolphin presence. The highest bycatch risk values were estimated for the 8 or 10 first weeks of each year within each division of the study area (**Figure 3**). Astarloa et al. (2021) found evidence of an increased abundance of common dolphins in the Bay of Biscay in recent years but weak correlations with biological and oceanographic variables, such as chlorophyll a concentration or sea surface temperature. ICES divisions 8.a and 8.b cover the continental shelf parts of the Bay of Biscay (**Figure 1**). These neritic divisions are witness to sub-mesoscale oceanographic processes and nutrient offloads from the Gironde estuary. Gilbert et al., 2021 correlated eddies and frontal structures with common dolphin mortality areas at sea in the Bay of Biscay (although these authors also concluded that oceanographic accounted for a small fraction of the overall variance in stranding numbers). In winter, the Bay of Biscay

---

[4] https://www.cetambicion-project.eu/

environment is characterised by a seasonal cross-shore (West to East) surface temperature gradient with the lowest temperature close to shore and intense frontal activity parallel to the coast (North to South) (Yelekçi et al., 2017). These frontal structures are freshwater fronts, correlated to the mixing of oceanic waters and cold freshwater inputs from river plumes (Yelekçi et al., 2017). These seasonal fronts may be targeted by both fisheries and common dolphins as areas where fish aggregate, thereby putting the latter at risk of bycatch by the former. In July and August, the mesoscale dynamic activity of the Bay of Biscay is rather different than in winter. In summer, there are mainly fronts due to tidal flow (Yelekçi et al., 2017). Summer tidal fronts are quite consistent from 1 year to the next because they are correlated to a repetitive process (i.e., tides) (Yelekçi et al., 2017). During summer, the main frontal activity is a seasonal tidal front, called the Ushant Front and located in front of the French Finistère county (Yelekçi et al., 2017). Its activity peaks in July and August (Yelekçi et al., 2017). We can speculate that the years associated with a high bycatch risk were also those when oceanographic processes favouring the spatial overlap (mediated by fish species; Spitz et al., 2013; Astarloa et al., 2021) between fisheries and common dolphins were particularly operant.

Stranding records are an independent source of data for estimating the number of bycaught dolphins (Peltier et al., 2016). Reverse drift modelling allows the death location of each stranded dolphin showing bycatch evidence for each month between 1990 and 2020 to be inferred. Observed stranding tallies for each month can be corrected for both stranding and buoyancy probabilities (Peltier and Ridoux, 2015). Reverse drift modelling cannot disaggregate estimates by métiers or fisheries but provides an independent estimates of total mortality due to bycatch in the study area: bycatch mortality due to PTM should be lower than the total estimated from strandings. This was verified for all years save for 2010, but uncertainties were large and credibility intervals had a large overlap. While the correlation between model-based and stranding-based estimates was modest at the year level, it was larger at the within-year level, especially after accounting for a lag due to drift (**Figure 4**). The magnitude of the within-year correlation decreased between 2005 and 2020. One interpretation is that of a change in the relative contribution of PTM in total dolphin mortality over time, with PTM having a lesser impact on common dolphins in recent years compared to the 2000s.

4.3. Limitations and Improvements

The model used to estimate the bycatch of common dolphins in PTM has been developed to address the issue of non-representative sampling (Authier et al., 2021). It relies on a post-stratification step that requires accurate effort data at the scale of the whole fleet. The effort measurement retained was that of DaS as in international working groups (e.g., ICES WGBYC; ICES, 2019). Leveraging this important piece of information required the joint modelling of risk at the haul level, haul duration, and that of the average number of hauls per DaS. This modelling choice proved successful for PTM but need not be so for other métiers, in particular for passive gears such as gillnets and setnets. In the later case, a better measure of effort at haul level is soak time, taking into account net length and height, and possible mesh size. These pieces of information may be difficult to collect and retrospectively obtain for post-stratification. Any method seeking to scale up a sample from onboard observer to the whole fleet must confront the difficult issue of accurate measurement and quantification of effort. The model developed for PTM may not necessarily transfer seamlessly to other gears or métiers.

Since 2021, PTM flying the French flag are required to use deterrent acoustic devices (pingers[5]). If these devices are efficient to reduce bycatch risk, this may be taken into account in the model, by adding a covariate in Equation (1). Doing so requires on the other hand to post-stratify on that covariate, which is likely to be a major hurdle. Ignoring the deployment of pingers need not be problematic as the model allows for between- and within-year variations in bycatch risk. Large-scale deployment of pingers in 2021, if effective in reducing risk, will manifest itself in an estimated risk lower compared to previous years. In other words, the model does not have to necessarily take into account all haul-level covariates as long as the aim is prediction rather than explanation (Authier et al., 2021). Taking explicitly into account the pinger effect is only required to make sense of the between- and within-year variations in risk, but not necessarily to estimate those variations.

While Authier et al. (2021) concluded on increased accuracy of using regularised multi-level regression with post-stratification to estimate bycatch with observer data, they also found that estimated precision was low. This was also the case in this study (**Table 5**). A simple way to increase precision is to include self-declared positive bycatch events from fishermen in Equation (7) and (8). Doing so provides a strong incentive for compliance on self-declaration and would result in increased precision as a greater number of hauls (and possibly DaS) would be monitored. Ultimately, full compliance would render modelling moot as bycatch would be perfectly known, if all events were properly recorded (e.g., with Electronic Remote Monitoring) or reported systematically and accurately in logbooks.

4.4. Implications for Common Dolphin Conservation

The common dolphin is one of the most abundant delphinid species within the North-East Atlantic (Hammond et al., 2021). This species may be described as a “keystone species” and an “umbrella species” considering its ecological importance (Murphy et al., 2021). The large additional mortality due to anthropogenic activities on this species triggered a dedicated working group on emergency measures in 2020: the workshop on fisheries emergencies measures to minimise bycatch of short-beaked common dolphins in the Bay of Biscay and harbour porpoises in the Baltic Sea (WKEMBYC) took place remotely in spring 2020 (ICES, 2020d) and informed an ICES advice that same year (ICES, 2020c). This advice led to an infringement procedure issued in July 2020 against France for

---

[5] https://www.mer.gouv.fr/protecting-cetaceans-annick-giradin-presents-7-commitments-made-french-state-fishermen-and

failing its obligations under the Habitats Directive, which lists the common dolphin as a species requiring full protection on its Annex IV. The same day, the Paris Administrative Court of Justice condemned the French government for failing to transpose and apply in a timely manner the dispositions of the Habitats Directive and Technical Measures regulating fisheries[6] (in French). Following the unprecedented number of strandings in 2017, a national working group with fishermen, their representatives, government officials, Non-Governmental Organizations, and academics was initiated to address the bycatch issue (Peltier et al., 2021). One recommended action was to improve estimates of bycatch due to high-risk métiers, and to develop adequate methodologies to analyse data from non-representative samples (Authier et al., 2021). The present work reports on a case study on PTM and operating for a large part in the Bay of Biscay, and to a lesser extent in the Celtic seas. The model-based estimates (i) can inform on pressures acting on common dolphins as required by the Marine Strategy Framework Directive (EU 2008/56) and (ii) heed ICES recommendation to develop estimation methods to make the best use of already collected data to inform management in a timely manner (ICES, 2020c).

Using a Potential Biological Removal (PBR) approach (Wade, 1998), ICES (2020d) estimated a removal limit of common dolphin for the whole North-East Atlantic of 4, 926 individuals. An annual bycatch no greater than PBR would allow the population of common dolphins to recover to or be maintained at or above 50% of carrying capacity with a probability of 0.95 (Wade et al., 2021). This conservation objective is, however, different from the ASCOBANS interim objective "to restore and/or maintain stocks/populations to 80% or more of the carrying capacity." Genu et al. (this issue) tuned a modified PBR to a quantitative interpretation of the ASCOBANS interim objective: "a population should be able to recover to or be maintained at 80% of carrying capacity, with probability 0.8, within a 100-year period." The removals limit computed using the modified PBR was down to 985 animals (that is, one fifth of PBR; Genu et al., this issue): in 2017, the estimated bycatch due to PTM and operating the Bay of Biscay amounted to more than 60% of this limit (**Table 5**). In recent years, the estimated contribution of this métier relative to the modified PBR remained large according to our results. Other fishing métiers could potentially impact the common dolphins in the Bay of Biscay resulting in mortality exceeding the threshold inferred by both modified and non-modified PBR. Regarding vessels flying the French flag, gill trammel netters (GTR), gillnetters (GNS), and pair trawlers were potentially associated with common dolphin mortality in ICES divisions 8.a and 8.b for different years (regarding the co-occurrence of mortality and fishing effort) (Peltier et al., 2021). Estimating the contribution of each métiers to overall mortality remains a difficult endeavor. Regarding the PBR removals limit used in WKEMBYC (ICES, 2020d), the overall mortality considering all the fishing métiers exceed PBR, notably from 2016 to 2019, years associated with the suspected highest contribution for the métiers listed above.

5. CONCLUSION

We have provided a case study on estimating bycatch of common dolphins by PTM and operating in the Bay of Biscay from a non-representative sample of bycatch events collected by non-dedicated onboard observers. Leveraging recent methodological developments in statistical modelling, we have illustrated how to use imperfect but currently available data to inform management. Our contribution thus heeds two recent recommendations: to use adequate estimation methods on existing data and to gauge the resulting estimates against threshold values for incidental bycatch, tuned to relevant conservation objectives. We evidenced a substantial contribution of PTM to common dolphin bycatch in the Bay of Biscay, especially in 2017. Considering the entire time series and the correlations with the estimates made from strandings, it is possible that other métiers than PTM were associated with bycatch, especially in recent years. Currently, the main mitigation measures recommended are spatio-temporal closures and the widespread use of acoustic deterrent devices on PTM/OTM and PTB to repel dolphins (ICES, 2020b). Spatio-temporal closures were not implemented in 2021 but systematic and mandatory deployment of pingers on trawls were[7]. Relevant to management in broadening the scope of potential measures is the evidenced correlation between bycatch risk and haul duration: further studies should investigate limiting haul duration (for example, below 5 h) as a complementary mitigation strategy, especially in winter.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/supplementary material, further inquiries can be directed to the corresponding author/s.

ETHICS STATEMENT

Ethical review and approval was not required for the animal study because this study uses data that is collected as part of the Data Collection Framework under the Common Fishery Policy of the European Union.

AUTHOR CONTRIBUTIONS

ER and MA led the analyses, the conception, and writing of the paper. LD, TC, and SD curated and prepared the data. MG designed the shiny application. All authors support in analyses, paper conception, and writing, contributed to the article, and approved the submitted version.

---

[6] http://paris.tribunal-administratif.fr/content/download/172866/1715763/version/1/file/1901535.pdf

[7] https://www.mer.gouv.fr/protecting-cetaceans-annick-giradin-presents-7-commitments-made-french-state-fishermen-and

FUNDING

This work was supported in the form of salaries from (ADERA to MA) and (HiDef to KM) but did not have any additional role in the study design, data collection, and analysis, decision to publish, or preparation of the manuscript.

ACKNOWLEDGMENTS

We thank the reviewers, Emilie Leblond, Thomas Rimaud, and Julien Lamothe for their constructive comments. We are indebted to Mikaël Guichard and the dedicated staff at the Mésocentre de Calcul de Poitou Charentes.

REFERENCES

Alverson, D. L., Freeberg, M. H., Murawski, S. A., and Pope, J. (1994). *A Global Assessment of Fisheries Bycatch and Discards*, Vol. 339. Rome: Food & Agriculture Organization.

Anderson, C. R., Herrera, M., Ilangakoon, A. D., Koya, K. M., Moazzam, M., Mustika, P. L., et al. (2020). Cetacean bycatch in Indian Ocean tuna gillnet fisheries. *Endang. Species Res.* 41, 39–53. doi: 10.3354/esr01008

Anonymous (2016). *Rapport annuel sur la mise en øeuvre du règlement Européen (CE) 812/2004 établissant les mesures relatives aux captures accidentelles de cétacés dans les pêcheries*. Rapport annuel réglementaire. Technical report, Direction des Pêches Maritimes et de L'aquaculture (DPMA).

Anonymous (2019). *Commission Implementing Decision (EU) 2019/910 of 13 March 2019 Establishing the multiannual Union Programme for the Collection and Management of Biological, Environmental, Technical and Socioeconomic Data in the Fisheries and Aquaculture Sectors*. Brussels: European Comission

Astarloa, A., Louzao, M., Andrade, J., Berrow, S., Boisseau, O., Brereton, T., et al. (2021). The role of climate, oceanography, and prey in driving decadal spatio-temporal patterns of a highly mobile top predator. *Front. Mar. Sci.* 8:665474. doi: 10.3389/fmars.2021.665474

Authier, M., Rouby, E., and Macleod, K. (2021). *Estimating Bycatch from Non-Representative Samples with (Regularized) Multilevel Regression with Post-Stratification*. Technical report, Observatoire Pelagis UMS3462 CNRS-LRUniv.

Avila, I. C., Kaschner, K., and Dormann, C. F. (2018). Current global risks to marine mammals: taking stock of the threats. *Biol. Conserv.* 221, 44–58. doi: 10.1016/j.biocon.2018.02.021

Babcock, E. A., Pikitch, E. K., and Hudson, C. G. (2003). *How Much Observer Coverage Is Enough to Adequately Estimate Bycatch*? Miami, FL: Pew Institute of Ocean Science.

Bearzi, G., and Reeves, R. R. (2021). Shifting baselines of cetacean conservation in Europe. *ICES J. Mar. Sci.* 78, 2337–2341. doi: 10.1093/icesjms/fsab137

Benoît, H., and Allard, J. (2009). Can the data from at-sea observer surveys be used to make general inference about catch composition and discards? *Can. J. Fish. Aquat. Sci.* 66, 2025–2039. doi: 10.1139/F09-116

Bessesen, B. (2018). *Vaquita - Science, Politics, and Crime in the Sea of Cortez*. Washington: Island Press.

Brownell, R. J., Reeves, R., Read, A., Smith, B., Thomas, P., Ralls, K., et al. (2019). Bycatch in gillnet fisheries threatens critically endangered small cetaceans and other aquatic megafauna. *Endang. Species Res.* 40, 285–296. doi: 10.3354/esr00994

Carpenter, B., Gelman, A., Hoffman, M. D., Lee, D., Goodrich, B., Betancourt, M., et al. (2017). `Stan`: a probabilistic programming language. *J. Stat. Softw.* 76, 1–32. doi: 10.18637/jss.v076.i01

Chen, Z., and Dunson, D. (2003). Random effects selection in linear mixed models. *Biometrics* 59, 762–769. doi: 10.1111/j.0006-341X.2003.00089.x

Cornou, A. S., Goascoz, N., Scavinner, M., Prioul, F., Sabbio, A., Dubroca, L., et al. (2018). *Captures et rejets des métiers de pêche français. résultats des observations à bord des navires de pêche professionnelle en 2017*. Technical report, Ifremer.

Course, G. P. (2021). *Monitoring Cetacean Bycatch: An Analysis of Different Methods Aboard Commercial Fishing Vessels*. ASCOBANS Secretariat, Bonn.

Davies, R., Cripps, S., Nickson, A., and Porter, G. (2009). Defining and estimating global marine fisheries bycatch. *Mar. Policy* 33, 661–672. doi: 10.1016/j.marpol.2009.01.003

Fernández-Contreras, M., Cardona, L., Lockyer, C., and Aguilar, A. (2010). Incidental bycatch of short-beaked common dolphins (*Delphinus delphis*) by pairtrawlers off Northwestern Spain. *ICES J. Mar. Sci.* 67, 1732–1738. doi: 10.1093/icesjms/fsq077

Gao, Y., Kennedy, L., Simpson, D., and Gelman, A. (2019). *Improving Multilevel Regression and Poststratification with Structured Priors*. Report, University of Toronto & Columbia University.

García-Baron, I., Santos, M. B., Uriarte, A., Inchausti, J. I., Escribano, J. M., Albisu, J., et al. (2019). which are the main threats affecting the marine megafauna in the Bay of Biscay? *Continent. Shelf Res.* 186, 1–12. doi: 10.1016/j.csr.2019.07.009

Gelman, A., Hill, J., and Vehtari, A. (2021). *Regression and Other Stories, 1st Edn*. Cambridge: Cambridge University Press. doi: 10.1017/9781139161879

Genu, M., Gilles, A., Hammond, P., Macleod, K., Paillé, J., Paradinas, I. A., et al. (this issue). Evaluating strategies for managing anthropogenic mortality on marine mammals: an `R` implementation with the package `RLA`. in preparation.

Gilbert, L., Rouby, E., Tew-Kaï, E., Spitz, J., Peltier, H., Quilfen, V., et al. (2021). Spatiotemporal models highlight influence of oceanographic conditions on common dolphin bycatch risk in the Bay of Biscay. *Mar. Ecol. Prog. Ser.* 679, 195–212. doi: 10.3354/meps13894

Griffin, J., and Brown, P. (2017). Hierarchical shrinkage priors for regression models. *Bayesian Anal.* 12, 135–159. doi: 10.1214/15-BA990

Hall, M. A. (1996). On bycatches. *Rev. Fish Biol. Fish.* 6, 319–352. doi: 10.1007/BF00122585

Hammond, P. S., Lacey, C., Gilles, A., Viquerat, S., Börjesson, P., Herr, H., et al. (2021). *Estimates of Cetacean Abundance in European Atlantic Waters in Summer 2016 from the SCANS-III Aerial and Shipboard Surveys*. Technical report, Sea Mammal Research Unit, University of Saint Andrews.

Hoffman, M. D., and Gelman, A. (2014). The no-U-turn sampler: adaptively setting path lengths in Hamiltonian Monte Carlo. *J. Mach. Learn. Res.* 15, 1593–1623.

ICES (2019). *Working Group on Bycatch of Protected Species (WGBYC)*. Technical report, International Council for the Exploration of the Sea, Faro.

ICES (2020a). *Bycatch of Protected and Potentially Vulnerable Marine Vertebrates - Review of National Reports under Council Regulation (EC) No. 812/2004 and Other Information. ICES Advice: Ecoregions in the Northeast Atlantic and Adjacent Seas*. Technical report.

ICES (2020b). *EU Request on Emergency Measures to Prevent Bycatch of Common Dolphin (Delphinus delphis) and Baltic Proper Harbour Porpoise (Phocoena phocoena) in the Northeast Atlantic*. Report of the ICES Advisory Committee.

ICES (2020c). *Road Map for ICES Bycatch Advice on Protected, Endangered, and Threatened Species*. Report of the ICES Advisory Committee.

ICES (2020d). *Workshop on Fisheries Emergency Measures to Minimize BYCatch of Short-Beaked Common Dolphins in the Bay of Biscay and Harbour Porpoise in the Baltic Sea (WKEMBYC)*. International Council for the Exploration of the Sea.

Kuiken, T. (1994). "Review of the criteria for the diagnosis of by-catch in cetaceans," in *Diagnosis of By-Catch in Cetaceans. Proc. 2nd. ECS Workshop on Cetacean Pathology* (Montpellier), 38–43.

Louis, T., and Zeger, S. (2009). Effective communication of standard error and confidence interval. *Biostatistics* 10, 1–2. doi: 10.1093/biostatistics/kxn014

Luck, C., Jessopp, M., Tully, O., Cosgrove, R., Rogan, E., and Cronin, M. (2020). Estimating protected species bycatch from limited observer coverage: a case study of seal bycatch in static net fisheries. *Glob. Ecol. Conserv.* 24:e01213. doi: 10.1016/j.gecco.2020.e01213

Moore, J., Heinemann, D., Francis, T., Hammond, P., Long, K. J., Punt, A. E., et al. (2021). Estimating bycatch mortality for marine mammals: concepts and best practices. *Front. Mar. Sci.* 8, 752356. doi: 10.3389/fmars.2021.752356

Murphy, S., Evans, P. G. H., Pinn, E., and Pierce, G. J. (2021). Conservation management of common dolphins: lessons learned from the North-East Atlantic. *Aquat. Conserv.* 31, 137–166. doi: 10.1002/aqc.3212

Murphy, S., Pinn, E. H., and Jepson, P. D. (2013). "The short-beaked common dolphin (*Delphinus delphis*) in the North-East Atlantic: distribution, ecology, management and conservation status," in *Oceanography and Marine Biology: an Annual Review*, ASCOBANS eds. (The Hague: CRC Press), 123–210.

Peltier, H., Authier, M., Caurant, F., Dabin, W., Daniel, P., Dars, C., et al. (2021). In the wrong place at the wrong time: identifying spatiotemporal co-occurrence of bycaught common dolphins and fisheries in the Bay of Biscay (NE Atlantic) from 2010 to 2019. *Front. Mar. Sci*. 8:617342. doi: 10.3389/fmars.2021.617342

Peltier, H., Authier, M., Deaville, R., Dabin, W., Jepson, P. D., van Canneyt, O., et al. (2016). Small cetacean bycatch as estimated from stranding schemes: the common dolphin case in the Northeast Atlantic. *Environ. Sci. Policy* 63, 7–18. doi: 10.1016/j.envsci.2016.05.004

Peltier, H., Dabin, W., Daniel, P., Van Canneyt, O., Dorémus, G., Huon, M., et al. (2012). The significance of stranding data as indicators of cetacean populations at sea: modelling the drift of cetacean carcasses. *Ecol. Indicat*. 18, 278–290. doi: 10.1016/j.ecolind.2011.11.014

Peltier, H., Jepson, P., Dabin, W., Deaville, R., Daniel, P., Van Canneyt, O., et al. (2014). The contribution of stranding data to monitoring and conservation strategies for cetaceans: developing spatially explicit mortality indicators for common dolphins (*Delphinus delphis*) in the eastern north-atlantic. *Ecol. Indicat*. 39, 203–214. doi: 10.1016/j.ecolind.2013.12.019

Peltier, H., and Ridoux, V. (2015). Marine megavertebrates adrift: a framework for the interpretation of stranding data in perspective of the European marine strategy framework directive and other regional agreements. *Environ. Sci. Policy* 54, 240–247. doi: 10.1016/j.envsci.2015.07.013

Pérez, M., Perrichi, L. R., and Ramírez, I. (2017). The scaled Beta2 distribution as a robust prior for scales. *Bayesian Anal*. 12, 615–637. doi: 10.1214/16-BA1015

R Core Team (2020). *R: A Language and Environment for Statistical Computing*. Vienna: R Foundation for Statistical Computing.

Rayner, S. (2012). Uncomfortable knowledge: the social construction of ignorance in science and environmental policy discourses. *Econ. Soc*. 41, 107–125. doi: 10.1080/03085147.2011.637335

Read, A. J. (2008). The looming crisis: interactions between marine mammals and fisheries. *J. Mammal*. 89, 541–548. doi: 10.1644/07-MAMM-S-315R1.1

Read, A. J., Drinker, P., and Northridge, S. (2006). Bycatch of marine mammals in us and global fisheries. *Conserv. Biol*. 20, 163–169. doi: 10.1111/j.1523-1739.2006.00338.x

Rogan, E., and Mackey, M. (2007). Megafauna bycatch in drift nets for albacore tuna (*Thunnus alalunga*) in the ne atlantic. *Fish. Res*. 86, 6–14. doi: 10.1016/j.fishres.2007.02.013

Rogan, E., Read, A. J., and Berggren, P. (2021). Empty promises: the European union is failing to protect dolphins and porpoises from fisheries by-catch. *Fish Fish*. 22, 1–5. doi: 10.1111/faf.12556

Sala, A., Konrad, C., and Doerner, H. (eds.). (2019). *Review of the Implementation of the EU Regulation on the Incidental Catches of Cetaceans (STECF-19-07)*. Publications Office of the European Union, Luxembourg.

Scarff, J. E. (1977). The international management of whales, dolphins, and porpoises: an interdisciplinary assessment. *Ecol. Law Quart*. 6, 323–427.

Schweder, T. (2000). Distortion of uncertainty in science: Antarctic fin whales in the 1950s. *J. Int. Wildlife Law Policy* 3, 73–92. doi: 10.1080/13880290009353947

Spitz, J., Chouvelon, T., Cardinaud, M., Kostecki, C., and Lorance, P. (2013). Prey preferences of adult sea bass *Dicentrarchus labrax* in the Northeastern Atlantic: implications for bycatch of common dolphin *Delphinus delphis*. *ICES J. Mar. Sci*. 70, 452–461. doi: 10.1093/icesjms/fss200

Stan Development Team (2020). *RStan: the R Interface to Stan*. R package version 2.21.2.

Systéme d'Information Halieutique (2017). *Données de production et d'effort de pêche (SACROIS) - 2003–2017*.

Thorson, J. T. (2019). Guidance for decisions using the vector autoregressive spatio-temporal (VAST) package in stock, ecosystem, habitat and climate assessments. *Fish. Res*. 210, 143–161. doi: 10.1016/j.fishres.2018.10.013

Thorson, J. T., Skaug, H. J., Kristensen, K., Shelton, A. O., Ward, E. J., Harms, J. H., et al. (2015). The importance of spatial models for estimating the strength of density dependence. *Ecology* 96, 1202–1212. doi: 10.1890/14-0739.1

Ulrich, C., and Doerner, H. (eds.). (2021). *Scientific, Technical and Economic Committee for Fisheries (STECF) - 66th Plenary Report (PLEN-21-01)*. Publications Office of the European Union, Luxembourg.

Wade, P. R. (1998). Calculating limits to the total allowable human-caused mortality of cetaceans and pinnipeds. *Mar. Mammal Sci*. 14, 1–37. doi: 10.1111/j.1748-7692.1998.tb00688.x

Wade, P. R., Long, K. J., Francis, T. B., Punt, A. E., Hammond, P. S., Heinemann, D., et al. (2021). Best practices for assessing and managing bycatch of marine mammals. *Front. Mar. Sci*. 8:757330. doi: 10.3389/fmars.2021.757330

Ward, E. J., Jannot, J. E., Lee, Y.-W., Ono, K., Shelton, A. O., and Thorson, J. T. (2015). Using spatiotemporal species distribution models to identify temporally evolving hotspots of species co-occurrence. *Ecol. Appl*. 25, 2198–2209. doi: 10.1890/15-0051.1

Yelekçi, Ö., Charria, G., Capet, X., Reverdin, G., Sudre, J., and Yahia, H. (2017). Spatial and seasonal distributions of frontal activity over the French continental shelf in the Bay of Biscay. *Continent. Shelf Res*. 144, 65–79. doi: 10.1016/j.csr.2017.06.015

Conflict of Interest: MA and HP are employed by the commercial company ADERA which did not play any role in this study beyond that of employer.

The remaining authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

ORIGINAL RESEARCH
published: 28 January 2022
doi: 10.3389/fmars.2021.779448

Low Resighting Rate of Entangled Humpback Whales Within the California, Oregon, and Washington Region Based on Photo-Identification and Long-Term Life History Data

Jennifer Tackaberry[1]*, Elana Dobson[1], Kiirsten Flynn[1], Ted Cheeseman[2,3] and John Calambokidis[1]

[1] Cascadia Research Collective, Olympia, WA, United States, [2] Happywhale, Santa Cruz, CA, United States, [3] Marine Ecology Research Centre, Southern Cross University, Lismore, NSW, Australia

OPEN ACCESS

Edited by:
Randall Reeves,
Okapi Wildlife Associates, Canada

Reviewed by:
Paul R. Wade,
National Oceanic and Atmospheric Administration, United States
Yulia Ivashchenko,
SeaStar Scientific, United States

***Correspondence:**
Jennifer Tackaberry
jtackaberry@coastalstudies.org

Specialty section:
This article was submitted to Marine Megafauna, a section of the journal Frontiers in Marine Science

Received: 18 September 2021
Accepted: 23 December 2021
Published: 28 January 2022

Citation:
Tackaberry J, Dobson E, Flynn K, Cheeseman T and Calambokidis J (2022) Low Resighting Rate of Entangled Humpback Whales Within the California, Oregon, and Washington Region Based on Photo-Identification and Long-Term Life History Data. Front. Mar. Sci. 8:779448. doi: 10.3389/fmars.2021.779448

Bycatch, commonly referred to as entanglement, is a leading source of human-caused mortality of baleen whales. A better understanding of the individuals that survive (or perish from) their entanglement can help reduce the risk of bycatch by informing gear modifications and fisheries management. However, determining survival rates is restricted by the ability to track individuals once they become entangled. Historically, the effort to identify and resight individuals from known entanglement cases was low along the West Coast. The recent increase of entanglements and photo-identification efforts in the California, Oregon, and Washington region provides an opportunity to assess entangled humpback whales' resighting rates to better understand the effect of entanglements from the individual to the population level. We used photo-ID images of entangled humpback whales between 1982 and 2017 to examine pre- and post-entanglement sighting histories from longitudinal catalogs and life history data (Cascadia Research and Happywhale). We compared the entangled whales (*n* = 37) against control whales (*n* = 2,296), selected based on the date and location of the entanglement reports, to evaluate the deviation from the expected mortality rates (or lack of resighting) caused by entanglements and to help inform/support/test National Oceanic and Atmospheric Association's Serious Injury and Mortality (SI/M) index. Our results suggest that entangled whales were resighted less often than the control groups. Entangled whales with short pre-entanglement sighting histories and without post-entanglement resights did not match other feeding ground populations. Therefore, the higher proportion of entanglements with shorter sighting histories is likely due to their being alive for fewer years, indicating a higher risk of entanglement for younger whales. This indicates that entangled humpback whales may not be as large or strong as mature individuals, and future gear modifications should reflect that possibility. Additionally, the severity of the initial SI/M score aligned well with our resighting rates, though this worsened with the final score assigned. Continued effort to gather and improve data collection about entanglements will help enhance the SI/M determinations. Our findings show the value of photo-identification of entangled whales and how it dramatically increases our understanding of entanglements.

Keywords: entanglement, humpback whale, life history, *Megaptera novaeangliae*, photo-identification, resighting rate, survival, West Coast USA

INTRODUCTION

Bycatch is a leading source of human-caused mortality of baleen whales (Robbins and Mattila, 2004; Carretta et al., 2013; Pace et al., 2014). Unlike baleen whales, smaller species of fishes, birds, and mammals are usually caught, perish relatively quickly, and documented when the gear is hauled—providing a method to obtain accurate estimates of bycatch for specific fisheries. Unfortunately, baleen whales that become caught in fishing gear are usually strong enough to carry all or some of the gear away from the gearset location. This action results in few opportunities to see the animal once it becomes entangled, resulting in substantial under-reporting of bycatch rates, more commonly referred to as entanglement rates. Reports of entangled whales are estimated to capture under 10% of large whale entanglements (Robbins, 2009, 2012). Of the 10% of entanglements reported, even fewer opportunities exist to fully document the entanglement and remove the life-threatening material. Without intervention, life-threatening entanglements can result in a slow decline in health over months, or in some cases years, before the animal starves or succumbs to its injuries (Moore and van der Hoop, 2012; van der Hoop et al., 2017).

National Oceanic and Atmospheric Association (NOAA) started logging large whale entanglement reports on the West Coast of the US in 1982. The recent increase of entanglement reports along the U.S. West Coast provided an opportunity to assess the resighting rates of entangled humpback whales, *Megaptera novaeangliae*, in the California, Oregon, and Washington (CA-OR-WA) region. Although determining the resighting rate of entangled humpback whales seems straightforward, different spatial and temporal encounter rates along the West Coast complicate resighting rate comparisons to the overall humpback whale population.

The CA-OR-WA region includes roughly 1300 miles of coastline with human-use clustered by population centers and commercial ports. The majority of entanglement cases with proper photo-ID documentation occur in areas with higher whale watching effort or within range of a trained response team (**Figure 1**). One method to account for the different encounter rates along the West Coast is to use control groups based on the initial entanglement report's date and location. This method provides a comparison for entangled whales that involves individuals with similar opportunities to being resighted.

In contrast to resighting rates, serious injuries and mortality (SI/M) scoring is a method managers use to estimate how many individuals die due to known human interaction (Carretta et al., 2013, 2015, 2017, 2018, 2019, 2020). Established protocols to determine an entangled individual's survival likelihood are primarily based on well-studied, small populations of baleen whales in the North Atlantic (Guidelines for distinguishing serious from non-serious injury of marine mammals pursuant to the Marine Mammal Protection Act, 2012). These health and survival trends are extrapolated to all baleen whale species throughout the United States (Guidelines for distinguishing serious from non-serious injury of marine mammals pursuant to the Marine Mammal Protection Act, 2012). Unlike some of the populations in the North Atlantic, the CA-OR-WA humpback whale population encompasses a vast area, and sightings gaps of 10–20 years are not unusual for individuals in areas of low research effort. The current SI/M matrix is presently the most accurate method to estimate anthropogenic mortality of U.S. West Coast humpback whales. However, the resighting rates determined in this study can provide the first steps to gauge the accuracy of this matrix for the CA-OR-WA humpback whale population.

One major challenge in reducing large whale bycatch is not knowing how many individuals survive or die after becoming entangled. Once survival and resighting rates are determined, the lethal effects of different gear types or entanglement configurations can be defined. Further, which demographics and distinct population segments (DPS) are more at risk for entanglement-induced mortality can be determined. Our study is one step along that path to reduce large whale bycatch. This study used photo-IDs to provide the first estimates of resighting rates for entangled humpback whales in the CA-OR-WA region. Photo-IDs and data collected from entanglements linked with long-term population data allowed us to test the following hypotheses. (1) Entangled humpback whales have a lower resighting rate than other whales documented in the same region at a similar time of year as the initial entanglement report. (2) Entangled whales had a similar proportion of sighting history lengths (a proxy for age-class) as the control whales. (3) The initial entanglement report's location was within the same region the individual historically used and within the gearset deployment subregion. (4) The resighting rates of entangled whales within the CA-OR-WA region supported the expected survival estimates from NOAA's SI/M scores.

MATERIALS AND METHODS

National Marine Fisheries Service (NMFS) within NOAA provided records of large whale entanglement cases from 1982 to 2019 along the CA-OR-WA region. Each record included varying amounts of information, but all cases had the initial report's location and date. When available, additional information included estimated age-class, estimated sex, gear type, and gearset location. Based on methodologies in Saez et al. (2020b), NMFS/NOAA staff categorized cases as "confirmed" or "unconfirmed" entanglements. There were 217 confirmed entanglement cases involving humpback whales from 1982 to 2019. Only 64 of the confirmed humpback whale entanglements cases included fluke images. Eighty-three percent (n = 53) of those cases included images of sufficient quality to be matched to cataloged individuals (Cheeseman et al., 2021) or were of adequate quality to be added to Cascadia Research Collective's (CRC) humpback whale photo-ID catalog to elicit resights. Images were deemed adequate quality if they received a score of 1–3 for "proportion of fluke visible," "fluke angle," "photographer lateral angle," and "focus/sharpness" based methods from Calambokidis et al. (1997). Our study used CRC's humpback whale photo-ID catalog and sightings databases collected from 1985 through 2018. The CRC databases also included data submitted to Happywhale from the CA-OR-WA region. These expansive datasets provided the maximum opportunity to resight entangled whales and individuals in our control groups. The CRC and Happywhale catalogs elicit sightings from along the entire CA-OR-WA region as well as a North Pacific-wide reference catalog of 18,844 individuals; therefore, they cast a wide net to resight whales that may have relocated outside of the target study area.

Control Groups

We created a control group for each case based on initial entanglement reports' date and location to provide resighting rate comparisons for each entangled whale. Within the CRC sightings database, the CA-OR-WA region was divided into subregions by latitudes (**Figure 1**). We assigned each entanglement report case to a subregion (E.R. region) by location. Then, we selected control groups based on sighting records (from the CRC sightings database) of individuals documented in those subregions within 3 months of each initial report. We chose a time span of 45 days before the initial entanglement report's date until 45 days after the report. This period's length ensured a minimum control group size of 15 individuals since some of the subregions had very few sightings around the time of the entanglement reports. No control groups or entanglement cases were binned because they occurred in the same subregion around the same date; therefore, individuals could belong to multiple control groups. We also gathered demographic data and sighting histories on all the unique individuals (n = 2,296) that formed the control groups.

Resight Analysis

To put the resighting rate of individuals into a historical context, we used sighting data from 1985 through 2018 to ascertain the year and location of all sightings before and after the entanglement year for each entangled whale. To ensure each whale had the opportunity to be resighted a year after its entanglement, we excluded cases that occurred after 2017. The research effort in 2018 was higher than usual, with coordinated research effort and transect lines covering the entire CA-OR-WA region as well as across the borders into Canada and Mexico. This higher effort documented a larger proportion of the population than during an average year (Calambokidis and Barlow, 2020).

We used sighting-history data to determine the number of years individuals were documented, the number of years individuals were known to be alive, and the number of years an individual could have been seen after the entanglement year through 2018. We termed the number of years an individual could have been seen after the entanglement year as "opportunity years." Next, we assigned a "1" to each opportunity year when the whale was documented between January 1 and December 31 of that year and assigned a "0" if the whale was not documented. We calculated the number of years a whale was known to be alive by assigning a "1" to the most recent year the animal was documented and then assigning a "1" to every year between that year and the entanglement year.

We used two methods to determine the resighting rate of entangled whales and their control groups. For the first method, we used paired t-tests to compare the percentage of years an entangled whale was seen to the mean percentage of years seen for its control group (determined as the percentage of opportunity years each control whale was seen, averaged for each control group). We repeated this process to compare the percentage of years known to be alive for the entangled whales and their control groups. This method accounted for any influence caused by the initial report's location, year, or time of year.

For the second method, we pooled all of the sightings data for entangled whales into one group and the control whales into one large pooled control group. By pooling the data, we acknowledge that individuals can have varying opportunity years available to be seen depending on how long ago the entanglement occurred; therefore, it should not be weighted equally. We summed the number of years all individuals were resighted for each pooled group and divided it by the sum of the opportunity years to provide a percentage of years the pooled group was resighted. Then, we repeated the process for the number of years all individuals were known to be alive. We used the Chi-square goodness of fit test to compare the pooled entanglement group to the pooled control group to ascertain if the entangled whales were less likely to be resighted and were known to be alive for fewer years than the control group.

Demography

The sightings data provided information on the sex, age, and age-class of individual whales. Sex was previously determined where possible through genetic analysis of a biopsy or skin sample collected during research or rescue efforts (Palsbøll et al., 1992; Bérubé and Palsbøll, 1996a,b), presence of hemispherical lobe (Glockner, 1983), or based on a whale's behavioral role (Steiger and Calambokidis, 2000). An individual was classified as a mature female after being documented with a dependent calf and determined to be its mother (Steiger and Calambokidis, 2000). The exact age of individuals was known for individuals first identified during their calf year. For individuals not documented during their calf year, we used the length of their sighting history to estimate their age-class.

In our study, we used four age-class categories: calf, juvenile, "likely juvenile," and mature. Calves were individuals in their

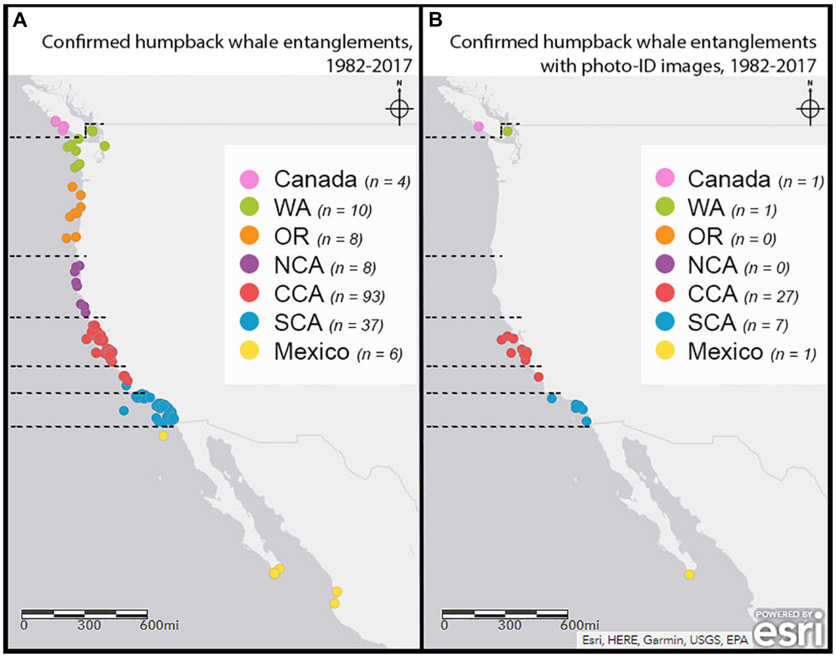


FIGURE 1 | Location of the initial report of confirmed humpback entanglement cases. **(A)** The distribution of initial reports of confirmed entangled humpback whale cases along the CA-OR-WA region from 1982 to 2017. The data included cases from Canada and Mexico that involved gear from CA, OR, or WA, or gear was removed in US waters. **(B)** The distribution of initial reports of confirmed entangled humpback whale cases with proper photo-ID images along the CA-OR-WA region from 1982 to 2017. Saez et al. (2020a,b) assigned each case to a "Report Region" (designated by color) based on the initial report location (Canada, WA-Washington, OR-Oregon, NCA-northern California, CCA- central California, SCA- southern California, and Mexico). The dashed horizontal lines designated the subregions used in this study to determine the entanglement reporting region (E.R. region). We used the E.R. regions to select individuals for the control groups and determine if the initial report occurred while the entangled whales were out of their preferred habitat.

first year of life and were typically still dependent on their mothers. Juveniles were individuals with known ages at least a year old but under 5 years old (Barco et al., 2002; Robbins, 2007), since the minimum age at first calving is 5 years old (Clapham, 1992). Individuals at least 5 years old were considered mature (Chittleborough, 1965; Clapham, 1992; Steiger and Calambokidis, 2000; Barco et al., 2002; Robbins, 2007). In this study, the term "mature" only signifies the individual has reached the minimum age of first calving and does not assume the individual is sexually mature or is a reproductive member of the population (Gabriele et al., 2007). For whales not documented as calves, we assumed they were at least 1 year old at first sighting (Steiger and Calambokidis, 2000; Barco et al., 2002; Robbins, 2007). Therefore, mature whales included whales of unknown ages with a sighting history that spanned at least 4 years pre-entanglement (Steiger and Calambokidis, 2000; Barco et al., 2002; Robbins, 2007). When whales had sighting histories that spanned less than 4 years, it was unknown if they were actually young or possibly mature whales that had not been detected previously (Barco et al., 2002). However, based on a population with a high percent of known-aged whales due to a large majority of the population documented each year, Robbins (2007) found that, on average, animals with shorter sighting histories have similar characteristics to known juveniles. Therefore, we considered individuals with sighting histories that spanned less than 4 years pre-entanglement to be "likely juveniles" (Robbins, 2007). It is likely that a percentage of the individuals classified as "likely juveniles" are mature individuals who avoided detection for a number of years. We determined the percentage of control whales which were classified as "likely juvenile" but had a calf before their sighting history extended beyond 4 years. This calculation allowed us to better understand the sighting history length as a proxy for age-class when exploring if age-class was a confounding factor in the resighting rate of entangled whales.

We used the Chi-square goodness of fit test to compare the proportion of "likely juveniles" to mature whales within the entangled whales to the proportion within the control whales. We also used the Chi-square goodness of fit test to determine the resighting rate of "likely juveniles" compared to mature whales, regardless of if they were entangled or not. Humpback whale calves and juveniles have lower resightability (lower probability for photographic identification) (Carlson et al., 1990; Blackmer et al., 2000; Robbins, 2007). Therefore, to ensure the lower resightability of juveniles was not a confounding factor in this study, we used the Chi-square goodness of fit test to compare the resighting rate of entangled "likely juveniles" to control whales classified as "likely juveniles."

Distinct Population Segments

The sightings data also provided information on the distinct population segments (DPS) of individual whales. Under the Endangered Species Act, populations of vertebrate species that are deemed discrete and significant are recognized as separate DPS's, even if not reproductively isolated from each other. Humpback whales that forage off the U.S. West Coast belong to the Central American (endangered), Mexican (threatened), or Hawaiian (not at risk) DPS. We assigned DPS to individuals if they were photographed on the Central American, Mexican, or Hawaiian breeding grounds. If an individual was documented on both the Mexican and Central American breeding grounds, the individual was assigned to the Central American DPS. It is not uncommon for whales to be documented off Mexico during their migration to and from Central America. DPS designation allowed us to determine the proportion of entangled whales that belong to each DPS and where those initial entanglement reports occurred.

Entanglement Report Location Analysis

In addition to determining the presence or absence of whales pre- and post-entanglement, we also compared sighting histories to determine if each whale was seen within, outside, or both within and outside its E.R. region. We carried out the location analysis for sightings that occurred before, during, or after its entanglement year. This comparison determined if individuals were seen "out-of-habitat" within the CA-OR-WA region while entangled. We also compared the entangled whales to other feeding ground populations to ensure they did not immigrate from or emigrate to other feeding grounds.

Additionally, we used multiple Fisher's exact tests to determine if the percentage of whales seen within or outside the E.R. region differed significantly from the control whales' sighting history. Therefore, determining if control groups based on the initial report's location were representative of the entangled whale's likelihood of being seen.

Gearset Location Analysis

Saez et al. (2020a,b) provided published data regarding the type of entangling material and the gearset location when known. Sixty-five confirmed humpback whale entanglement cases from 1982 to 2017 included a known fishery and gearset location (Saez et al., 2020a,b). Saez et al. (2020a,b) classified gearset location as Washington, Oregon/Washington, Oregon, Northern California, Central California, Southern California, or "California but an unknown region." For this analysis, the "Report Region" published in Saez et al. (2020a,b) was used to determine if the initial report occurred in the same subregion as the gearset location. This process ensured consistency between terms applied to the initial report location and the location of the gearset. We used Fisher's exact tests to determine if the initial report location compared to the gearset location affected entangled whales' resighting rate or the number of years known to be alive.

Serious Injury and Mortality Score

We used published data from Southwest Fisheries Science Center's (SWFSC) "Sources of human-related injury and mortality for U.S. Pacific west coast marine mammal stock assessments" from 2013 through 2020 (Carretta et al., 2013, 2015, 2017, 2018, 2019, 2020) to compare their estimated levels of mortality caused by entanglements to the percentage of individuals from our study resighted after their entanglement year. Carretta et al.(2013,2015,2017,2018,2019,2020) calculated estimated mortality levels using a system of Serious Injury and Mortality (SI/M) scoring assigned to humpback whales involved in entanglements. The SI/M scores included non-serious injury (NSI), serious injury (SI), and serious injury prorated by 0.75 (prorated-SI) (see Carretta et al., 2020 for methodologies). Individuals scored as NSI were estimated to survive their interaction, while individuals scored as SI were estimated to die. Seventy-five percent of individuals classified as prorated-SI were estimated to die, while the remaining quarter was estimated to survive. Carretta et al.(2013,2015,2017,2018,2019,2020) gave an initial SI/M score based on the entanglement configuration and health of the whale. Then, they assigned a second final score based on resights or human intervention to remove the life-threatening gear to improve the whale's chances of survival. For our analysis, we grouped entangled whales by their initial SI/M score, and compared their expected survival rate to the percentage of the group resighted post- entanglement year. The process was repeated using the final SI/M score and then compared to the average percentage of control groups resighted. We applied two exact binomial tests to determine if the initial and final NSI score deviated from the expected value based on the average percentage of control groups resighted.

RESULTS AND DISCUSSION

Entanglement Reports and Cases With Identifiable Individuals

Between 1982 and 2017, a total of 37 confirmed humpback whale entanglement cases included photo-ID images of sufficient quality for use in the resight analysis. The 37 cases represented 37 unique humpback whales, and none of those individuals were known to be involved in more than one confirmed case during the study period. The confirmed cases represented a combination of life-threatening and non-life-threatening entanglements with different likelihoods of survival. Thirty-five percent (n = 13) of those individuals were resighted at least 1 year after their initial entanglement report and survived the immediate effect of their entanglement.

The low number (n = 37) of confirmed humpback whale entanglement cases with proper photo-ID within the CA-OR-WA region was not surprising. Although NOAA began logging entanglement reports within the CA-OR-WA region in 1982, a formal regional response network was not formed until the 2000s. More recently, the lower cost and increased use of digital cameras, underwater video, and camera phones have increased the ability of reporting parties and trained responders to obtain higher quality photo-ID images.

All of the resighted humpback whales were gear-free at the year mark; therefore, there were no multi-year entanglements. The lack of multi-year entanglements was contrary to findings in other species, such as gray whales and North Atlantic right whales, which can live for over a year with chronic entanglements (Moore et al., 2006, 2013; Cassoff et al., 2011; Carretta et al., 2019). The lack of humpback whales with chronic, multi-year entanglements suggests that they cannot compensate for the energetic cost of entanglements long-term (van der Hoop et al., 2016), resulting in death, not that humpback whales are better suited than other species to free themselves from entanglements.

Of the 37 entanglement cases with identifiable humpback whales, only one individual was documented to have died. She was 19 years old during the initial entanglement report in early August 2016. However, her health declined (extensive cyamid coverage) between her last sighting with the original entanglement configuration and her sighting a month later after the removal of some gear. She appeared to be gear-free by late September, and although her heavy cyamid coverage decreased, she remained emaciated with pale skin until her last sighting in December 2016. It is unknown if she migrated to the breeding grounds. However, she was seen in early April 2017 in a similar body condition with pale skin and shoulder blades visible, suggesting her health did not significantly improve over the 3 months. Nine days later, her carcass was floating near-shore in April of 2017. Unfortunately, local agencies were not able to perform a necropsy, so researchers could not determine the extent of her entanglement injuries or if the cause of death was related to the entanglement or not. This case is the only time in 40 years in the CA-OR-WA region that the carcass of a known humpback whale, let alone one with a sighting history since it was a calf, was first seen alive while entangled and found dead 6 months later.

This single case of a cataloged individual documented entangled and then dying 6 months later highlights the rarity of those data. The increased use of digital cameras and the advent of automated matching in systems like Happywhale are improving the ability to match and verify essential cases in real time. Therefore, population biologists and response networks can relay the necessary information (life history, entanglement configuration, known body condition/injuries prior to death) to stranding networks in order to recover and perform a focused necropsy on whales known to be recently entangled. These processes will expedite future information exchange among entanglement response networks, stranding networks, and population biologists to guarantee that all the necessary data are collected when rare opportunities arise.

Resighting Rates of Entangled Whales Compared to Control Groups

During this study, post-entanglement resighting rates were lower for whales documented while entangled than for control whales. The percentage of years whales were seen post-entanglement was more often lower for the entangled whales than their control group (paired *t*-test; $t = -3.8313$, df = 1,36, *p*-value = 0.0005); therefore, they may have a higher mortality rate than the overall population. However, even when we knew entangled whales were alive post-entanglement year, they were seen less often than the mean of their control group (paired *t*-test; $t = -4.008$, df = 1,36, *p*-value = 0.0003). We reached the same conclusion using the pooled data. Entangled whales ($n = 37$) were resighted less often (33.9%) and known to be alive for a smaller percentage of years (55.4%) post-entanglement than the pooled control whales ($n = 8256$; resighted: 51.1%; known to be alive: 66.2%) (resighted: $\chi^2 = 14.3440$, df = 1, $p = 0.0002$; known to be alive; $\chi^2 = 6.3344$, df = 1, $p = 0.0118$; **Table 1**). Therefore, post-entanglement year, the entangled whales have a lower resightability even when they are known to be alive (61.2%) compared to the control groups (77.2%) ($\chi^2 = 9.7222$, df = 1, $p = 0.0018$).

A possible explanation for the lower resightability of whales that survive entanglements could be injuries sustained during the entanglement that reduce the ability to identify them, such as scars causing drastic pigmentation changes or chronic injuries that reduce the frequency or ability to fluke. However, photo-ID images were matched with highly accurate automated image recognition that appears to be robust to changes in pigmentation, pattern, and shape of flukes (Cheeseman et al., 2021). A second possibility is that entanglements disproportionately affect juveniles, who also tend to have higher mortality and lower resightablility rates than sexually mature individuals (Rosenbaum et al., 2002; Robbins, 2007).

Demography of Entangled Humpback Whales

Known Sex and Age

Unfortunately, only 15% ($n = 8$) of the cataloged individuals involved in confirmed entanglement cases could be assigned a sex. Seven were females, and one was male. Due to the small sample size, we could not determine if sex influenced the probability of becoming entangled. The sighting history of individuals first documented as calves provided their exact age when they became entangled. Known-aged individuals represented 13.5% ($n = 5$) of the entanglement cases with usable ID images. One whale was entangled during its calf year while the others were 3, 10, 11, and 19-years-old during their entanglement. The small sample size of known-aged whales from each age-class did not provide enough information to determine if entangled whales' ages or known age-classes were confounding factors in the resighting or resightability of entangled whales.

Alternatively, without photo-IDs and corresponding life history data, an entangled whale's estimated sex, age, or life history state is commonly based on visual assessments. Twenty-nine percent ($n = 64$) of the confirmed humpback whale entanglement reports ($n = 217$) included age estimates based on visual assessments. However, they lacked consistency in terms used, and age-classes designations overlapped in the range of ages they included. Of the cases with visual assessment data, 40.1% ($n = 26$) of whales were classified as adult/mature, 53.2% ($n = 34$) were assigned to non-exclusive categories of yearling, juvenile, subadult, or 3–5 years old, and 6.2% ($n = 4$) of whales were classified as calves, Additionally, the estimated length given to individuals with age-class information greatly overlapped. Individuals visually classified as adult/mature ranged in estimated lengths from 9 to 15.2 m. While the estimated length of individuals estimated to be yearlings/juveniles/subadults ranged from 6.1 to 12.2 m, and those estimated to be calves ranging from 6.1 to 7.6 m.

The terms used for age-class estimates (yearling, juvenile, subadult, 3–5-year-old, adult, and mature) prevented the comparison between the visually estimated data and our use of

TABLE 1 | Comparison of resights of the entangled whale to control groups using pooled data.

	Number of years whales were seen after ent. year	Number of years whales were not seen after ent. year	Number of years whales were known to be alive after ent. year	Number of years whales were not known to be alive after ent. year
Entangled whales pooled data	41 (34%)	80 (66%)	67 (55%)	54 (45%)
Control group pooled data	10,310 (51%)	9,847 (49%)	13,350 (66%)	6,807 (34%)

The number of years whales were seen after the entanglement year represents each year a whale was resighted. The number of years whales were known to be alive includes every year between the entanglement year and the most recent sighting of that individual, regardless of whether it was seen or not during that period. The results of this study show that entangled whales had a lower resighting rate (χ^2 = 14.34, df = 1, p = 0.0002) and were seen less often than expected (χ^2 = 6.33, df = 1, p = 0.0118) compared to the control whales post-entanglement.

sighting history as a proxy for age-class. Regardless, humpback whales exhibit a range of sizes in which length does not always indicate age (Stevick, 1999); therefore, length is not the best indicator of age-class. However, knowing the life history state of entangled whales is important since demographics have different natural mortality rates and risk levels for anthropogenic threats (Rosenbaum et al., 2002; Robbins, 2007; Robbins, 2009).

Sighting History Length as an Age-Class Proxy

In our study, sighting history length as a proxy for age-class suggests that, on average, entangled whales were likely younger than the control whales. The majority (67.6%, n = 25) of entangled whales had sighting histories shorter than 3 years prior to the entanglement year; therefore, classified as "likely juveniles" (**Table 2**). In contrast, only 38.3% (n = 3,165) of the control whales had shorter sighting histories. Our results suggest that the proportion of individuals with shorter sightings histories was disproportionately higher for entangled whales compared to the control group (χ^2 = 13.30, df = 1, p = 0.0003, **Table 2**).

Despite the majority of entangled whales within the CA-OR-WA region being categorized as "likely juveniles," the lower resightability of juveniles did not account for the different resighting rates found between entangled and control whales. Regardless of whether they were entangled or not, of the individuals in this study categorized as "likely juveniles" (n = 3,190), a smaller percentage were resighted (58%, n = 1,846) compared to the percentage mature individuals resighted (78%, n = 3,967; Chi-square, χ^2 = 369.73, df = 1, p-value < 2.2e-16). Even so, the comparison within "likely juveniles" (n = 3,190) showed that a smaller percentage (28%, n = 7) of entangled whales classified as "likely juveniles" (n = 25) were resighted than the percentage resighted (58%, n = 1,839) of the control whales classified as "likely juveniles" (n = 3,165; Chi-square, χ^2 = 9.22, df = 1, p-value = 0.0024, **Figure 2**).

The stark reversal of the proportion of "likely juveniles" to mature individuals when comparing the entangled whales to the control groups was initially surprising. However, the high proportion of entangled whales that fell within the category of "likely juveniles" agrees with other studies that found juveniles have a higher risk of entanglement than mature animals (Robbins, 2009). Similarly, our results show "likely juveniles" (both entangled and control) were less likely to be resighted than mature whales – which is in agreement with the lower juvenile survival rate suggested by other studies (Rosenbaum et al., 2002; Robbins, 2007). Juveniles may also have lower detection rates due to changes in their fluke patterns (Carlson et al., 1990; Blackmer et al., 2000), the behavioral tendency to fluke less often, and higher emigration rates from their maternal feeding ground (Robbins, 2007). However, the characteristic of the juvenile demographic did not fully account for the difference in resighting rates between entangled whales and control whales since entangled whales classified as "likely juveniles" were resighted less often than the control whales classified as "likely juveniles." Therefore, our study suggests that entanglements lower the resighting rate of humpback whales, and disproportionately affects "likely juveniles."

Compared to larger mature whales, the smaller body size of juveniles with fewer energy stores decreases the likelihood of individuals being able to compensate for the loss of feeding opportunities, the cost of drag, or fighting an infection caused by the entangling gear (Cassoff et al., 2011; Moore and van der Hoop, 2012; van der Hoop et al., 2016; Pettis et al., 2017; Tulloch et al., 2020). Additionally, most gear modifications to reduce entanglement duration or severity tend to be focused on the size and strength of mature right whales, leaving the duration or severity for younger individuals or smaller species unchanged by newer regulations (Pace et al., 2014; Knowlton et al., 2016).

Although using known ages based on birth year is a preferred, more accurate method, those data are rare for most baleen whale populations. Of the 8,256 control whales in our study, 755 were females that had been documented with a calf at least once during their sighting history. Of those individuals, 728 were properly classified as mature individuals based on their longer sighting history length prior to the entanglement year of the control group, or were classified as "likely juveniles" during the entanglement year and had their calves in later years. Only 4% (n = 27) of the 755 individuals were misclassified as "likely juveniles" since they had a calf prior to their sighting history reaching 4 years. We recognize that some mature individuals who previously went undetected fall under the "likely juvenile" category. Still, this was a critical step in determining the probability of resighting entangled whales, and further application of these results should acknowledge that this method provided estimated age-classes.

Distinct Population Segments

External to the resight analysis, we expanded the number of entangled humpback whale cases with proper ID images

TABLE 2 | Sighting history as a proxy for age-class.

	"Likely juvenile"		Mature
	Not seen prior to entanglement year	**Seen 1–3 years prior to entanglement year**	**Seen four or more years prior to entanglement**
Entangled whales (*n* = 37)	20 (54%)*	5 (14%)*	12 (32%)$^{+}$
Control group (*n* = 8,256)	1,942 (24%)	1,223 (15%)	5,091 (62%)

**Including one known aged individual* $^{+}$*including three known aged individuals.*
Whales that were first documented less than 4 years prior to their entanglement year, or first seen during their entanglement year, were classified as "likely juveniles." Sexually mature individuals were first documented at least 4 years prior to their entanglement year, ensuring they were at least a minimum of 5-years old (minimum age of calving) during the entanglement year (Clapham, 1992; Robbins, 2007). A higher percentage of entangled whales (68%) had shorter sighting histories compared to the control group (38%, $\chi^2 = 13.30$, *df = 1, p = 0.0003).*

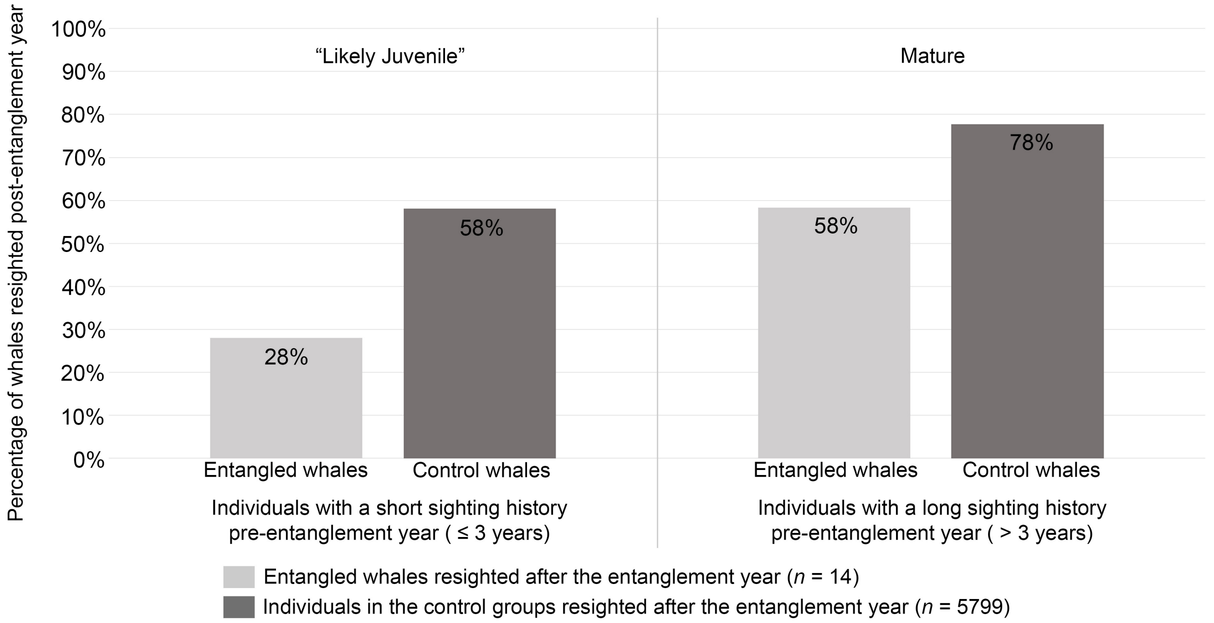


FIGURE 2 | Percentage of whales seen post-entanglement grouped by their sighting history length as a proxy for age-class. Fewer "likely juveniles" were resighted compared to mature individuals, regardless of being entangled or not ($\chi^2 = 369.73$, df = 1, p-value < 2.2e-16). However, the sighting history length did not account for the difference in resighting rates, since a smaller percentage of both "likely juvenile" (28%, *n* = 7) and mature entangled whales (58%, *n* = 7) were resighted compared to their counterparts in the control group ("likely juvenile": 58%, *n* = 1,839; mature: 78%, *n* = 3960).

from 37 to 54 by including cases from 2018 to 2019. Thirty percent (*n* = 16) of entangled humpback whales with proper ID images (*n* = 54) were documented during the breeding season and assigned to a DPS. Based on photo IDs captured in the breeding grounds, 37.5% (*n* = 6) of the entangled whales matched to the Central American DPS, 62.5% (*n* = 10) to the Mexican DPS, and none to the Hawaiian DPS. Unfortunately, the DPS was unknown for any case initially reported in or that involved gear from Oregon or Washington. All of the initial reports with known DPS in California occurred in the central California region.

Since the majority of entangled whales with proper ID images were initially reported in California (89%, *n* = 48), a higher percentage of entangled whales tied to the threatened Mexican DPS and endangered Central American DPS was expected (Calambokidis et al., 1996, 2001, 2008, 2017). Considering only a small proportion of entanglements are reported, let alone documented well enough to determine DPS, researchers should explore alternative methods, such as scar studies, to understand how pervasive entanglements are within the endangered Central American population.

Entanglement Report Location

Seventy-seven percent of whales with pre-entanglement sighting history (*n* = 17) were seen within their E.R. region in years before their entanglement (**Figure 3**). All of the whales resighted post-entanglement were also seen within their E.R. region (**Figure 3**). Additionally, none of the entangled whales lacking post-entanglement sightings had a sighting history before, during, or after their entanglement year in a Northeast Pacific feeding ground separate from the CA-OR-WA region.

Since 100% of whales with post-entanglement resights were seen within their E.R. region, our results suggest that entangled whales do not appear to be "out-of-habitat" when initially

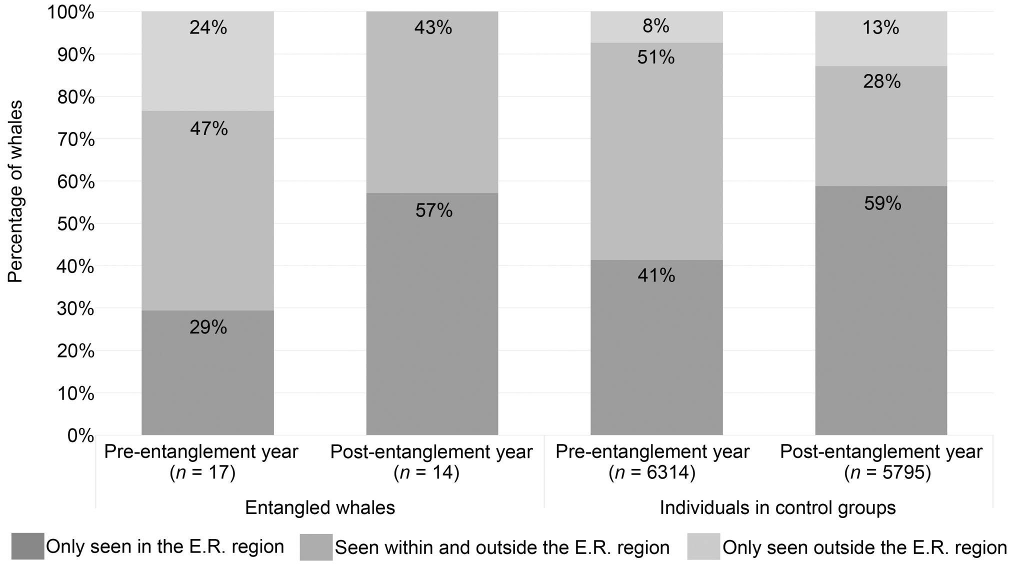


FIGURE 3 | Percentage of whales seen within and outside of the initial entanglement report region (E.R. region) pre- and post-entanglement year. All the entangled whales resighted post-entanglement were documented at least once in the same region as their initial entanglement report. Returning to the same region post-entanglement, combined with the high percentage of whales with pre-entanglement sighting history seen within the region of their entanglement, suggests that the entangled whales in this study were not "out-of-habitat" when initially reported entangled. Additionally, the percentage of entangled whales seen within, outside, or both within and outside the E.R. regions compared to the control groups did not deviate from expected values for before, during, or after the entanglement year (Fisher's Exact test two-sided, before: $p = 0.0605$; during: $p = 0.3526$; after: $p = 0.2615$), suggesting that selecting individuals based on the E. R. region proved suitable control groups.

reported entangled. Little exchange occurs between northern feeding grounds and the CA-OR-WA region, even during migration (Baker et al., 1990; Calambokidis et al., 1996, 2001, 2008). Therefore, these results suggest that the entangled whales within the CA-OR-WA region were members of this feeding group population.

When compared to the control whales, the percentage of entangled whales seen within, outside, or both within and outside their E.R. regions did not deviate from expected values for before, during, or after the entanglement year (Fisher's Exact test two-sided, before: $p = 0.0605$; during: $p = 0.3526$; after: $p = 0.2615$). Therefore, selecting control groups based on the E.R. region provided a comparable group for resight analyses and did not inflate the expected resighting rate.

Gearset Location

Fifteen entanglement cases with identified humpback whales (41%) involved gear with a known gearset state and fishery. Sixty percent ($n = 9$) of the 15 cases involved gear set in one of three subregions within California (Northern, Central, or Southern California) and the initial entanglement report occurred within the same region as the gearset location. In 20% ($n = 3$) of cases, the gearset and initial entanglement report occurred within California, but the subregion of the gearset was unknown. In the remaining 20% ($n = 3$) of cases, the gearset location was in a different state or country from the initial entanglement report.

Even though most reports occurred in the same state or region as the gearset location, inferring that the E.R. region is commonly where the whale becomes entangled is inexact. The gearset data in this study were biased toward entanglement involving weighted gear since the Dungeness crab fishery was one of the few fisheries within the CA-OR-WA region required to mark their gear systematically during our study period. Crab gear usually involves at least one weighted pot/trap that may restrict an animal's movement or prevent it from dragging the gear very far. However, after the lines part and weighted gear, such as pots/traps, fall away, entangled whales can carry the remaining gear for greater distances, such as across the state or country lines (Robbins et al., 2007; Bradford and Lyman, 2015; Saez et al., 2020b). Unfortunately, the chances of fishery identifying marks remaining on entangling gear are likely lower for gear that has been on an animal for a longer duration and exposed to more drag caused by a free-swimming entangled whale. Therefore, the longer the gear is on a whale, the greater the likelihood the gear is less recognizable (lack of fishery identifying marks, buoys, tags, or traps- representing 59% of the cases in this study); making it more challenging to confirm where, when, and in what type of gear the whale became entangled.

During this study, the difference in gearset location and the initial report location did not influence the percentage of years the whales were seen post-entanglement (Fisher's Exact test $p = 0.3293$, **Table 3**). However, the percentage of years the animals were known to be alive (50%, $n = 14$) was higher than expected for reports that occurred within the same region as the gearset location (Fisher's Exact test $p = 0.0161$, **Table 3**). Although responders removed some or all of the gear from entangled whales whose gearset was in a different state or country than

TABLE 3 | Gearset location compared to the location of the initial report.

	Number of opportunity years	Percentage of years whales were seen after entanglement year	Percentage of years whales were known to be alive after entanglement year
Reported inside the same region gear was set *(n = 9)*	28	28.6%	50.0%
Reported inside the same state gear was set, but region unknown *(n = 3)*	6	16.7%	16.7%
Reported outside of the state gear was set (*n* = 3)	7	0.0%	0.0%

Forty-one percent (n = 15) of the entanglement cases of identified humpback whales (from 1982 to 2017) involved gear linked to a state and fishery. The number of opportunity years represents the pool of years since each animals' entanglement in which those individuals could have been seen. The percentage of years whales were seen after the entanglement year represents each year a whale was resighted. The percentage of years whales were known to be alive includes every year between the entanglement year and the most recent sighting of that individual, regardless of it was seen or not during that period. Although the gearset location compared to the location of the initial report did not influence the number of years the whales were seen post-entanglement (Fisher's Exact test p = 0.3293); the number of years the animals were known to be alive was higher than expected for reports that occurred within the same region as the gearset location (Fisher's Exact test p = 0.0161).

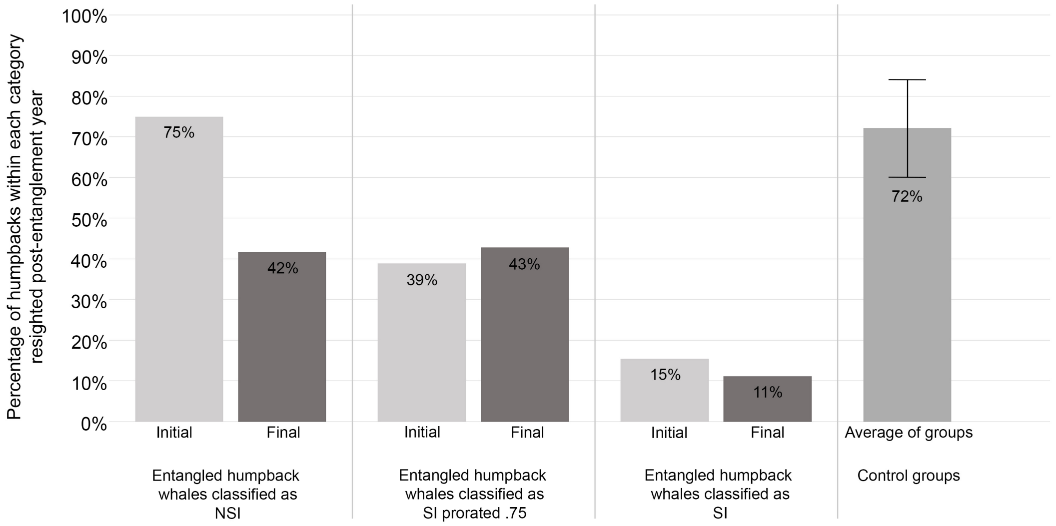


FIGURE 4 | Initial and final serious injury and mortality (SI/M) scores compared to resightings of entangled whales. Ninety-five percent (*n* = 35) of the confirmed entangled humpback whale cases used in our resight analysis were assigned initial and final SI/M scores from Southwest Fisheries Science Center. Based on the resighted individuals in our study, the adjustment from initial to final score appeared to provide survival estimates that were less in line with our resightings of individuals classified as having a non-serious injury (NSI) and a serious injury prorated by 75% (SI prorated 0.75). However, the percentage of individuals resighted post-entanglement assigned a final score of serious injury (SI) was closer to the 0% survival estimated compared to those initially assigned a SI score.

the report, none of those whales were seen post-entanglement during the study. However, none of those individuals have pre-entanglement sightings either.

Based on our results, one might infer that the sooner an animal is found entangled, the better chance it has of surviving. Although survival is likely higher for shorter duration entanglements (Robbins et al., 2015), increased distance between the initial report and gearset location does not necessarily mean increased entanglement duration. The severity and configuration of the entanglement can allow a whale whose movement is less restricted to travel great distances in a short period. In contrast, a whale whose movements are more restricted may remain in the same region it became entangled in, but as the entanglement's duration increases, the likelihood of matching the gear to that region can decrease. Additionally, the individuals resighted in this study were not all disentangled. Therefore, although the initial report occurred in the same region as the gearset location, the entangled whales may have traveled outside that region but were not documented during that time. Further analysis regarding the severity of the entanglements as well as the temporal and spatial overlap between fisheries, whale watching, and research efforts will shed more light on this result.

Serious Injury and Mortality Score Comparison

Thirty-five of the 37 cases of known individuals were scored for serious injury and mortality (SI/M) by Southwest Fisheries Science Center (SWFSC) (Carretta et al., 2013, 2015, 2017, 2018, 2019, 2020). One of the 37 cases occurred before the publications of SI/M scores related to human interactions. Another individual (initially scored as a serious injury) died a few months after its entanglement; therefore, she was not given a final SI/M score.

Based on the initial scoring, the percentage of entangled whales resighted post-entanglement year and classified as NSI (non-serious injury, 75%, *n* = 3) was comparable to the average percentage of control groups resighted (72% ± 12%, exact binomial test *p*-value = 1.0, **Figure 4**). More than 25% of entangled whales initially classified as prorated-SI were resighted (39%, *n* = 7), while only two individuals (15%) classified as SI (serious injury) were resighted (**Figure 4**).

In contrast, after the scores were adjusted, the final SI/M did not fit as well with the percentage of known individuals that were resighted. The percentage of entangled whales resighted post-entanglement year and classified as NSI (42%, *n* = 5) was much lower than the expected average percentage of control groups resighted (72% ± 12%, *n* = 35, exact binomial test *p*-value = 0.0461). A higher proportion (43%, *n* = 6) of individuals classified as prorated-SI were resighted compared to the initial score (39%) and the expected estimate of 25%. However, fewer (11%, *n* = 1) of the whales with a final SI classification were resighted than those initially scored SI, therefore closer to the estimated 0% survival rate (**Figure 4**).

Seventy-five percent (*n* = 3) of the cases adjusted from SI to NSI were not resighted. Seventy-five percent (*n* = 3) of the cases adjusted from prorated-SI to NSI were not resighted either. The six cases of whales not resighted but adjusted to NSI fell into the classification of mature (*n* = 1) and "likely juvenile" (*n* = 4) based on their sighting history length or were known to be a dependent calf (*n* = 1). The outcomes of these cases were split between responses in which all or some of the gear was removed through human intervention (*n* = 3) and events where the animal appeared to shed the gear without human intervention (*n* = 3).

Fourteen cases remained in the prorated-SI category after the final scoring. Six of those were mature during their entanglement, and only two of those individuals (33%) were resighted. With the small sample size, the resighting rate was as close to the 25% predicted survival rate as the data would allow. The other eight cases classified as prorated-SI were "likely juveniles," with half of them resighted after their entanglement year. Many of the cases of "likely juveniles" lacked documentation to understand the entanglement configuration fully. However, based on the information collected, the entanglement had the potential to become life-threatening.

Based on the resighting rate of entangled whales in our study, the final SI/M score was slightly more accurate for individuals scored as SI, but the adjustments made to NSI and prorated-SI created less accurate estimates. Our results suggest that the age-class of individuals may partially explain the differences between the expected survival and the actual resights. Although the prorated-SI scoring of mature humpback whales along the West Coast appeared to be accurate for our data subset, our results highlight the possibility that entanglements categorized as non-life-threatening (NSI) for mature whales may have a more significant negative impact on younger whales. Unfortunately, information regarding the whale's health was not systematically collected during responses or consistently captured by images submitted by reporting parties. After removing life-threatening gear, the survival prognosis for an animal with poor body condition is not the same as a healthy whale. Additionally, the timing of the entanglement (early vs. late in the year) may also influence the probability of a whale recovering from its entanglement prior to the end of the feeding season. A better means to obtain documentation is needed to understand the entanglement configuration fully, determine if the entanglement is life-threatening (SI), verify the removal of all the life-threatening gear, and capture the overall health/body condition and wound profiles. The obtainment of proper documentation requires increased effort to enhance trained responders' ability to mobilize and respond to entanglements. Additionally, systematic data collection during the response combined with long-term population monitoring provides better data to create a more accurate survival estimate while considering environmental and biological factors.

CONCLUSION

This study provides a better understanding of the effects of entanglements on the humpback whale population within the CA-OR-WA region.

We acknowledge that the subset of entangled whales in this study represents individuals whose health and entanglement configuration did not prevent them from bringing their flukes above the surface of the water. Therefore, our subset of entangled whales was likely biased toward free-swimming whales, healthier individuals, or animals with less severe injuries. To remove this bias, an increased effort is needed to enable trained response teams to respond to entanglements and safely obtain underwater documentation of flukes held subsurface due to weighted gear or injuries. Additional use of dorsal fin images in regional catalogs can also increase the percentage of whales with known life history data, especially since entanglement injuries can obscure individual identity based on fluke patterns alone.

Since this study ended, new fishing gear regulations were mandated to increase linking gear to a fishery and location. In Washington, new regulations introduced line marking schemes to allow gear type determination even without the buoys and traps. The California Department of Fish and Wildlife mandated new buoy markings and tags across multiple fisheries. The improved gear marking schemes along the West Coast should reduce entanglement cases with unknown gear and provide better information to target regions or types of fishing gear that entangle large whales.

Our results suggest that entangled whales have a lower resighting rate and are known to be alive for fewer years post-entanglement. Although the adverse effects of entanglements span estimated age-classes, it appears that individuals with shorter sighting histories pre-entanglement (therefore likely younger) are at a higher risk of becoming entangled. Further efforts to collect long-term population data and increase the capacity for response teams to respond and collect data during entanglements will provide a better understanding of this disproportionality. However, until further data are available, gear modifications and management should consider that most humpback whales becoming entangled may not be as large or strong as mature individuals. Since the entangled whales reported within the CA-OR-WA region were not from northern feeding ground populations, this result highlights the high number of individuals from the threatened (Mexico) and endangered

(Central American) DPS becoming entangled within the CA-OR-WA region. Therefore, the CA-OR-WA region should have the highest urgency within the US to collect proper documentation during humpback whale entanglements to determine how the whales interacted with the gear and what gear causes individuals to have the lowest resighting rate. The collection of these data during responses will determine where mitigation efforts and gear modifications should be focused and implemented to reduce the rate of baleen whale bycatch.

DATA AVAILABILITY STATEMENT

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

ETHICS STATEMENT

Ethical review and approval was not required for the animal study since NOAA provided the data on large whale entanglements used in this study.

AUTHOR CONTRIBUTIONS

JT: conceptualization, formal analysis, investigation, methodology, visualization, writing – original draft, writing – review and editing. ED and KF: data curation, investigation, writing – review and editing. TC: data curation, investigation, software, writing – review and editing. JC: conceptualization, data curation, formal analysis, funding acquisition, investigation, methodology, project administration, supervision, validation, writing – review and editing. All authors contributed to the article and approved the submitted version.

FUNDING

This work was funded by NOAA as part of several grants/contracts, including a Section 6 grant through the Washington Department of Fish and Wildlife.

ACKNOWLEDGMENTS

We would like to thank other members of the West Coast Large Whale Entanglement Network for their efforts to collect data during responses, the vessels that reported or stood by entangled whales, and NOAA for sharing the case information. We would also like to thank all the contributors to the CRC long-term humpback study and Happywhale catalog, especially the Aquarium of the Pacific, Channel Islands Naturalist Corps, Marine Life Studies, Monterey Bay Whale Watch, Oregon State University, and SeaLife Response Rehabilitation and Research.

REFERENCES

Baker, C. S., Palumbi, S. R., Lambertsen, R. H., Weinrich, M. T., Calambokidis, J., and O'Brien, S. J. (1990). Influence of seasonal migration on geographic distribution of mitochondrial DNA haplotypes in humpback whales. *Nature* 344, 238–240. doi: 10.1038/344238a0

Barco, S. G., Mclellan, W. A., Allen, J. M., Asmutis-Silvia, R. A., Meagher, E. M., Pabst, D. A., et al. (2002). Population identity of humpback whales (*Megaptera novaeangliae*) in the waters of the US mid-Atlantic states. *J. Cetacean Res. Manag.* 4, 135–141.

Bérubé, M., and Palsbøll, P. (1996a). Identification of sex in cetaceans by multiplexing with three ZFX and ZFY specific primers: Erratum. *Mol. Ecol.* 5:602. doi: 10.1111/j.1365-294X.1996.tb00355.x

Bérubé, M., and Palsbøll, P. (1996b). Identification of sex in cetaceans by multiplexing with three ZFX and ZFY specific primers. *Mol. Ecol.* 5, 283–287. doi: 10.1046/j.1365-294X.1996.00072.x

Blackmer, A. L., Anderson, S. K., and Weinrich, M. T. (2000). Temporal variability in features used to photo-identify humpback whales (*Megaptera novaeangliae*). *Mar. Mamm. Sci.* 16, 338–354.

Bradford, A. L., and Lyman, E. (2015). *Injury Determinations for Humpback Whales and Other Cetaceans Reported to NOAA Response Networks in the Hawaiian Islands During 2007-2012. U.S. Department of Commerce, NOAA Technical Memoradum, NOAA-TM-NMFS-PIFSC45*. Honolulu, HI: Pacific Islands Fisheries Science Center, doi: 10.7289/V5TX3CB1

Calambokidis, J., and Barlow, J. (2020). *Updated Abundance Estimates for Blue and Humpback Whales Along the U.S. West Coast using Data Through 2018. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-SWFSC-634*. Washington, DC: National Oceanic and Atmospheric Administration.

Calambokidis, J., Barlow, J., Flynn, K., Dobson, E., and Steiger, G. H. (2017). *Update on Abundance, Trends, and Migrations of Humpback Whales along the U.S. West Coast. International Whaling Commission, SC/A17/NP/13*. Cambridge: International Whaling Commission, 1–18.

Calambokidis, J., Falcone, E. A., Quinn, T. J., Burdin, A. M., Clapham, P. J., Ford, J. K. B., et al. (2008). *SPLASH: Structure of Populations, Levels of Abundance and Status of Humpback Whales in the North Pacific. Final Report to Southwest Fisheries Science Center, Contract AB133F-03-RP-00078*. Washington, DC: Cascadia Research.

Calambokidis, J., Steiger, G. H., Evenson, J. R., Flynn, K. R., Balcomb, K. C., Claridge, D. E., et al. (1996). Interchange and isolation of humpback whales off California and other North Pacific feeding grounds. *Mar. Mamm. Sci.* 12, 215–226. doi: 10.1111/j.1748-7692.1996.tb00572.x

Calambokidis, J., Steiger, G. H., Straley, J. M., Herman, L. M., Cerchio, S., Salden, D. R., et al. (2001). Movements and population structure of humpback whales in the North Pacific. *Mar. Mamm. Sci.* 17, 769–794. doi: 10.1111/j.1748-7692.2001.tb01298.x

Calambokidis, J., Steiger, G. H., Straley, J. M., Quinn, T. J. II, Herman, L. M., Cerchio, S., et al. (1997). *Abundance and Population Structure of Humpback Whales in the North Pacific Basin. Technical Report to the Southwest Fisheries Science Center #50ABNF500113*. Washington, DC: Cascadia Research.

Carlson, C. A., Mayo, C. A., and Whitehead, H. (1990). *Changes in the Vental Fluke Pattern of the Humpback Whale (Megaptera novaeangliae), and its Effect on Matching; Evaluation of its Significane to Photo-Identification Research. International Whaling Commission, SC/A88/ID35*. Cambridge: International Whaling Commission, 105–112.

Carretta, J. V., Delean, B., Helker, V., Muto, M. M., Greenman, J., Wilkinson, K., et al. (2017). *Sources of Human-Related Injury and Mortality for U.S. Pacific West Coast Marine Mammal Stock Assessments, 2011-2015. U.S. Department of Commerce, NOAA Technical Memoradum, NOAA-TM-NMFS-SWFSC-579*. Washington, DC: National Oceanic and Atmospheric Administration, 129.

Carretta, J. V., Delean, B., Helker, V., Muto, M. M., Greenman, J., Wilkinson, K., et al. (2020). *Sources of Human-Related Injury and Mortality for U.S. Pacific West Coast Marine Mammal Stock Assessments, 2014-2018. U.S. Department of Commerce, NOAA Technical Memoradum, NOAA-TM-NMFS-SWFSC-631*. Washington, DC: National Oceanic and Atmospheric Administration, 147.

Carretta, J. V., Helker, V., Muto, M. M., Greenman, J., Wilkinson, K., Lawson, D., et al. (2019). *Sources of Human-Related Injury and Mortality for U.S. Pacific West Coast Marine Mammal Stock Assessments, 2013-2017. U.S. Department of Commerce, NOAA Technical Memoradum, NOAA-TM-NMFS-SWFSC-616*. Washington, DC: National Oceanic and Atmospheric Administration, 150.

Carretta, J. V., Muto, M. M., Greenman, J., Wilkinson, K., Lawson, D., Viezbicke, J., et al. (2018). *Sources of Human-Related Injury and Mortality for U.S. Pacific West Coast Marine Mammal Stock Assessments, 2012-2016. U.S. Department of Commerce, NOAA Technical Memoradum, NMFS-SWFSC- 601*. Washington, DC: National Oceanic and Atmospheric Administration, 146.

Carretta, J. V., Muto, M. M., Wilkin, S., Greenman, J., Wilkinson, K., DeAngelis, M., et al. (2015). *Sources of Human-Related Injury and Mortality for U.S. Pacific West Coast Marine Mammal Stock Assessments, 2009-2013. U.S. Department of Commerce, NOAA Technical Memoradum, NOAA-TM-NMFS-SWFSC-548*. Washington, DC: National Oceanic and Atmospheric Administration, 110.

Carretta, J. V., Wilkin, S., Muto, M. M., and Wilkinson, K. (2013). *Sources of Human-Related Injury and Mortality for U.S. Pacific West Coast Marine Mammal Stock Assessments, 2007-2011. U.S. Department of Commerce, NOAA Technical Memoradum, NOAA-TM-NMFS-SWFSC-514*. Washington, DC: National Oceanic and Atmospheric Administration, 87.

Cassoff, R. M., Moore, K. M., McLellan, W. A., Barco, S. G., Rotstein, D. S., and Moore, M. J. (2011). Lethal entanglement in baleen whales. *Dis. Aquat. Organ.* 96, 175–185. doi: 10.3354/dao02385

Cheeseman, T., Southerland, K., Park, J., Olio, M., Flynn, K., Calambokidis, J., et al. (2021). Advanced image recognition: a fully automated, high-accuracy photo-identification matching system for humpback whales. *Mamm. Biol.* 2021, 1–15. doi: 10.1007/S42991-021-00180-9

Chittleborough, R. G. (1965). Dynamics of two populations of the humpback whale, *Megaptera novaeangliae* (Borowski). *Mar. Freshw. Res.* 16, 33–128. doi: 10.1071/MF9650033

Clapham, P. J. (1992). Age at attainment of sexual maturity in humpback whales, *Megaptera novaeangliae*. *Can. J. Zool.* 70, 1470–1472. doi: 10.1139/z92-202

Gabriele, C. M., Straley, J. M., and Neilson, J. L. (2007). Age at first calving of female humpback whales in southeastern Alaska. *Mar. Mamm. Sci.* 23, 226–239. doi: 10.1111/j.1748-7692.2006.00100.x

Glockner, D. A. (1983). "Determining the sex of humpback whales in their natural environment," in *Communication and Behavior of Whales*, ed. R. Payne (Boulder, CO: Westview Press), 447–464.

Guidelines for distinguishing serious from non-serious injury of marine mammals pursuant to the Marine Mammal Protection Act (2012). *Nation Marine Fisheries Service Instructions 02-238-01*. Available online at: https://www.avma.org/Advocacy/National/Federal/Documents/noaa_serious_injury_procedure.pdf (accessed September 17, 2021).

Knowlton, A. R., Robbins, J., Landry, S., McKenna, H. A., Kraus, S. D., and Werner, T. B. (2016). Effects of fishing rope strength on the severity of large whale entanglements. *Conserv. Biol.* 30, 318–328. doi: 10.1111/cobi.12590

Moore, M. J., Bogomolni, A., Bowman, R., Hamilton, P. K., Harry, C. T., Knowlton, A. R., et al. (2006). Fatally entangled right whales can die extremely slowly. *Oceans* 2006, 21–23. doi: 10.1109/OCEANS.2006.306792

Moore, M. J., and van der Hoop, J. M. (2012). The painful side of trap and fixed net fisheries: Chronic entanglement of large whales. *J. Mar. Biol.* 2012, 1–4. doi: 10.1155/2012/230653

Moore, M. J., van der Hoop, J. M., Barco, S. G., Costidis, A. M., Gulland, F. M., Jepson, P. D., et al. (2013). Criteria and case definitions for serious injury and death of pinnipeds and cetaceans caused by anthropogenic trauma. *Dis. Aquat. Organ.* 103, 229–264. doi: 10.3354/dao02566

Pace, R. M., Cole, T. V. N., and Henry, A. G. (2014). Incremental fishing gear modifications fail to significantly reduce large whale serious injury rates. *Endang. Spec. Res.* 26, 115–126. doi: 10.3354/esr00635

Palsbøll, P. J., Vader, A., Bakke, I., and El-Gewely, M. R. (1992). Determination of gender in cetaceans by the polymerase chain reaction. *Can. J. Zool.* 70, 2166–2170. doi: 10.1139/z92-292

Pettis, H. M., Rolland, R. M., Hamilton, P. K., Knowlton, A. R., Burgess, E. A., and Kraus, S. D. (2017). Body condition changes arising from natural factors and fishing gear entanglements in North Atlantic right whales *Eubalaena glacialis*. *Endang. Spec. Res.* 32, 237–249. doi: 10.3354/esr00800

Robbins, J. (2007). *Structure and Dynamics of the Gulf of Maine Humpback Whale Population*. Ph. D doctoral dissertation. Scotland: Univeristy of St. Andrews.

Robbins, J. (2009). *Scar-Based Inference into Gulf of Maine Humpback Whale Entanglement: 2003-2006. Report to the Northeast Fisheries Science Center, National Marine Fisheries Service, EA133F09CN0253*. Washington, DC: National Oceanic and Atmospheric Administration, 34.

Robbins, J. (2012). *Scar-Based Inference into Gulf of Maine Humpback Whale Entanglement: 2010. Report to the Northeast Fisheries Science Center, National Marine Fisheries Service, EA133F09CN0253*. Massachusetts, NY: Northeast Fisheries Science Center, 28.

Robbins, J., Kenney, J., Landry, S., Lyman, E., and Mattila, D. K. (2007). *Reliability of Eyewitness Reports of Large Whale Entanglement. International Whaling Commission, SC/59/BC2*. Cambridge: International Whaling Commission, 1–4.

Robbins, J., Knowlton, A. R., and Landry, S. (2015). Apparent survival of North Atlantic right whales after entanglement in fishing gear. *Biol. Conserv.* 191, 421–427. doi: 10.1016/j.biocon.2015.07.023

Robbins, J., and Mattila, D. (2004). *Estimating Humpback Whale (Megaptera novaeangliae) Entanglement Rates on the Basis of Scar Evidence. Report to the Northeast Fisheries Science Center, National Marine Fisheries Service, 43EANF030121*. Washington, DC: National Oceanic and Atmospheric Administration, 22.

Rosenbaum, H. C., Weinrich, M. T., Stoleson, S. A., Gibbs, J. P., Baker, C. S., and DeSalle, R. (2002). The effect of differential reproductive success on population genetic structure: correlations of life history with matrilines in humpback whales of the Gulf of Maine. *J. Hered.* 93, 389–399. doi: 10.1093/jhered/93.6.389

Saez, L., Lawson, D., and Deangelis, M. (2020a). *Large Whale Entanglements Off the U.S. West Coast, from 1982-2017, Appendix 2. U.S. Department of Commerce, NOAA Technical Memoradum, NMFS-OPR-63*. Washington, DC: National Oceanic and Atmospheric Administration, 11.

Saez, L., Lawson, D., and Deangelis, M. (2020b). *Large Whale Entanglements Off the U.S. West Coast, from 1982-2017. U.S. Department of Commerce, NOAA Technical Memoradum, NMFS-OPR-63*. Washington, DC: National Oceanic and Atmospheric Administration, 62.

Steiger, G. H., and Calambokidis, J. (2000). Reproductive rates of humpback whales off California. *Mar. Mamm. Sci.* 16, 220–239. doi: 10.1111/j.1748-7692.2000.tb00914.x

Stevick, P. T. (1999). Age-length relationships in humpback whales: a comparison of strandings in the western North Atlantic with commercial catches. *Mar. Mamm. Sci.* 15, 725–737. doi: 10.1111/j.1748-7692.1999.tb00839.x

Tulloch, V., Pirotta, V., Grech, A., Crocetti, S., Double, M., How, J., et al. (2020). Long-term trends and a risk analysis of cetacean entanglements and bycatch in fisheries gear in Australian waters. *Biodiv. Conserv.* 29, 251–282. doi: 10.1007/s10531-019-01881-x

van der Hoop, J. M., Corkeron, P., Kenney, J., Landry, S., Morin, D., Smith, J., et al. (2016). Drag from fishing gear entangling North Atlantic right whales. *Mar. Mamm. Sci.* 32, 619–642. doi: 10.1111/mms.12292

van der Hoop, J. M., Corkeron, P., and Moore, M. (2017). Entanglement is a costly life-history stage in large whales. *Ecol. Evol.* 7, 92–106. doi: 10.1002/ece3.2615

Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

frontiers in Marine Science

ORIGINAL RESEARCH
published: 16 February 2022
doi: 10.3389/fmars.2022.799102

Assessment of Australian Sea Lion Bycatch Mortality in a Gillnet Fishery, and Implementation and Evaluation of an Effective Mitigation Strategy

Simon D. Goldsworthy[1,2]*, Brad Page[1,3], Derek J. Hamer[1,4], Andrew D. Lowther[1,5], Peter D. Shaughnessy[1,6], Mark A. Hindell[7], Paul Burch[1], Daniel P. Costa[8], Shannon L. Fowler[8,9], Kristian Peters[1,10], Rebecca R. McIntosh[1,11], Frederic Bailleul[1,2], Alice I. Mackay[1], Roger Kirkwood[1], Dirk Holman[12] and Simon Bryars[10]

[1] South Australian Research and Development Institute, Aquatic Sciences, West Beach, SA, Australia, [2] School of Earth and Environmental Sciences, University of Adelaide, Adelaide, SA, Australia, [3] Department of Primary Industries and Regions, Urrbrae, SA, Australia, [4] DBMS Global Oceans, Hobart, TAS, Australia, [5] Norwegian Polar Institute, Tromsø, Norway, [6] South Australian Museum, North Terrace, Adelaide, SA, Australia, [7] Institute of Marine and Antarctic Studies, University of Tasmania, Hobart, TAS, Australia, [8] Ecology and Evolutionary Biology, Long Marine Laboratory, University of California, Santa Cruz, Santa Cruz, CA, United States, [9] Department of Life Sciences, University of Roehampton, London, United Kingdom, [10] Department for Environment and Water, Adelaide, SA, Australia, [11] Conservation Department, Phillip Island Nature Parks, Cowes, VIC, Australia, [12] Department for Environment and Water, Port Lincoln, SA, Australia

Edited by:
Jeff Moore,
Southwest Fisheries Science Center
(NOAA), United States

Reviewed by:
Kimberly Murray,
Northeast Fisheries Science Center
(NOAA), United States
Maritza Sepulveda,
Universidad de Valparaiso, Chile

***Correspondence:**
Simon D. Goldsworthy
simon.goldsworthy@sa.gov.au

Specialty section:
This article was submitted to
Marine Megafauna,
a section of the journal
Frontiers in Marine Science

Received: 21 October 2021
Accepted: 12 January 2022
Published: 16 February 2022

Citation:
Goldsworthy SD, Page B,
Hamer DJ, Lowther AD,
Shaughnessy PD, Hindell MA,
Burch P, Costa DP, Fowler SL,
Peters K, McIntosh RR, Bailleul F,
Mackay AI, Kirkwood R, Holman D
and Bryars S (2022) Assessment
of Australian Sea Lion Bycatch
Mortality in a Gillnet Fishery,
and Implementation and Evaluation
of an Effective Mitigation Strategy.
Front. Mar. Sci. 9:799102.
doi: 10.3389/fmars.2022.799102

Globally, the bycatch of marine mammals in fisheries represents the greatest source of human-caused mortality that threatens the sustainability of many populations and species. The Australian sea lion (*Neophoca cinerea*) is an endangered species, whose populations off South Australia (SA) have been subject to bycatch in a demersal gillnet fishery targeting sharks since the 1960s. A comprehensive assessment was undertaken of sea lion bycatch mortality that combined independent fishery observer data with species distribution models (underpinned by satellite tracking, abundance data and population modeling) to model the relationship between at-sea foraging effort and bycatch rate. Combined with the distribution of fishing effort, these models enabled the overall level of bycatch mortality to be estimated by age, sex and subpopulation, facilitating population viability analyses that indicated most subpopulations were declining, and subject to unsustainable levels of bycatch mortality. To reduce this mortality, the Australian Fisheries Management Authority implemented an Australian Sea Lion Management Strategy that included an independent observer program (ultimately 100% electronic monitoring of gillnet fishing off SA), permanent spatial gillnet closures around all sea lion breeding sites, bycatch mortality limits that triggered temporal (18 months) spatial closures when zone-specific bycatch trigger limits were reached, and incentives for gillnet fishers to switch to an alternate fishing method (longlines). The Strategy had immediate impacts on the fishery: it resulted in significant reductions in gillnet fishing effort and on the reported bycatch of sea lions. In the next decade, there was an estimated 98% reduction in sea lion bycatch mortality from gillnet interactions and an apparent stabilization of the decline in sea lion pup abundances at some impacted breeding sites. There was an almost complete transition in the fishery from gillnets to longlines, and fishing catches returned to pre-management levels. The successful implementation of management measures to mitigate sea lion bycatch mortality in the gillnet fishery off SA was rapid, science informed, adaptive, comprehensive, and backed by strong compliance and monitoring of the fishery. It provides an important case study which demonstrates how management measures can be effectively applied to mitigate bycatch mortality of marine mammals and other marine protected species.

Keywords: Australian sea lion, bycatch estimation, bycatch mitigation, bycatch management, gillnet fishery, South Australia

INTRODUCTION

Incidental catch or bycatch in fisheries presents a major threat to many marine species and is one of the most significant sources of anthropogenic mortality of marine mammals (Read et al., 2006; Sims et al., 2008; Lewison et al., 2009; Kovacs et al., 2012; Komoroske and Lewison, 2015; Peltier et al., 2016, 2021; Gray and Kennelly, 2018; Luck et al., 2020). Although marine mammals are known to interact with most fishing gear types, interactions with gillnets are particularly pervasive and a significant source of marine mammal mortality (Read et al., 2006; Reeves et al., 2013). Gillnet fishery interactions pose the principal threat to many endangered small cetaceans (Brownell et al., 2019; Gulland et al., 2020; Cisneros-Mata et al., 2021; Owen et al., 2021), and also threaten many pinniped species (Hamer et al., 2011, 2013; Kovacs et al., 2012; Cosgrove et al., 2016; Machado et al., 2016; Jounela et al., 2019; Ramos et al., 2020). Although advances have been made in mitigating interactions with many gear types (Hamilton and Baker, 2019), interactions with gillnet fisheries continue to pose some of the greatest management and conservation challenges (Read et al., 2006; Reeves et al., 2013).

Australian sea lions (*Neophoca cinerea*) (ASLs) are an endangered pinniped, endemic to southern Australia (Goldsworthy, 2015; Goldsworthy et al., 2021). They are unique among pinnipeds in having a non-annual breeding cycle (∼18 months between successive breeding seasons) in which breeding can occur at any time of the year, breeding seasons last 4 to 9+ months and occur asynchronously across the species range (i.e., neighboring colonies can breed at different times). Asynchronous breeding is thought to be maintained by extreme philopatry and population sub-structuring that effectively makes most breeding sites subpopulations (Campbell et al., 2008; Lowther et al., 2012). The non-annual and asynchronous breeding habit of ASL makes systematic monitoring of their status and trends in abundance extremely challenging (Goldsworthy et al., 2021). Despite pup births being spread over many months, pup counts are still the main method to estimate the number of pups produced in a breeding season, and for monitoring changes in the status and trends of ASL populations (Goldsworthy et al., 2021). Breeding season pup production is a common measure of abundance in seals, because pups form the only age-class that is easily identifiable (natal pelage), and most pups are ashore at the end of a breeding season.

Australian sea lions are demersal foragers that are restricted to continental shelf waters; a foraging strategy that increases their likelihood of interacting with demersal gillnet fisheries (Goldsworthy and Page, 2007). During the early to mid-2000s, multiple lines of evidence suggested that interactions between ASLs and the demersal gillnet sector of the Gillnet Hook and Trap (GHAT) fishery off South Australia (SA) were a significant threat to the species, including: anecdotal reports of bycatch from some fishers (Shaughnessy et al., 2003); high incidence of sea lions entangled in gillnet material (Page et al., 2004); substantial overlap in fishing effort and sea lion foraging distributions (Goldsworthy and Page, 2007) and population recovery at a major breeding site following cessation of the fishery in the region (Goldsworthy et al., 2014). Furthermore, a recent assessment of status and trends in abundance of the species has indicated that populations off SA have declined by 67% over the last four decades, much of which may be attributable to historic bycatch in gillnet fisheries (Goldsworthy et al., 2021).

In the late 2000s, a major study was undertaken to assess the risk to ASL subpopulations from bycatch mortality in the GHAT fishery. It: (i) assessed interaction rates through a dedicated fishery observer program; (ii) developed species distribution models using satellite tracking data; (iii) compiled spatial data on the distribution of fishing effort and (iv) developed population models from sea lion surveys and demographic data to assess the likely impact from different levels of bycatch mortality on the sustainability of ASL populations (Goldsworthy et al., 2010). The study found that observed bycatch mortality rates were highly correlated with estimated sea lion foraging effort (a proxy for sea lion density at sea), enabling the estimation of the bycatch mortality that would result from different spatial distributions and levels of fishing effort. Population viability analyses (PVA) indicated that most ASL subpopulations off SA were declining and exposed to unsustainable levels of bycatch mortality. The study concluded that further declines, subpopulation extinctions and reductions in sea lion range would be likely unless bycatch mortality was reduced (Goldsworthy et al., 2010).

In response to these findings, the Australian Fisheries Management Authority (AFMA) introduced the Australian Sea Lion Management Strategy in June 2010 (Australian Fisheries Management Authority, 2010). Most of the core management measures in the Strategy were developed and refined in stages between 2010 and 2012 and included the introduction of an independent observer program, permanent spatial gillnet closures around all sea lion breeding sites, bycatch mortality limits that triggered temporal spatial closures, and incentives to switch to alternate fishing methods (longlines). The objectives of these measures were directed toward enabling the recovery of the species including all subpopulations (Australian Fisheries Management Authority, 2010). The Strategy has now been in place for over a decade, but its success in mitigating sea lion bycatch and enabling the recovery of affected populations has not been assessed.

The objectives of our study were to: (i) provide an overview of the methods used to estimate seal lion bycatch mortality in the GHAT fishery in the late 2000s and revise species distribution models and estimates of bycatch mortality and impacts on populations using updated abundance information; (ii) detail the chronology of bycatch mitigation measures developed and implemented as part of the ASL Management Strategy between 2010 and 2012; (iii) assess changes in fishing effort, bycatch mortality and the size of the sea lion population since the introduction of mitigation measures and (iv) evaluate the success of the ASL Management Strategy in reducing bycatch and enabling the recovery of sea lion populations. The steps of estimating sea lion bycatch and population impacts, implementation of bycatch management measures, and evaluation following management, are presented below in chronological order.

ESTIMATING SEA LION BYCATCH AND POPULATION IMPACTS

Fishery Background

The fishery for shark in southern Australia extends back to early European settlement in the 1800s and was enhanced during the Second World War (Kailola et al., 1993). At that time the fishery targeted school shark (*Galeorhinus galeus*) with longlines. In 1964, monofilament gillnet was introduced and by the early 1970s gillnetting was the main fishing method (Kailola et al., 1993; Larcombe and McLoughlin, 2007). Catch and effort records exist for this fishery in SA and adjacent Commonwealth waters since at least 1973, with catch of school shark peaking in 1987. The fishery now targets gummy shark (*Mustelus antarcticus*) and over the last decade and a half there have been efforts to reduce the catch of school shark to allow its stocks to rebuild (Australian Fisheries Management Authority, 2015b).

In 2001, an Offshore Constitutional Settlement (OCS) transferred State management of school and gummy shark in coastal waters (extending out to 3 nautical miles offshore, excluding internal waters in bays and inlets) to AFMA (Larcombe and McLoughlin, 2007). It is managed as part of the GHAT fishery. The gillnet sector of the GHAT fishery is restricted to depths shallower than 183 m.

Bycatch Assessment Approach

Bycatch rate (numbers of animals caught per unit of fishing effort) is a common metric used to estimate the level of bycatch to which a species is subjected (Sims et al., 2008; Moore et al., 2021). Bycatch models may be improved, however, by incorporating geographic information on species distribution at sea (e.g., species distribution models) and distribution of fishing effort, as bycatch rates are often highly heterogeneous. ASLs are good candidate species for this approach because they are a non-migratory, breed colonially, undertake regular foraging trips to sea and show strong philopatry to natal breeding sites (Campbell, 2003; Kirkwood and Goldsworthy, 2013). Furthermore, ASL display a high degree of faithfulness to foraging locations and foraging modes, with individuals settling on a mode of foraging at a very young age (Lowther et al., 2011, 2012, 2013). ASLs are demersal foragers and dive continuously to the seabed throughout foraging trips (Kirkwood and Goldsworthy, 2013).

The assessment of bycatch impacts on ASLs from the gillnet fishery off SA integrated data on subpopulation abundance (the size and status of individual breeding sites/subpopulations can be estimated from the number of pups born per breeding season and demographic data, Goldsworthy et al., 2021) and foraging data (satellite telemetry and diving data) to develop at-sea species distribution models (SDMs), with most data obtained between 2000 and 2009 (**Supplementary Tables 1**, **2**). Bycatch rates were estimated from a sample of observed gillnet hauls (2006–2008, see Hamer et al., 2013) and were then modeled in relation to ASL foraging effort at the location of the observed hauls, derived from the SDMs. This bycatch-rate estimation model was then applied to estimate the expected yearly bycatch that would have occurred from the spatial distribution of fishing effort between 2006 and 2009 (inclusive). Population viability analyses (PVA) then assessed bycatch impacts on the sustainability of individual subpopulations (see schematic in **Figure 1**).

Population Model Development

Subpopulation Pup Production

Information on the location and size (pup production) of ASL breeding sites (subpopulations) within SA waters in 2010 was based on Goldsworthy et al. (2010), with some modifications including: removal of eight locations that are now recognized not to be breeding sites [Bunda 1, 2, 4, Point Fowler (Camel Foot Bay), Dorothee Island, North Islet, Cave and Black Points]; the addition of nine recently identified breeding sites (Bunda 09, 152, 155, Cap, Rocky (South), Little Hummock and Williams Islands, Curta Rocks and the Western Isles), and revision of pup production estimates for Nuyts Reef due to poor earlier surveys (combined pup production revised from 15 to 112) (Goldsworthy et al., 2021). This has increased the estimated ASL pup production in SA in 2010 from 3,107 pups to 3,271 (5% increase) (see **Supplementary Table 1** and **Supplementary Figure 1**). Details on survey methodologies are presented in Goldsworthy et al. (2021).

Estimating the Size and Age Structure of Australian Sea Lion Subpopulations

The size of individual ASL subpopulations including the number of females and males was based on life-tables developed from demographic data collected at the Seal Bay subpopulation on Kangaroo Island (Goldsworthy et al., 2020). Age-specific survival estimates at Seal Bay were based upon the resight/return rates of 1,855 pups, microchipped across 11 consecutive breeding seasons between 2003 and 2018 (Goldsworthy et al., 2020).

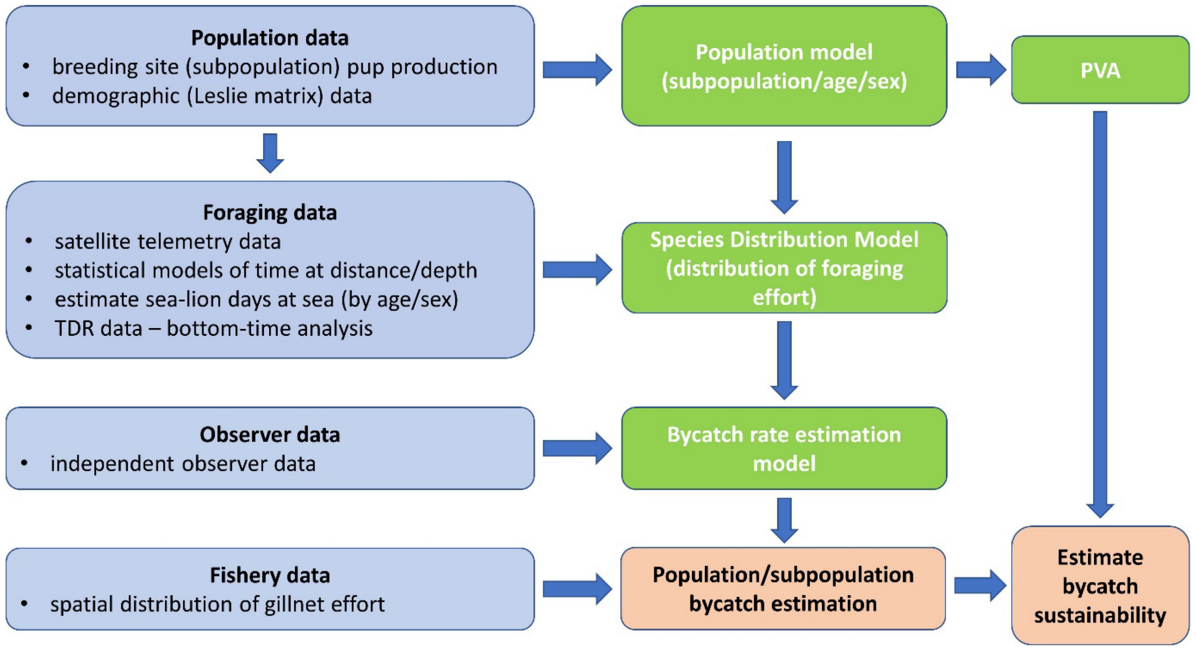


FIGURE 1 | Schematic representation of the various components of data used, and the steps undertaken to develop the suite of models (population, species distribution and bycatch rate estimation models) used to estimate the bycatch mortality of ASL and its impact on population sustainability.

Capture-history matrices were constructed from the re-sight histories of individual sea lions over eleven cohorts. Re-sights were grouped into 32, 6-month intervals (Summer/Autumn: December to May and Winter/Spring: June to November). Capture-history matrices were used as input files for the capture–mark–recapture (CMR) program MARK (White and Burnham, 1999) to estimate survival and capture probabilities. As there were limited data for animals older than 12 years, age-specific survival estimates were restricted to animals < 12. RMark estimates of mean survival were fitted to a fourth order polynomial model, assuming a maximum longevity of 26 years for females and 21.5 years for males (McIntosh, 2007; Goldsworthy et al., 2020; **Supplementary Table 3**).

Species Distribution Model Development

Satellite Telemetry Data

Satellite telemetry data from 210 instrumented ASLs provided the raw data from which the spatial distribution of foraging effort was modeled (**Supplementary Table 2**). These included 157 adult females from 17 subpopulations, 31 adult males from 8 subpopulations and 22 juveniles from 4 subpopulations. Data amounted to 3,321 individual foraging trips (foraging trips are discrete at-sea events between protracted periods ashore): 2,334 from adult females, 566 from adult males and 421 from juveniles (**Supplementary Table 2**). Pup foraging was not included in the analysis. Although the foraging abilities of pups develop markedly from 6 to 18 months (Fowler et al., 2006; Lowther and Goldsworthy, 2012), information about the distribution of their foraging effort is limited. Telemetry data were derived from both ARGOS linked platform transmitting terminals (PTTs), and fully archival or archival/ARGOS linked GPS tags. A total of 100,934 satellite-derived locations were available for analysis.

Filtering and Analysis of Time Spent in Areas

Platform transmitting terminal satellite location data were obtained through CLS ARGOS (Toulouse, France). The location-class Z positions were omitted due to the magnitude of their error (Sterling and Ream, 2004), leaving location classes B, A, 0, 1, 2, 3 for subsequent analyses. For GPS telemetry units, location data were solved either using the LocSolve (Wildlife Computers, Redmond, WA, United States) or Sirtrack (Havelock North, New Zealand) software packages. The R statistical software (version 2.8.1, R Development Core Team, R Foundation for Statistical Computing, Vienna) and the Trip package (M. D. Sumner, University of Tasmania, Hobart) were used to apply a speed filter as described by McConnell et al. (1992) to remove erroneous positions. The maximum horizontal speed considered possible was 11.0 km/h. In order to remove all time on land and restrict subsequent analyses to data on foraging trips only, the departure and arrival times, and locations of successive foraging trips were calculated following the methods detailed by Goldsworthy et al. (2009).

To determine key areas used during foraging trips, a grid of cells (1.5 km × 1.5 km, i.e., 2.25 km^2) was developed using the Trip package, and the amount of time that each sea lion spent within each cell was calculated assuming a constant horizontal speed between successive filtered locations and interpolated new positions every 15 min. Numbers of original and interpolated positions located within these cells were summed and assigned to the central node. To ensure that different deployment durations recorded for different sea lions did not bias comparisons, the amount of time spent in each cell was converted to a proportion of the total time spent at sea for each individual, subpopulation and or juvenile/adult (female/male) group (see examples in **Supplementary Figure 2**).

Model Development

The spatial distribution of foraging effort of ASL subpopulations throughout SA was estimated using statistical models. Alternate approaches using general linear models and generalized additive models were trialed but could not capture the over-dispersed nature of the observations and resulted in unrealistic truncations at natural limits to foraging distance and foraging depth.

Continental shelf and slope waters in SA were overlaid with a 1 km × 1 km grid and the distance from each subpopulation site to each node in the array was calculated. The depth at each cell node was calculated using bathymetric data from GeoScience Australia. For each subpopulation, the time spent at distance and depth from the subpopulation site was examined using density plots created within R. The fits of these density plots to the normal probability function

$$f(x;\mu,\sigma) = \frac{1}{\sqrt{2\pi\sigma}} e^{-\left(\frac{(x-\mu)^2}{2\sigma^2}\right)},$$

and the gamma probability density function,

$$f(x;k,\theta) = x^{k-1}\frac{e^{\frac{-x}{\theta}}}{\theta^k\Gamma(k)},$$

were examined using the MASS package. The means (μ) and standard deviations (σ) were calculated for normal probability distributions, while the shape (k) and scale (θ) functions were determined for gamma distributions. The x variable represented either distance (km) or depth (m). Where distance or depth distributions appeared to be bimodal, mixed models of two normal or gamma distributions were fitted using maximum likelihood in R (MASS package). Where there was support for two separate distributions, means and standard deviations were estimated for each and the proportion that each distribution contributed to the overall foraging distribution was calculated.

The probability of an animal from a given subpopulation foraging in a particular cell was modeled as a continuous variable on the range [0, 1]. The probabilities were calculated as the joint probability (i.e., product) of distance and depth (using the means and standard deviations, or shape and scale functions for the normal or gamma probability functions, respectively), based on the distance of the node from the subpopulation and its depth, after standardizing each onto the range [0,1]. For adult females, subpopulation models were used where tracking data from more than two individuals were available. For all other subpopulations, a combined model (all females) was used to estimate foraging distribution. For adult males, pooled models were used to estimate the foraging distributions for subpopulations in the western Eyre Peninsula (West), Spencer Gulf and Gulf St Vincent (Gulf), and Kangaroo Island regions (KI). Because data for juveniles were limited, a pooled model for all subpopulations was used. Models for depth and distance were assumed to be independent. Each foraging model was constrained by the upper limits of the observed distance and depth in the data on which it was based, and coastal distance data were corrected to exclude over-land routes.

Estimates of the proportion of time spent at sea by adult females (0.517), adult males (0.580) and juveniles (0.471) were based on those calculated by Goldsworthy and Page (2007). Total foraging effort (seal days/year) was estimated as the product of the number of individual sea lions in each age stage and sex (**Supplementary Table 3**), the proportion of time at sea and the number of days in a year. The distribution of foraging effort was apportioned to each subpopulation based on the proportion of pup production (relative to total SA pup production, **Supplementary Table 1**), the number of individual seals from each age stage and gender, and the proportion of time they spent at sea. This enabled the estimation of total foraging effort (seal days/year) for each subpopulation and its adult female, male and juvenile components. The product of the adult female, adult male and juvenile foraging probabilities at each node for each subpopulation, and their total foraging effort, provided the enumeration of the estimated spatial distribution of foraging effort (seal days/year). For any 1 km × 1 km node in the array, this enabled the estimation of total ASL foraging effort and the proportion of that effort attributable to females and males from any subpopulation.

To estimate the amount of time ASLs could be vulnerable to bycatch in gillnets, which are set on the seabed and extend only 3–4 m up into the water column, the seals' diving behaviors were assessed to determine how long they spent in this bottom phase of each dive. ASLs typically maintain a relatively constant bottom time duration independent of depth by increasing the duration of dives (Costa and Gales, 2003). However, as descent, ascent and inter-dive intervals increase with greater bottom depths, bottom time may decrease. To account for the potential effect of depth on bottom time, we examined data files from time-depth recorder (TDR, MK-7 TDRs Wildlife Computers, Redmond, WA, United States) deployments for 11 adult female and four adult male ASLs. These were analyzed using Instrument Helper (Version 1.0.0.5, Wildlife Computers) to derive dive depth, duration, bottom time (set as the time spent deeper than 80% of the maximum depth of each dive) and inter-dive interval. Percent bottom time in each dive cycle was then calculated as the duration of bottom time, divided by the dive duration plus the previous inter-dive interval. The effect of depth (log transformed) on percent bottom time (complementary log-log transformed) was examined using generalized linear mixed-effects models (GLMMs), using the lme4 package in R. A function describing how the proportion of bottom time varied with depth was derived and applied to each node within the foraging distribution model to provide an estimate of the total bottom (i.e., demersal foraging) time. For any 1 km × 1 km node in the array, this enabled the estimation of total ASL demersal foraging effort during which sea lions were at risk from interactions with set gillnets. As with overall foraging effort estimates, the proportion of demersal foraging effort attributable to females and males from any subpopulation for any node could be estimated. Foraging effort models were visualized and then interpolated (triangular

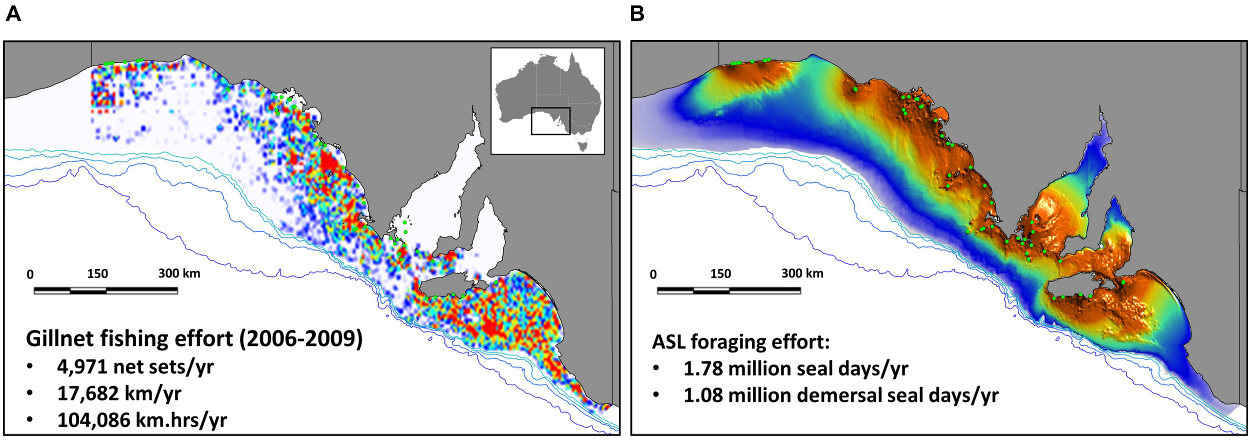


FIGURE 2 | Heat map representing the spatial distribution of gillnet fishing effort off SA, between 2006 and 2009 (inclusive, i.e., 4 years), with average annual effort provided (net-sets, km, and km.hrs) **(A)**; species distribution model (heat map) of ASL demersal foraging effort off SA based on estimated abundance in 2010, showing the location of 48 breeding sites (green circles) **(B)**. Bathymetry lines are indicated from light to dark blue (200, 500, 1000, 2000 m).

method) and plotted using MapInfo Pro Advanced^{TM} (Version 2019.3, Pitney Bowes Software Inc.).

Model Output

The SDM of the ASL population off SA emphasizes the importance of shallow coastal waters adjacent to breeding sites, and that the species' foraging effort is restricted to shelf waters (**Figure 2**). Details on the parameters of the normal and gamma probability density functions used to model the distributions of foraging effort, their assessment and evaluation, are detailed in Goldsworthy et al. (2010).

Bycatch Impact Assessment

Independent Fishery Observer Program

Independent observers accompanied shark gillnet vessels in SA shelf waters on ten trips between February 2006 and January 2008, with most trips undertaken between August and March (Hamer et al., 2013). Observations were made from slightly outboard of the gunwale to obtain an unimpeded view of the net ascending vertically through the upper water column and onto the net roller during net-hauls. This ensured that any sea lions that could drop out of the net as it came out of the water were recorded. Records were made of the time and location of each of the 234 hauls, and the presence of drowned ASLs. Where possible, the sex and age class of ASLs were recorded (Hamer et al., 2013).

Bycatch Rate Estimation Model

The latitude and longitude of net-sets were plotted onto the species distribution model of ASL foraging effort, and the foraging effort for each net-set was extracted using the point-inspection feature in MapInfo^{TM}. Observer data were then sorted from least to most ASL demersal foraging effort days, and then successive ranges (bins) in demersal foraging effort were examined in terms of the number of ASL bycatch mortalities and total fishing effort. Bycatch mortality rates for each successive bin were then calculated in terms of net-set length (seals/km net-set) and net-set length × soak-time duration (seals/km.hr net-set). An unsupervised discretization approach examined different binning options based on bins of equal length (range in demersal foraging effort) or equal frequency (number of observed net-sets). Many of these produced bin ranges or sample sizes that were too small or did not contain bycatch mortality and hence precluded a valid bycatch-rate calculation. Because of the low number of observations at areas of high sea lion foraging effort, binning based on equal sample sizes produced better model fits. Models to examine how bycatch rate per unit of fishing effort varied in response to sea lion demersal foraging effort were examined using linear regressions.

Bycatch Estimation

Bycatch was estimated using two approaches. The first multiplied the bycatch rates derived directly from the observer data by the total fishing effort to provide a simple approximation of total bycatch. For the second approach, we estimated ASL bycatch by using the estimation model to predict bycatch rates for values of demersal foraging effort estimated for each net-set location, and then multiplied these rates by total fishing effort at that location.

The foraging distribution models of adult females, adult males and juveniles enabled the total foraging effort at each node to be apportioned by sex and subpopulation. From this it was possible to estimate bycatch of each component for each subpopulation. Based on the ASL life-table (**Supplementary Table 3**) and the number of foraging days of juvenile, adult female and adult male sea lions (**Table 1**), total female and male bycatch at any node was calculated as the total adult female, or male bycatch plus the proportion of juvenile bycatch estimated to be female (0.4141), or male (0.5859), respectively.

To estimate ASL bycatch using the bycatch rate estimation model, the location of each net-set was required. However, prior to 2006, fishers only recorded catch and effort data within 1° × 1° blocks. In 2006, latitude and longitude recording of catch and effort was mandated by AFMA. As such, the level of historic

TABLE 1 | Estimates of the number of juvenile, adult female and adult male ASL in the SA population and the proportion of time spent at sea and onshore based on satellite tracking data and their estimated overall total foraging and demersal foraging effort (seal days/yr).

ASL age/sex	Estimated no.	Proportion of time		Foraging effort (days/yr)	Demersal foraging effort (days/yr)
		At sea	Onshore		
Juvenile	3,358	0.471	0.529	577,599	412,008
Adult female	3,582	0.517	0.483	676,158	357,440
Adult male	2,508	0.580	0.420	531,109	308,598
	9,448			1,784,866	1,078,046

bycatch was estimated based on fishing effort data for the period from the beginning of 2006–2009 inclusive (i.e., 4 years).

For this study, catch and effort data reported by latitude and longitude (to the nearest minute) were provided by AFMA. Effort data was reported in meters of net-set, and the duration of net-sets (soak-time, in minutes) was estimated based on the recorded time difference between the commencement of net-set to the commencement of net-haul.

Bycatch Impact on Population Viability

Population viability analyses incorporating Leslie matrix data for the Seal Bay subpopulation (**Supplementary Table 4**; Goldsworthy et al., 2020) were used to model changes in the abundance of all ASL subpopulations in SA through time, using the RAMAS® Metapop software (Version 3.0, Applied Biomathematics, Setauket, New York; Akçakaya, 1998). Only the female part of each subpopulation was modeled, so the estimated numbers in the first stage (pups) equaled half of the estimated pup production (assuming 1:1 sex-ratio at birth, **Supplementary Table 1**). Final stage survival rates were set to zero, with a standard deviation of 0.01 for all stage survival and fecundity estimates to provide a measure of environmental stochasticity (Akçakaya, 1998).

Density-independent PVAs were used to investigate the potential impacts of different levels of bycatch on ASL subpopulations. Individual subpopulations were modeled separately and assumed to be closed (i.e., no immigration or emigration). For ASLs, there is good evidence to support this assumption, with population genetic data indicating that the species demonstrates one of the highest levels of population subdivision among pinnipeds, with very high levels of mtDNA haplotype fixation among subpopulations (Campbell, 2003; Campbell et al., 2008; Lowther et al., 2012). These findings suggest that ASL females display extreme levels of philopatry, with little or no interchange of females among breeding colonies. Demographic stochasticity was simulated within RAMAS® Metapop, by sampling the number of survivors from a binomial distribution and pups from a Poisson distribution (Akçakaya, 1998).

PVAs were also used to investigate the potential implication of additional bycatch mortality on the status of each subpopulation. Conditional harvests within the population management feature of RAMAS® Metapop simulated the impacts of different levels of fishery bycatch, defined as the proportion of the total number of females aged > 1.5 years in a subpopulation removed per breeding season (1.5 years). Conditional harvests select only whole (integer) animals and select them from across all ages > 1.5 years, relative to their abundance within the subpopulation at the beginning of each modeled time step.

As the underlying rates of intrinsic growth are unknown for most ASL subpopulations, the implications of different bycatch rates were estimated for four intrinsic growth rates: 0%, 1%, 2% and 3% per breeding season. The different population growth models were simulated by adjusting relative survival levels and then calculating the resultant population trajectory (100 replicates of 34 breeding cycles, or 49.5 years).

Population viability analyses were used to predict the impact of different bycatch levels on future pup production. Non-pup stages were excluded from population totals at each time step to provide a time-series of estimated pup production for each simulation, because pup production is the principal metric used to estimate the status and trends in abundance of ASL populations (Goldsworthy et al., 2021). These were expressed as the exponential rate of increase (r), calculated from the slope of the exponential regression of pup numbers over time (breeding cycles); it was expressed as a percentage using the formula (e^r − 1) × 100.

Results of Bycatch and Population Impact Assessment

Overlap in Spatial Distribution of Gillnet Fishing Effort and Australian Sea Lion Foraging Effort

Annual fishing effort in the gillnet sector of the GHAT fishery off SA increased from around 3,000–12,000 km of net-set per year between 1973 and 1983, then to 43,000 km net-set in 1987. Fishing effort then decreased annually to about 23,000 km net-set in 1993 then increased to just over 32,000 km net-set in 1998. Annual fishing effort reduced in 2000 and remained at about 19,000 km net-set through to 2010, before declining to ∼3,000 km net-set to the end of June 2021 (most of this east of 139° longitude, i.e., outside of the ASL Management zones, see below) (**Supplementary Figure 3**).

The average annual fishing effort in the gillnet sector of the GHAT fishery off SA between 1 January 2006 and 31 December 2009 (4 years) was 4,971 sets (range 4,467–5,612), 17,682 km net-set (range 16,442–20,401) and 104,086 km.hr (range 92,213–121,258) (**Figure 2A**). Length of net-sets ranged between 1.0 and 6.2 km, with the most common lengths being 4.2 km (78%), 1.8 km (9%), 3.5 km (4%) and 2.4 km (3%). Soak-times averaged 5.8 h (sd = 2.6, range 0–24.0) and the net length times duration

TABLE 2 | Summary of the independent observer data collected in the shark gillnet GHAT fishery between February 2006 and February 2008.

Observation	Number	Bycatch rates
ASL bycatch	12	
Fishing trips	10	1.2 seals/trip
Observer days	146	0.082 seals/day
Net-sets	234	0.051 seals/net-set
km	944	0.013 seals/km
km.hrs	5,794	0.002 seals/km.hr

gave an average of 20.9 km.hrs per net-set (sd = 11.4, range 0.1–102.0).

Based on estimates of ASL pup production in 2010 (3,271), the total SA population size was 12,719, including 3,358 juveniles, 3,582 adult females, and 2,508 adult males (**Table 1** and **Supplementary Table 3**). The total seal lion foraging effort was estimated to be 1,784,866 seal days/yr, or 1,078,046 demersal seal days/yr (**Table 1**). There was almost complete spatial overlap in gillnet fishing effort and sea lion foraging effort off SA in the mid-late 2000s (**Figure 2**). The main exceptions are the absence of fishing effort in the two gulfs and some of the coastal bays. These near coastal areas became closed to the fishery in 2001 when management of the school and gummy shark fishery was transferred from the State to the Commonwealth (Larcombe and McLoughlin, 2007). The very high degree of overlap in the gillnet fishing and sea lion foraging efforts is also apparent when effort is compared relative to depth and the minimum distance from ASL subpopulations (**Supplementary Figure 4**).

Observer Data

Observer data were collected over 146 sea days on ten trips (**Table 2**). A total of 994 km of net was observed hauled during 234 net-sets (Hamer et al., 2013), which equated to 19% of the total gillnet fishing effort over the 2-year program, or 37% of mean annual fishing effort. Twelve ASL bycatch mortalities were recorded; 10 (83%) of the dead sea lions dropped-out of the gillnet before or on contacting the net roller, as they ascended from the water (Hamer et al., 2013). The two (17%) dead sea lions that made it onto the deck of the vessel were small juveniles (one female, one male). Eleven of the 12 sea lions could be sexed, nine were female (6 adult, 3 juvenile) two were male (1 adult, 1 juvenile) and one juvenile was unable to be sexed (Hamer et al., 2013). The bycatch mortality rates based on the pooled observer data equated to 0.0513 seals/net-set, 0.0127 seals/km and 0.0021 seals/km.hr net-set (**Table 2**). Although the observer effort was largely collected from fishing activity off the western and lower Eyre Peninsula (**Figure 3**), the distribution of observer data relative to fishing depth and ASL foraging effort was representative of that of the broader fishery (**Supplementary Figure 5**).

Bycatch Rate Estimation Model

Using the species distribution model, sea lion foraging effort was estimated at the locations of the 234 independently observed net-sets (**Figure 3**). The expectation was that sea lion bycatch per unit of fishing effort would increase with increasing demersal foraging effort (the probability of encountering sea lions) (**Figure 3**). Because the percentage of net-sets observed was low and there was a large variation in underlying demersal foraging effort (0–44 d), individual observed net-sets were binned by increasing foraging effort. Five models were compared, with bin size ranging from four to eight (**Table 3**). The sample size (number of net-sets observed) within each bin was approximately equal for each model. Bycatch rate (seals/km and seals/km.hr) was significantly related to the underlying likelihood of encountering sea lions (demersal foraging effort) for all models (**Table 3**), was strongly linear and there was support for models with regression lines passing through the origin (**Figure 3**). This enabled the slopes of all models to be easily compared (**Table 3**). Model fits to variable bin numbers and sample sizes were examined, with the optimum derived from a 5-bin model (**Table 3** and **Figure 3**). The statistical strength of these relationships supports the use of linear models to estimate the likely level of sea lion bycatch that would result from any level of fishing effort for any location.

The 1 km × 1 km array of ASL foraging effort contained 258,235 nodes where demersal foraging effort was > 0 (maximum value 687 seal days/year). Demersal foraging effort at the locations of the 234 observed net-sets ranged from 0–44 seal days/year, with only 1.3% of the 1 km × 1 km array nodes exceeding 44 seal days/yr. As no data were available to determine if the rates of sea lion bycatch per unit of fishing effort were the same as those described above in regions with > 44 seal days/yr, an upper limit of demersal foraging effort was set to the mean and ± 95% CL for 44 seal days/yr. This approximation will likely under-estimate the bycatch mortality rate in areas of very high ASL demersal foraging effort.

Estimated Sea Lion Bycatch

Based on vessel-observed ASL bycatch rates and the average level of fishing effort prior to bycatch mitigation efforts (2006–2009 inclusive), the annual bycatch mortality of ASLs in the gillnet (GHAT) fishery off SA was estimated to be 225 based on a bycatch rate of 0.0127 seals per km net-set, and 216 based on a bycatch rate of 0.0021 seals per km.hr net-set (**Table 4**).

Based on the spatial distribution of fishing effort between 2006 and 2009, and the km.hr net-set model, the average annual ASL bycatch using the bycatch rate estimation method was 242 (209–278 ± 95% CL, **Table 4**). Annual and per-breeding cycle bycatch estimates for the km net-set models were about 18% higher than the km.hr net-set models. Bycatch estimates using the latter models are expected to be more accurate, given that the risk of bycatch is a function of both the net-set length and soak-time duration. Female ASLs accounted for 51% of the total estimated bycatch mortality, with an estimated bycatch of 125 (108–143, ± 95% CL) annually, or 182 (157–209) per breeding cycle (**Table 4**).

Bycatch Impact on Population Sustainability

Based on the ASL life-table with stable (0%) intrinsic growth, natural mortality of females > 1.5 years of age is 17.5% per 18 month breeding cycle. With the addition of the estimated

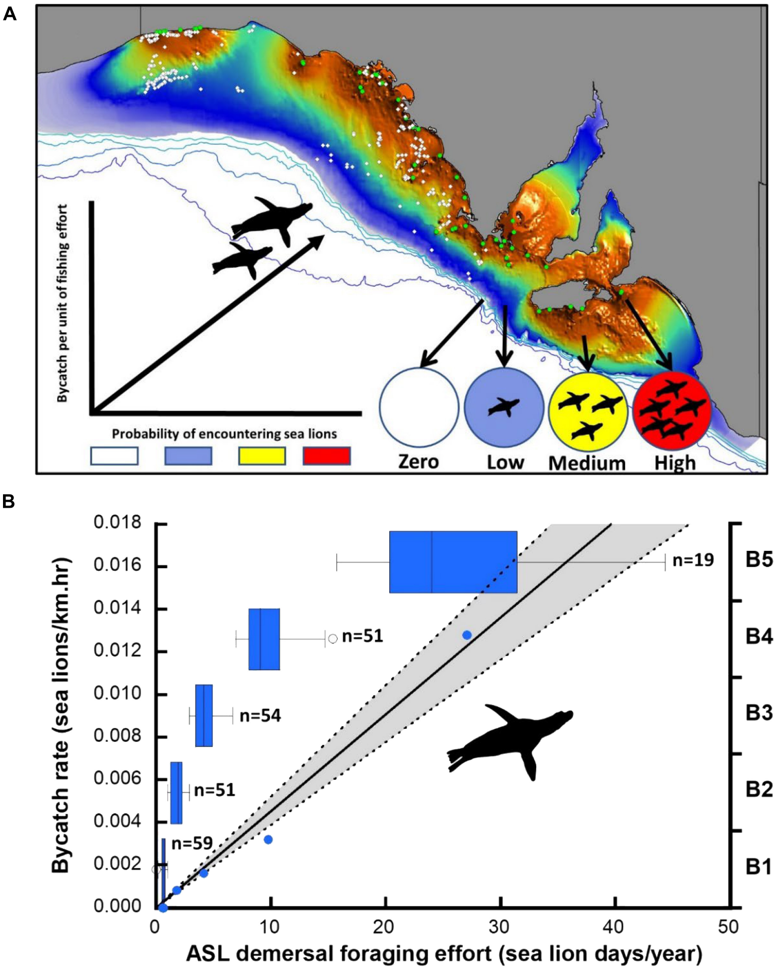


FIGURE 3 | Distribution of observed net-sets (open circles) in the gillnet sector of the GHAT fishery off SA relative to the species distribution model of ASL demersal foraging effort **(A)**. Colony locations (green circles) and the expected relationship between the probability of encountering sea lions and the bycatch rate per unit of fishing effort are indicated. **(B)** Linear model of the observed bycatch rate per unit of fishing effort and estimated ASL demersal foraging effort for the optimal 5-bin model (B1-B5), including box-plots to illustrate the sample size (observed net-sets) and foraging effort range differences of successive bins (blue circles are the data points underpinning the regression, 95% CL shaded).

level of female bycatch mortality (based on fishing effort between 2006 and 2009, inclusive), these rates increase by 3.6% (3.1–4.1%, ± 95% CL) to 21.0% (20.5–21.6%, ± 95% CL), representing a 20.5% (17.6–23.5%, ± 95% CL) increase above natural mortality levels in a stable population.

Between 2006 and 2009 (inclusive), an average of 2.6% (2.2–3.0% ± 95% CL, range 0–9.2%) of females (>1.5 years) in each subpopulation was estimated to have been lost to gillnet bycatch mortality per breeding cycle, assuming stable population growth.

Estimated impacts of bycatch on each subpopulation under different intrinsic growth scenarios (0, 1, 2, and 3%/year) vary markedly, with those off West Coast and southern Eyre Peninsula and in the Kangaroo Island region, expected to have the greatest rates of decline (**Figure 4**). Subpopulations within Spencer Gulf were least impacted as there was limited overlap between foraging distributions and fishing effort. Under the stable (0%) growth scenario, 92% of subpopulations were estimated to be in decline, including all subpopulations that overlapped with the fishery, with average growth rates of −3.5%/year (−4.1 to −2.4% ± 95% CL, range −20.1 to 0.0%). With a 1% growth scenario, 52% of subpopulations were estimated to be in decline, with average growth rates of −1.8%/year (−2.4 to −0.9% ± 95% CL, range −18.9 to 1.0%). With a 2% growth scenario, 40% of subpopulations were estimated to be in decline, with average growth rates of −0.1%/year (−0.6 to 0.6% ± 95% CL, range −16.3 to 2.0%). With a 3% growth scenario, 13% of subpopulations were estimated to be in decline, with average growth rates of 1.5%/year (1.1–1.9 to % ± 95% CL, range −9.4 to 3.0%) (**Figure 4**).

IMPLEMENTATION OF BYCATCH MANAGEMENT MEASURES

A timeline of the key bycatch mitigation and management actions implemented as part of the ASL Management Strategy between 2010 and 2013 by AFMA is presented in **Box 1**.

TABLE 3 | Statistical and model coefficient outputs for alternate bycatch estimation models based on the number of data bins and their expression in terms of fishing effort (km and km.hrs).

Model	Seals/km net-set/demersal foraging effort					Seals/km.hr net-set/demersal foraging effort				
	Slope	−95%CL	+95% CL	*P*	r^2	Slope	−95%CL	+95% CL	*P*	r^2
4-bin	0.00253	0.00148	0.00359	<0.05	0.95	0.00042	0.00025	0.00060	<0.05	0.95
5-bin	0.00289	0.00176	0.00402	<0.01	0.93	0.00048	0.00029	0.00062	<0.001	0.93
6-bin	0.00287	0.00182	0.00392	<0.01	0.91	0.00045	0.00029	0.00062	<0.001	0.91
7-bin	0.00265	0.00214	0.00316	<0.001	0.96	0.00042	0.00034	0.00050	<0.001	0.97
8-bin	0.00278	0.00206	0.00351	<0.001	0.93	0.00044	0.00032	0.00055	<0.001	0.92
Mean	0.00275	0.00148	0.00402			0.00044	0.00025	0.00062		
Opt. bin	0.00313	0.00251	0.00375	<0.001	0.98	0.00045	0.00039	0.00052	<0.001	0.99

*Comparison of the slope coefficients (±95% CL), significance (P) and r^2 are presented, as is the mean slope coefficients of the 4-8-bin models (±95% CL). Opt. bin is the optimal bycatch rate estimation model derived from a 5-bin model with variable sample size (see **Figure 3**).*

TABLE 4 | Estimated ASL bycatch mortality in the gillnet sector of the GHAT fishery off SA based on observed rates of bycatch calculated from seals/km net-set (0.0127) and seals/km.hr (0.0021); and based on bycatch rate estimation (seal/km net-set and seals/km.hr), net-length (km) and net-length × soak time (km.hrs).

Method	Estimated bycatch mortalities	
	Annual bycatch	Breeding cycle bycatch
Observer-based (km)	225	337
Observer-based (km.hr)	216	323
Model-based (km) Females	147 (117–176)	214 (171–256)
Males	139 (111–167)	203 (162–243)
Total ASL	286 (229–343)	417 (334–500)
Model-based (km.hr) Females	125 (108–143)	182 (157–209)
Males	118 (102–135)	172 (148–197)
Total ASL	242 (209–278)	353 (306–406)

Bycatch mortality based on bycatch rate estimation relates to the actual distribution and level of fishing effort between 1 January 2006 and 31 December 2009. Bycatch mortalities have been estimated on an annual and per breeding cycle (1.5 year) basis.

Prior to the release of a report in April 2010 detailing unsustainable levels of ASL bycatch in the gillnet sector of the GHAT (Goldsworthy et al., 2010), AFMA undertook an observer program off SA (July 2009 to June 2010) that included sea lion specific observer protocols (100% of net-hauls observed to check for drop-outs). This included 108 sea days and 109 net-hauls (Australian Fisheries Management Authority, 2010). In December 2009 the gillnet fishing industry also introduced voluntary gillnet closures, 4 nm (7.3 km) in radius around all 48 known ASL breeding sites in SA (Australian Fisheries Management Authority, 2010).

In June 2010, AFMA implemented the ASL Management Strategy (**Box 1**). Key elements included increased independent observer coverage set at 11% of net sets within each of seven management zones (A-G) and formalizing the 4 nm fishery closures around all ASL colonies in SA (Baseline Closures), increasing the radius of closures to 6 nm (11.1 km) and 8 nm (14.8 km) around a number of ASL breeding sites estimated to be exposed to higher levels of bycatch mortality in the fishery (Goldsworthy et al., 2010) and additional 4 nm strip closures along the Bunda Cliffs and south coast of Kangaroo Island (Australian Fisheries Management Authority, 2010; **Box 1**). To reduce the incidence of bycatch in the areas open to the fishery, AFMA introduced bycatch mortality limits across each of the seven management zones that would trigger temporal spatial closures for the remainder of the financial year when zone-specific bycatch trigger limits were reached. Zone bycatch trigger limits ranged from 3–6 ASL per season with an overall trigger limit of 15 ASL for all of SA, based on 11% observer coverage. If 15 or more ASL mortalities were observed in a season, the remaining areas of the fishery would be closed until the end of the fishing season (Australian Fisheries Management Authority, 2010).

In May 2011, in response to under-reporting of marine mammal interactions by some fishers, AFMA introduced its Sea lion Temporary Order 1 (TO1). It increased observer coverage to 100% within an "Australian Seal Lion Management Zone" (SA coastal waters between 129° and 139° east longitude), either using on-board observers or electronic monitoring systems (EMS) that were being introduced into the fishery at the time. With observer effort increasing from 11 to 100%, the per zone observed trigger limits were increased to 3–16 ASL per season, with an overall trigger of 52 females or 104 ASL in total. The radius of some gillnet fishing closures around sea lion breeding sites were increased to 11 nm (20.4 km), including extension of the Bunda Cliffs and Kangaroo Island strip closures. In addition, TO1 gave affected fishers the option to switch to hooks (demersal longlines) in areas closed to gillnets, with 10% (rather than 100%) observer coverage (**Box 1**).

In September 2011, following 49 dolphin interactions in the previous 12 months, AFMA enacted a Dolphin Temporary Order which introduced a dolphin gillnet closure (27,239 km^2) between Kangaroo Island and Cape Jaffa (see location in **Box 1**). A Dolphin Observer Zone was introduced in waters adjacent to the closed area that required 100% observer effort when using gillnets (**Box 1**). As with

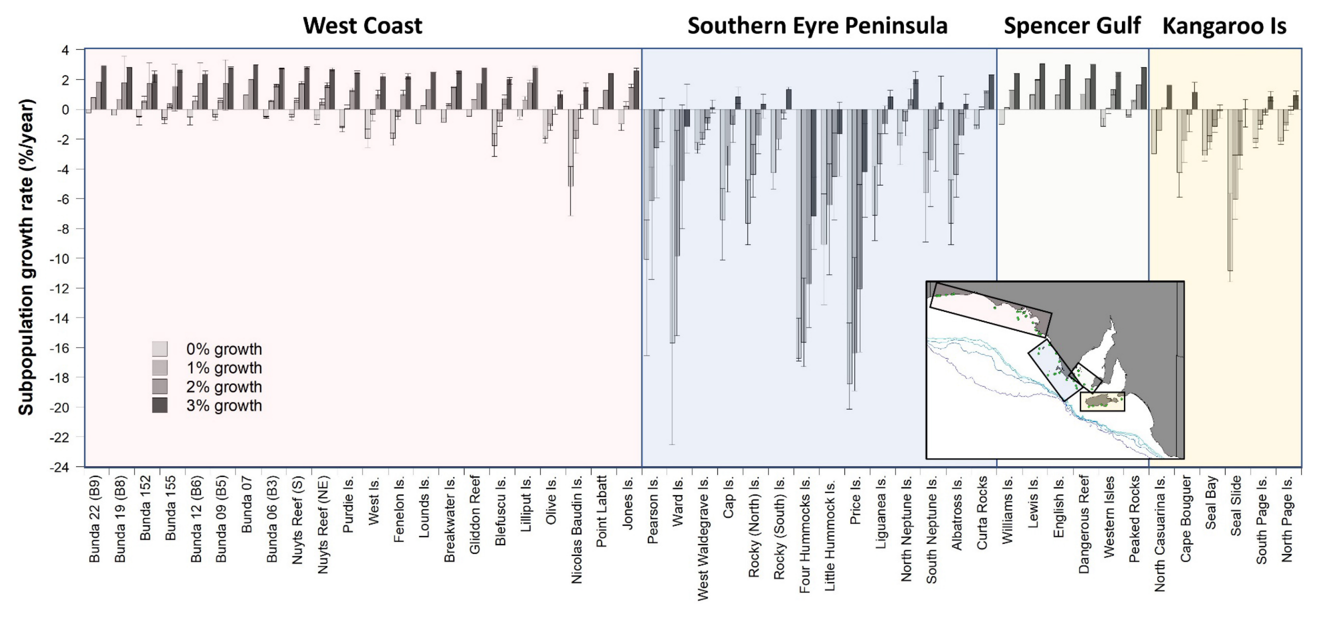


FIGURE 4 | Estimated ASL subpopulation growth rates (±95% CL) based on the estimated bycatch mortalities resulting from the distribution of fishing effort in the gillnet GHAT fishery off SA from 2006 to 2009 (inclusive). Subpopulation growth rates are estimated for 34 breeding cycles (~50 years) for four different intrinsic annual growth-rate scenarios (0, 1, 2, and 3%). Four regions are identified: West Coast, Southern Eyre Peninsula, Spencer Gulf and Kangaroo Island.

TO1, affected fishers were given the option to switch to longlines in gillnet closures, with 10% observer coverage. In November 2011, a second Sea lion Temporary Order (TO2) commenced that effectively extended the ASL gillnet fishery closures and required 100% observer coverage from TO1 until April 2013.

Following consultation with marine mammal experts and other stakeholders, AFMA introduced several changes to the bycatch trigger limits and management zones in January 2012. These included a marked reduction in the per zone bycatch trigger limits to 1–5 ASL per fishing season with an overall trigger limit of 15 ASL, and with zones closed for 18 months from the time the trigger limit was reached (instead of for the remainder of the fishing season). In addition, two zone boundaries (B/C, C/D) were modified to be almost perpendicular to the coast (**Box 1**). Following these management changes and the bycatch mortality of seven ASL over a 4-month period, three zone closures were triggered: Zone A (1 ASL death; closed Feb 2012, reopened May 2013); Zone B (3 ASL deaths; closed Mar 2012, reopened Aug 2013) and Zone D (1 ASL death; closed Apr 2012, reopened Aug 2013). Single ASL bycatch mortalities were reported for Zone C (trigger limit of 2 ASL) and Zone E (trigger limit of 2 ASL), with their zone trigger limits reset to zero in May 2012.

In April 2013, AFMA extended the TO1 gillnet fishing closures and added new radial closures around two newly detected ASL breeding sites at Cap and Rocky (South) Islands. Since then, two management zones were triggered and subjected to temporal closures: Zone C (2 ASL deaths; closed Jan 2016, reopened Jun 2017) and Zone D (1 ASL death; closed Sep 2017, reopened Mar 2019) (**Box 1**). The permanent spatial closures to gillnet fishing off SA introduced between 2010 and 2013 covered a combined area of 18,500 km^2 (Australian Fisheries Management Authority, 2015a).

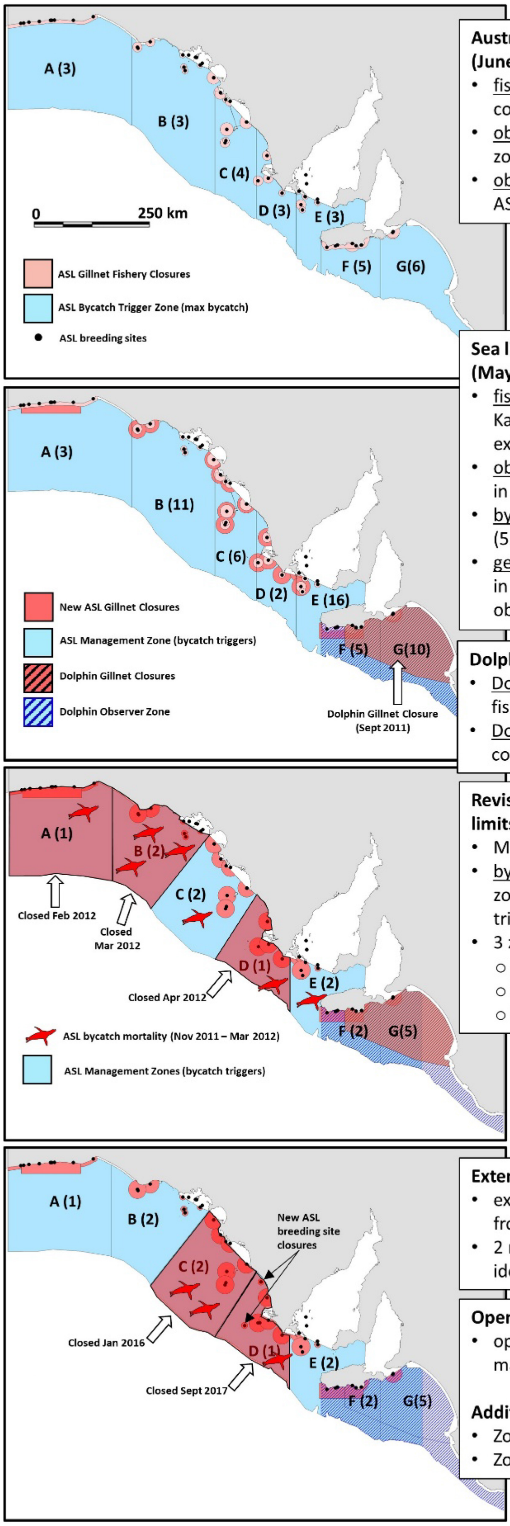
EVALUATING SUCCESS OF THE AUSTRALIAN SEA LION MANAGEMENT STRATEGY

Impact of Management Actions on Fishing Effort and Estimated Australian Sea Lion Bycatch Mortality

Using the methods developed to quantify ASL bycatch mortality (section "Estimating Sea Lion Bycatch and Population Impacts"), fishing effort data from 1 January 2006 to 30 June 2021 were used to estimate changes in ASL bycatch in the gillnet sector of the GHAT fishery off SA (i.e., estimating ASL demersal foraging effort for each net-set location, and with net soak time, applying the bycatch rate estimation model to estimate sea lion bycatch mortality) and concomitant changes in effort of the gillnet and the longline fisheries.

The introduction of management measures to mitigate ASL bycatch (largely through the ASL Management Strategy) had immediate effect on gillnet fishing effort and ASL bycatch mortality (**Figure 5**). Within 2 years of the implementation of the ASL Management Strategy, gillnet fishing effort had reduced by 80%, and ASL bycatch mortality had declined by an estimated 84% (**Figure 5**). By June 2021, gillnet fishing effort had declined by 95% off SA (and by 98% within the ASL Management Zone), and estimated ASL bycatch mortality had declined by

BOX 1 | A schematic of the key bycatch mitigation and management actions implemented as part of the ASL Management Strategy between 2010 and 2013.



Australian Sea Lion Management Strategy (June 2010)
- fishery closures: 4-8nm around ASL colonies
- observer coverage: 11% within each of 7 zones (A-G)
- observed bycatch trigger limit: 3-6 ASL/zone/season (15 ASL overall)

Sea lion Temporary Order 1 (TO1) (May 2011)
- fishery closures: increased up to 11nm, Kangaroo Is & Bunda Cliffs closures extended
- observer coverage: 100% (on-board/EMS) in ASL Management Zone (129-139° E)
- bycatch trigger limit: 3-16 ASL/zone/season (52 female ASL overall)
- gear switching: option to switch to longline in areas closed to gillnets with 10% observer coverage

Dolphin TO1 (Sept 2011)
- Dolphin Gillnet Closure: closed to gillnet fishing
- Dolphin Observer Zone: 100% observer coverage when using gillnets

Revised Management Zones/Bycatch trigger limits (Jan 2012)
- Modified zone boundaries
- bycatch trigger limits: reduced (1-5 per zone, 15 in total), 18m zone closure when trigger limits reached
- 3 zone closures in three months:
  - Zone A (1 ASL) Feb 2012 - May 2013
  - Zone B (3 ASL) Mar 2012 - Aug 2013
  - Zone D (1 ASL) Apr 2012 - Aug 2013

Extension of ASL fishing closures (Apr 2013)
- extension of ASL colony fishing closures from TO1
- 2 radial fishing closures to protect newly identified ASL breeding sites

Opening Dolphin Gillnet Closures (Sept 2014)
- opened to gillnet fishing under new management arrangements

Additional Zone closures
- Zone C (2 ASL) Jan 2016 - June 2017
- Zone D (1 ASL) Sept 2017 - Mar 2019

98% from pre-bycatch management levels. Concomitant with the management restrictions on gillnets, the use of demersal longlines increased fivefold over pre-bycatch management levels (**Figure 5**). Despite these marked changes in fishing effort and gear type, the catch of the main target species (gummy shark) taken within the ASL Management Zone was similar in

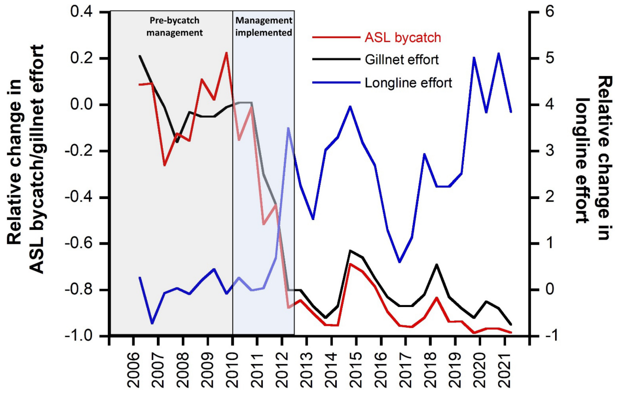


FIGURE 5 | Changes in gillnet and longline fishing effort and estimated ASL bycatch relative to average pre-management levels (2006–2009, inclusive) in the GHAT fishery off South Australia. Time periods are highlighted for before and after implementation of bycatch management.

2020/21 to that in 2009/10, the year prior to the implementation of the ASL Management Strategy (264,342 and 265,038 kg, respectively). Over this time, the portion of the catch taken by gillnets declined from 96 to 2%, while the portion of the catch taken using demersal longlines increased from 4 to 98%.

Impact of Management Actions on Australian Sea Lion Bycatch

Reported interactions with seals in the GHAT fishery off SA since the introduction of 100% monitoring of fishing effort in 2011 (by observers, electronic monitoring and vessel logbooks) declined consistently (**Figure 6**). The species of seals recorded interacting with fishing activity by fishers in their logbooks, or recorded through EM is not always reliable, but it is likely that most were ASL. In 2011 and 2012, a total of 11 and 10 interactions were recorded, respectively, and following the 18-month closures of three ASL Management Zones in 2012, only one seal interaction was reported. When these three zones re-opened, seal interactions increased to eight in 2014, and steadily declined thereafter.

Impact of Management Actions on Change in Sea Lion Abundance

Although the ratio of pups to total population varies in relation to a pinniped's population status (declining, stable, increasing), monitoring changes in pup numbers over time still provides an valid index of change in population growth (Berkson and DeMaster, 1985). The data on pup abundance for ASL breeding sites off SA is patchy, with time-series data only available for a subset of breeding sites and for variable time periods (Goldsworthy et al., 2021). To assess if there has been a change in population abundance prior to and following the introduction of bycatch mitigation measures in the GHAT fishery off SA, we compared the total pup abundance from a subset of 12 ASL breeding sites from the area of the fishery (Spencer Gulf populations excluded) that had been surveyed in each of three main surveys conducted in the mid-2000s (2004–2006), the mid-2010s (2014–2015) and the late 2010s (2019–2020). Total pup numbers declined by 26.6% between the mid-2000s and mid-2010s. This decline appeared to have been arrested by 2020, with a 1.5% increase in pup numbers between 2015 and 2020 (**Figure 6**). The strongest indication that the decline had been arrested came from the west-coast Eyre Peninsula breeding sites that made up most (75%) of the monitored breeding sites. These sites showed a 34.5% decline between the mid-2000s and mid-2010s (701–459) then increased by 5.2% up to 2020 (**Figure 6**).

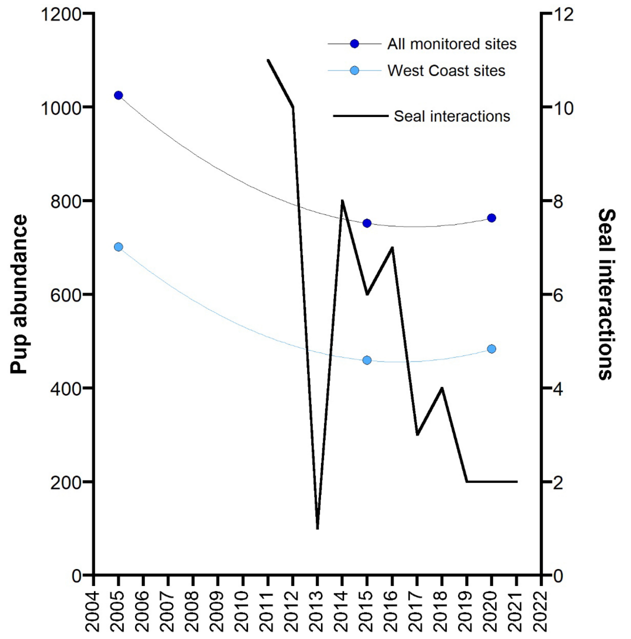


FIGURE 6 | Changes in the total pup abundance from 12 monitored ASL breeding sites in the area of the GHAT fishery off South Australia across three main survey periods: the mid-2000s (2004–2006), mid-2010s (2014–2015) and the late 2010s (2019–2020). Data for a subset of these sites on the west coast of the Eyre Peninsula (West Coast sites) are also presented. Both data sets are fitted to a second order polynomial. The reported number of interactions with seals in the GHAT fishery off SA since the introduction of 100% monitoring of fishing effort (2011) (by observers, electronic monitoring and vessel logbooks) are also presented.

DISCUSSION

Estimating Pre-management Bycatch and Its Impact on Australian Sea Lion Populations

This study confirms previous reports indicating that high levels of bycatch mortality of ASL occurred in the demersal gillnet sector of the GHAT fishery off SA, prior to the introduction of management measures to mitigate bycatch in 2010 (Goldsworthy et al., 2010; Hamer et al., 2013). The level of bycatch mortality impacting most subpopulations then was likely to be unsustainable and may have led to subpopulation extinctions and reductions in sea lion range unless the bycatch mortality was reduced.

This study provides a unique and comprehensive assessment of bycatch impact that combined independent observer data with species distribution models (underpinned by extensive satellite tracking, abundance data and population modeling), enabling the relationship between sea lion foraging effort and observed bycatch rate to be modeled. These models allowed sea lion bycatch to be estimated across the fishery, the impacts on subpopulations to be assessed, and the potential benefit of alternate management options (gillnet fishing closures, redistribution of fishing effort) to be evaluated (Goldsworthy et al., 2010). Results were clear and compelling and drove prompt management action. The approach provides a good example of how spatial analyses of marine megafauna movement data can be used to inform marine spatial management (Sequeira et al., 2019). Although species distribution models have been used previously to identify potential bycatch hotspots (Thorne et al., 2019), this study is unique in demonstrating that foraging effort (encounter probability) correlates with bycatch rate. Moreover, the results enabled the development of bycatch rate estimation models that were used to estimate bycatch across the fishery, and inform management options (Goldsworthy et al., 2010).

Spatial analyses indicated almost complete overlap between the distributions of ASL foraging and gillnet fishing effort with most of the pre-management fishing effort occurring within the depth and distance limits of foraging sea lions. The only subpopulations where overlap was likely to be low were in southern Spencer Gulf, which was closed to the GHAT fishery in 2000. Using the bycatch rate estimation model, average annual bycatch mortality of sea lions in the late-2000s, using updated population data, was estimated to be 242 (209- 278); very similar to that estimated by Goldsworthy et al. (2010) (256 sea lion per year, 187–347). These values are about 12–18% higher than those obtained using a simple multiplication of vessel-observed bycatch rates with total fishing effort and are likely to be more accurate because they consider the spatial heterogeneity in interaction probability. They are also about 15% greater than the estimate of Hamer et al. (2013) (193–227 per year) based on fishing effort off SA between 2000 and 2008. Bycatch mortality was estimated to increase total mortality (natural + bycatch mortality) by $\sim$20% (assuming stable intrinsic growth), and to have contributed significantly to the decline in many ASL subpopulations (e.g., 40–92% of subpopulations in decline with intrinsic growth rates ranging from 0–2%/year).

The impacts of bycatch mortality on the sustainability of ASL populations off SA have likely been substantial, given the levels of historic gillnet fishing effort and the absence of management regulations to reduce bycatch for almost 50 years since gillnets were introduced into the fishery in 1964 (Kailola et al., 1993; Larcombe and McLoughlin, 2007). Total fishing effort in the late 1980s and early 1990s was more than double the mean fishing effort between 2000 and 2010 when bycatch of ASL (pre-management) was assessed (**Supplementary Figure 3**). The distribution of ASL abundance across the species' range is notably uneven, with small subpopulations next to medium and larger sites and no apparent spatial pattern (Goldsworthy et al., 2021). This marked within-region heterogeneity in the size and trends in subpopulations likely reflects variability in natural and anthropogenic factors at a local scale (Goldsworthy et al., 2021). Pronounced inter-site and inter-individual differences in foraging strategies, identified from tracking studies (Lowther and Goldsworthy, 2011) potentially facilitate the shaping of within-region differences in population dynamics. Vulnerability to bycatch mortality is tightly coupled to the foraging distributions of individual sea lions. As such, subpopulations and individual sea lions with foraging distributions that closely match the distribution of fishing effort are likely to have been highly selected against over multiple decades of interactions with the gillnet fishery, potentially shaping the uneven distribution of abundance and the prevalence of small and declining subpopulations (Goldsworthy et al., 2021). Evidence from this study suggests that sequential depletion of ASL subpopulations from bycatch mortality in the gillnet fishery is likely to have contributed significantly to the marked (>60%) decline in the species abundance over the last 40 years, and to their Endangered status (Goldsworthy, 2015; Goldsworthy et al., 2021). Our analyses support the assessment of Goldsworthy et al. (2020) that historic (pre-management) bycatch in the gillnet fishery could explain an up to 7%/year decline in ASL numbers off SA.

As significant as historic (pre-management) bycatch levels have likely been, our analyses probably underestimated the magnitude of bycatch mortality in the gillnet fishery and its impacts on ASL populations. Our observer program recorded 10 of the 12 (83%) observed sea lion bycatch mortalities to drop out of the gillnet before reaching the net roller and deck. Based on this, fewer than 20% of bycaught animals could reach the deck of fishing vessels. The body size of ASLs caught in gillnets is likely to be the main factor determining the likelihood of dropping out, as the two animals observed to reach the deck were small juveniles. It is likely that an additional portion of bycaught sea lions drop out below the surface as the net tension increases during hauling operations. These would be undetectable to an observer on the vessel. The extent of sub-surface drop-outs, or cryptic mortality is unknown but its contribution to overall bycatch mortality could be significant. If cryptic bycatch contributed an additional 5–10% bycatch mortality, the implication for the sustainability of ASL populations would be substantial. A number of studies have attempted to estimate cryptic bycatch mortality of marine mammals, mostly of small cetaceans (Moore et al., 2021). Cryptic sources of fishing mortality are not just an issue for marine mammal bycatch and its estimation can be very challenging (Gilman et al., 2013).

Key Management Actions

Following the release of a report detailing unsustainable levels of ASL bycatch in the gillnet sector of the GHAT off SA (Goldsworthy et al., 2010), AFMA's implementation of the ASL Management Strategy had immediate impacts on the fishery. It resulted in significant reductions in fishing effort (especially in proximity to sea lion breeding sites) and on the reported bycatch mortality of sea lions. In the decade since these measures were introduced, there has been an almost complete transition to alternate fishing gear (gillnets to longlines), a reduction in the numbers of reported ASL interactions and an apparent stabilization in pup abundances of some impacted ASL populations. Key elements of the ASL Management Strategy that have reduced the bycatch of ASLs are discussed below.

Independent Observer Program

The comprehensive observer program developed through the ASL Management Strategy has arguably underpinned the success of the other measures introduced to reduce the bycatch mortality of ASLs. Following introduction of the Strategy, independent observer coverage was set at 11% of net sets within each of seven ASL Management Zones (A-G) but was increased to 100% in May 2011 following under-reporting of marine mammal interactions by some fishers. Observer coverage was met initially by on-board observers or using electronic monitoring systems, but from July 2015, electronic monitoring became mandatory (Australian Fisheries Management Authority, 2015a). AFMA independently reviewed all footage from vessels using gillnets in the ASL Management Zone, and between 2011 and 2015 detected no cases where a fisher failed to report an ASL interaction (Australian Fisheries Management Authority, 2015a). The introduction of electronic monitoring across a number of AFMA managed fisheries has led to a significant increase in logbook reporting of protected species interactions, including ASLs and dolphins in the GHAT fishery (Emery et al., 2019). Under Australia's *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act), all interactions with protected (*EPBC Act*–listed) species must be reported in fishery logbooks.

Spatial Fishing Closures

A combination of permanent and temporal spatial fishing closures was the core mitigation strategy to reduce ASL bycatch mortality. The intent of permanent gillnet fishing closures was to remove fishing effort from areas of high sea lion foraging effort where the impacts on subpopulation sustainability were potentially greatest. Permanent gillnet fishery closures were introduced around all SA ASL breeding sites in 2010, with further increases introduced in 2011 and 2013 (Australian Fisheries Management Authority, 2015a). The different sizes of permanent closures were based on the combination of bycatch risk (estimated number of mortalities), subpopulation vulnerability (extinction risk) and size (extra protection to larger breeding sites) (Goldsworthy et al., 2010; Australian Fisheries Management Authority, 2015a). As central-place foragers, breeding sites represent core locations where sea lions return to rest between foraging trips. As such, the areas around breeding sites are continually traversed by animals departing and returning from foraging trips and represent areas with the greatest risk of bycatch from any fishing effort. Placing permanent gillnet fishing closures around all breeding sites greatly reduced the risks of bycatch from these high-density areas.

In contrast to permanent spatial closures, temporal closures were applied to areas open to gillnet fishing within the ASL Management Zone, with the intent to reduce sea lion bycatch mortality to within sustainable limits. An upper bycatch limit, or trigger limit was set for each of seven zones, as well as an overall bycatch limit. The temporal closure duration when a zone trigger limit was reached was originally set for the remainder of the fishing season (irrespective of when a trigger was reached), but in 2012 it was increased to 18 months (the breeding cycle duration of ASLs) (Australian Fisheries Management Authority, 2015a). AFMA determined that a bycatch rate of 1.5% of sea lion females per breeding cycle would be sufficiently precautionary and applied this to the total female population within each zone. However, noting the large number of small and genetically isolated breeding sites, and that for most (80% of ASL sites) a single female bycatch mortality would exceed the 1.5% limit (if applied to subpopulations), AFMA recognized the need to manage bycatch mortality at the subpopulation level and introduced more precautionary zone trigger limits (1–5 sea lions per zone) (Australian Fisheries Management Authority, 2015a). These arrangements came into effect in January 2012, reducing the overall annual trigger from 52 to 15 sea lions, and most zone triggers to just 1–2 sea lions per fishing season. Following the bycatch mortality of seven ASL over a 4-month period, three zones were closed by April 2012. This management action had immediate effect in markedly reducing gillnet fishing effort, driving the transition to alternate fishing gear, and reducing ASL bycatch mortality.

Gear Switching

Recognizing the impact on fishers from the large permanent and temporal gillnet closures on their capacity to catch their shark quota, AFMA provided incentives for fishers to switch fishing gear. This included the ability to fish with shark hooks (bottom-set longlines) inside both permanent and temporal gillnet closures and to do so with just 10% observer coverage (Australian Fisheries Management Authority, 2015a). In the decade following introduction of the Strategy, there was a progressive decline in gillnet fishing effort from within the ASL Management Zone and an increase in longline effort. By the 2020/21 fishing season, longlines accounted for 98% of the gummy shark catch within the ASL Management Zone, and catch levels were back to pre-management levels. Although nine interactions between seals (including fur seals) and longline fishers were recorded off SA between 2011 and 2018, only one was recorded as fatal (a fur seal) and in all three encounters with ASL the animals were reported alive.

Despite the apparent success of gear switching, it created significant challenges for fishers. Some continued to use gillnets in fishing grounds outside of the ASL Management Zone, including Bass Strait. Fishers who switched to longlines have faced increased costs associated with purchasing or modifying their vessels, setting up new gear, buying bait and hiring additional crew to assist with baiting and setting/hauling gear. Furthermore, between 2005 and 2015, an Australian Government voluntary fishing concession buyback scheme resulted in a 27% reduction in the number of vessels fishing for shark off SA (Australian Fisheries Management Authority, 2015a). The impacts and cost associated with adapting to changes from the ASL Management Strategy, likely influenced the decision of some fishers to take a buyback and exit the fishery. There have also been stock sustainability concerns that have arisen with the transition from a more selective (gillnet) to less selective (longline) fishing method, as well some interactions with seabirds (shearwaters and albatross) that AFMA continue to monitor (Knuckey et al., 2014; Australian Fisheries Management Authority, 2015a).

Implications of the Australian Sea Lion Management Strategy for Australian Sea Lion Recovery

Australian Fisheries Management Authority's ASL Management Strategy was implemented to address Wildlife Trade Operations (WTO) requirements set on the gillnet sector of the GHAT fishery under Part 13A of Australia's *EPBC Act*, with similar requirements now listed under Part 13 (Protected Species) (Department of Agriculture Water and the Environment, 2019). Recent assessment of the Southern and Eastern Scalefish and Shark Fishery, which the GHAT fishery forms part of, requires "AFMA to continue to: (a) maintain management measures clearly directed toward limiting the impact of fishing activity on Australian Sea Lions to levels which will assist in enabling the recovery of the species, including all subpopulations, and (b) monitor and review the adequacy of its Australian Sea Lion management measures, in consultation with marine mammal experts." Results from this study suggest that the Strategy has been highly successful in significantly reducing ASL bycatch mortality resulting from gillnet fishing, through a combination of measures that included a comprehensive observer program, permanent and temporal fishing closures with bycatch trigger limits, and incentives to switch gear that resulted in an almost total transition from gillnets to longlines in the ASL Management Zone. These measures have directly led to an estimated 98% reduction in ASL bycatch from gillnet interactions. There are potential residual risks from cryptic mortality that remain uncertain but given low gillnet fishing effort in the ASL Management Zone, these risks are likely to be low. Ongoing assessment of the level and spatial distribution of gillnet effort is warranted.

Although there has been comprehensive monitoring of how management measures have reduced the mortality of ASL through AFMA's electronic monitoring program (with 100% review of footage from gillnet fishers in the ASL Management Zone), there has been no systematic monitoring of changes in abundance of ASL subpopulations to assess if these management measures have enabled recovery of the "species, including all subpopulations." Some population monitoring has occurred since the introduction of the ASL Management Strategy, but it has largely been opportunistic (Goldsworthy et al., 2021). Although pup abundance trends from some affected subpopulations detailed in this study suggest that declines have slowed and possibly halted, consistent and long-term monitoring is required to demonstrate that the fishery is meeting its requirements under Part 13 of the *EPBC Act*.

A recent study suggested that the minimum time to detect at least a 5% increase in pup abundance, with modest recovery rates (∼1%/year), following a hypothetical management action to eliminate bycatch mortality was 6 years (Goldsworthy et al., 2020). This period is consistent with the age of recruitment in ASL, where most females have their first pup at age six (although some not until 10.5 years). However, the actual time required to detect a recovery is uncertain, and would be influenced by multiple factors including the frequency, timing, and precision of surveys, as well as seasonal and stochastic environmental factors that affect pup production within any breeding season and the underlying intrinsic growth rate (Goldsworthy et al., 2020). Taking these factors into account, it may take 1–2 decades to be confident that a recovery has or has not occurred. Given this long period, it is important for current management strategies that aim to limit fishery impacts on ASL to remain in place, and for systematic monitoring of ASL populations to be implemented and continued. The extent to which mitigation of gillnet fishery bycatch mortality by AFMA has addressed broader ASL conservation concerns for the population off SA, such as those detailed in the species recovery plan (Department of Sustainability Environment Water Population and Communities, 2013), can only be evaluated with further monitoring of populations.

There are two other sectors in Australia that manage demersal gillnet fisheries that potentially interact with ASL populations. The large mesh gillnet component of the Marine Scalefish Fishery managed by the SA Government, and the Temperate Demersal Gillnet and Demersal Longline Fishery managed by the Western Australian (WA) Government. The SA and WA Governments are also required to limit gillnet fishery impacts on sea lions under Part 13 of the *EPBC Act*, but they have done this differently to the Australian Government (AFMA). SA fishers are not permitted to target gummy or school shark, and instead use large-mesh gillnets to target whaler sharks (*Carcharhinus brachyurus* and *C. obscurus*). The gillnet catch is small (<6 t between 2013 and 2018) with longlines accounting for ∼90% of recent catch (Steer et al., 2020). To address potential interactions between the fishery and sea lions, the Department of Primary Industry and Regions SA introduced large-mesh net effort-triggers in 2016 that could invoke fishery area closures, but no trigger limits have been reached since implementation (Department of Primary Industries and Regions South Australia, 2019). There are presently no restrictions on using this gear type in any of the AFMA ASL permanent closures, many of which occur within SA State waters, and there is no independent observer program in the fishery to monitor interactions with sea lions.

In WA, the Temperate Demersal Gillnet and Demersal Longline Fishery operates in continental shelf waters off the south and lower west coasts and utilizes similar vessels and gear to that used in the gillnet sector of the GHAT fishery, with about 820 t of sharks and rays landed in 2017/18, mostly using gillnets (Braccini and Blay, 2020). In 2018, the WA Department of Primary Industries and Regional Development introduced gillnet exclusion zones (6–33 km radial gillnet closures) around 33 ASL breeding sites covering a total of 17,300 km^2 (Department of Primary Industries and Regional Development, 2021). However, given the absence of vessel monitoring systems (independent observer coverage or electronic monitoring), management measures to reduce sea lion bycatch in the areas fished (e.g., bycatch trigger limits) and monitoring of ASL populations, there is no capacity to assess the degree to which gillnet closures are reducing the incidence of sea lion bycatch or assess if their populations are recovering.

Based on our assessment, without sea lion bycatch trigger limits and a comprehensive independent monitoring program, the introduction of permanent gillnet closures on their own are unlikely to reduce the bycatch mortality of ASL to sustainable levels. As such, there is a very significant risk that the current management measures in WA are inadequate and are not preventing further declines in its ASL populations. The absence of baseline data on ASL populations off the south coast of WA and the need for ongoing monitoring has been recognized (Goldsworthy et al., 2021). There is an urgent need to introduce further mitigation and monitoring measures in WA demersal gillnet fisheries to limit their impacts on sea lion populations. Greater consistency and coordination in how each fishing sector limits the impact of gillnet fisheries on ASLs, meets its Part 13 requirements and monitors the effectiveness of management measures would improve the conservation outcomes for sea lions.

Global Implications

Globally, the bycatch of marine mammals in gillnet fisheries poses one of the most significant sources of anthropogenic mortality, and one of the most challenging to manage (Read et al., 2006; Reeves et al., 2013). With many fishing gear types, the bycatch of marine mammals can be reduced through gear modification and/or changes to fishing practices and behaviors, but these approaches have typically been less successful in gillnet fisheries (FAO, 2021). Efforts to reduce the incidence of bycatch in gillnet fisheries using acoustic deterrents or alerting devices have generally had limited success, with some notable exceptions (Dawson et al., 2013; Kratzer et al., 2021). The greatest success in reducing the bycatch of marine mammals in gillnet fisheries has come from management measures that either limit or restrict gillnet effort (through spatial and temporal closures) or remove it entirely by switching to alternate fishing methods (Berninsone et al., 2020; FAO, 2021). This approach was taken in the gillnet sector of the GHAT fishery off SA, where a combination of permanent and temporary spatial closures linked to bycatch trigger limits, and the switching from gillnets to longline fishing methods, has seen a marked reduction in sea lion bycatch mortality.

The assessment of the impact of bycatch mortality on ASL in gillnet fisheries off SA, followed by a science-informed adaptive management processes, has arguably set an important precedent both nationally and internationally. Not only has bycatch mortality been reduced to levels that should enable sea lion populations to recover, a decade on from the introduction of management measures fishing catches have returned to pre-management levels. In the context of managing marine mammal bycatch globally, it is an extraordinary outcome, and as such, provides an important case study which will hopefully demonstrate how measures could be applied elsewhere to effectively manage the leading source of anthropogenic mortality of marine mammals and other marine protected species.

DATA AVAILABILITY STATEMENT

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

ETHICS STATEMENT

The animal study was reviewed and approved by Primary Industries and Regions SA Animal Ethics Committee (Project 32-12).

AUTHOR CONTRIBUTIONS

SG, BP, DHa, and PS planned and designed the project. SG, BP, DHa, PS, AL, DC, SF, KP, RM, FB, AM, RK, and DHo collected or contributed data. SG, MH, PB, and FB analyzed data. SG wrote the manuscript. SB provided project support. All authors contributed to the article and approved the submitted version.

FUNDING

Support for this project was provided by the Fisheries Research and Development Corporation (FRDC project number 2007-041), the Department of Agriculture, Water and the Environment (DAWE) and the Department for the Environment and Water (DEW).

ACKNOWLEDGMENTS

We would like to thank the Fish Factory (L., K. and P. Toumazos) and skippers and crew for support of the bycatch observer programs, and Alex Ivey (SARDI) for assisting with observer work. We would also like to thank FRDC staff (especially Caroline Stewardson and Crispian Ashby), AFMA staff (especially Shane Gaddes, Beth Gibson, Trent Timmiss, Matt Daniel, Neil Hughes, Sally Weekes, Aaron Puckeridge, Lou Cathro), former members of GHATMAC, Shark RAG (especially Jeremy Prince, Steve McCormack and Ian Freeman), and Ian Knuckey (Fishwell Consulting) for support. Geert Aarts and Sophie Brasseur (IMARES, University of Wageningen, Holland) and Rick McGarvey (SARDI) provided modeling and statistical advice. We would also further like to thank Simon Clark (formerly DEW) for assistance with project development, especially in the Great Australian Bight.

SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2022.799102/full#supplementary-material

REFERENCES

Akçakaya, H. R. (1998). *RAMAS Metapop: Viability Analysis for Stage-Structured Metapopulations (Version 3.0)*. Setauket, NY: Applied Biomathematics.

Australian Fisheries Management Authority (2010). *Australian Sea Lion Management Strategy, Southern and Eastern Scalefish and Shark Fishery*. Available online at: https://fish.gov.au/Archived-Reports/2012/reports/Documents/AFMA_2010b.pdf (accessed September 15, 2021).

Australian Fisheries Management Authority (2015b). *School Shark (Galeorhinus galeus) Stock Rebuilding Strategy: Revised 2015*. Available online at: https://www.afma.gov.au/sites/default/files/uploads/2014/12/School-Shark-Rebuilding-Strategy.pdf (accessed September 15, 2021).

Australian Fisheries Management Authority (2015a). *Australian Sea Lion Management Strategy, Southern and Eastern Scalefish and Shark Fishery (Arrangements Effective From 1 May 2013; Version 2.0–Updated July 2015*. Available online at: http://www.afma.gov.au/wp-content/uploads/2014/03/Australian-Sea-Lion-Management-Strategy-2015-v2.0-FINAL.pdf (accessed September 15, 2021).

Berkson, J. M., and DeMaster, D. P. (1985). Use of pup counts in indexing population changes in pinnipeds. *Canad. J. Fisher. Aquat. Sci.* 42, 873–879. doi: 10.1139/f85-111

Berninsone, L. G., Bordino, P., Gnecco, M., Foutel, M., Mackay, A. I., and Werner, T. B. (2020). Switching gillnets to longlines: an alternative to mitigate the bycatch of Franciscana dolphins (*Pontoporia blainvillei*) in Argentina. *Front. Mar. Sci.* 7:699. doi: 10.3389/fmars.2020.00699

Braccini, M., and Blay, N. (2020). "Temperate demersal gillnet and demersal longline fishery fisheries resource status report 2019," in *Status Reports of the Fisheries and Aquatic Resources of Western Australia 2018/19: The State of the Fisheries*, eds D. J. Gaughan and K. Santoro (Kensington, WA: Department of Primary Industries and Regional Development, Western Australia).

Brownell, R. L., Reeves, R. R., Read, A. J., Smith, B. D., Thomas, P. O., Ralls, K., et al. (2019). Bycatch in gillnet fisheries threatens critically endangered small cetaceans and other aquatic megafauna. *Endang. Spec. Res.* 40, 285–296. doi: 10.3354/esr00994

Campbell, R. (2003). *Demography and Population Genetic Structure of the Australian Sea Lion, Neophoca cinerea*. Ph.D. thesis. Perth, WA: University of Western Australia.

Campbell, R. A., Gales, N. J., Lento, G. M., and Baker, C. S. (2008). Islands in the sea: extreme female natal site fidelity in the Australian sea lion, *Neophoca cinerea*. *Biol. Lett.* 4, 139–142. doi: 10.1098/rsbl.2007.0487

Cisneros-Mata, M. A., Delgado, J. A., and Rodríguez-Félix, D. (2021). Viability of the vaquita, *Phocoena sinus* (Cetacea: Phocoenidae) population, threatened by poaching of *Totoaba macdonaldi* (Perciformes: Sciaenidae). *Rev. Biol. Trop.* 69:589447. doi: 10.15517/rbt.v69i2.45475

Cosgrove, R., Gosch, M., Reid, D., Sheridan, M., Chopin, N., Jessopp, M., et al. (2016). Seal bycatch in gillnet and entangling net fisheries in Irish waters. *Fisher. Res.* 183, 192–199. doi: 10.1016/j.fishres.2016.06.007

Costa, D. P., and Gales, N. J. (2003). Energetics of a benthic diver: seasonal foraging ecology of the Australian sea lion, *Neophoca cinerea*. *Ecol. Monogr.* 73, 27–43. doi: 10.1890/0012-9615(2003)073[0027:eoabds]2.0.co;2

Dawson, S. M., Northridge, S., Waples, D., and Read, A. J. (2013). To ping or not to ping: the use of active acoustic devices in mitigating interactions between small cetaceans and gillnet fisheries. *Endang. Spec. Res.* 19, 201–221. doi: 10.3354/esr00464

Department of Agriculture Water and the Environment (2019). *Assessment Report for Southern and Eastern Scalefish and Shark Fishery – February 2019, February 2019. Commonwealth of Australia*. Available online at: https://www.environment.gov.au/system/files/pages/bbf3e30a-2fa3-45d6-b20c-7623b995cbcd/files/assessment-southern-eastern-scalefish-shark-fishery-2019.pdf (accessed September 15, 2021).

Department of Primary Industries and Regional Development (2021). *Western Australia's Temperate Demersal Gillnet and Demersal Longline Fishery. Submission to the Department of Agriculture, Water and the Environment Against the Guidelines for Ecologically Sustainable Management of Fisheries, April 2021*. Available online at: https://www.environment.gov.au/system/files/consultations/0e2fdc9b-0b3d-4bf8-894a-b23244f63d70/files/wa-temperate-shark-application-2021.pdf (accessed September 15, 2021).

Department of Primary Industries and Regions South Australia (2019). *Ecological assessment of the South Australian Marine Scalefish Fishery – Reassessment report prepared for the Department of the Environment and Energy*. Available online at: https://www.environment.gov.au/system/files/consultations/0135a6c3-6009-4b29-a56a-24e3c004baa0/files/ecological-assessment-south-australian-marine-scalefish-fishery-2019.pdf (accessed September 15, 2021).

Department of Sustainability Environment Water Population and Communities (2013). *Recovery Plan for the Australian Sea Lion (Neophoca cinerea), Commonwealth of Australia*. Canberra, ACT: Department of Sustainability Environment Water Population and Communities.

Emery, T. J., Noriega, R., Williams, A. J., and Larcombe, J. (2019). Changes in logbook reporting by commercial fishers following the implementation of electronic monitoring in Australian Commonwealth fisheries. *Mar. Policy* 104, 135–145. doi: 10.1016/j.marpol.2019.01.018

FAO (2021). *Fishing Operations. Guidelines to Prevent and Reduce Bycatch of Marine Mammals in Capture Fisheries. FAO Technical Guidelines for Responsible Fisheries No.1, Suppl. 4*. Rome: FAO. doi: 10.4060/cb2887en

Fowler, S. L., Costa, D. P., Arnould, J. P. Y., Gales, N. J., and Kuhn, C. E. (2006). Ontogeny of diving behaviour in the Australian sea lion: trials of adolescence in a late bloomer. *J. Anim. Ecol.* 75, 358–367. doi: 10.1111/j.1365-2656.2006.01055.x

Gilman, E., Suuronen, P., Hall, M., and Kennelly, S. (2013). Causes and methods to estimate cryptic sources of fishing mortality. *J. Fish Biol.* 83, 766–803. doi: 10.1111/jfb.12148

Goldsworthy, S. D. (2015). *Neophoca cinerea. The IUCN Red List of Threatened Species. Version 2015.2.* Available online at: www.iucnredlist.org (accessed September 15, 2021).

Goldsworthy, S. D., and Page, B. (2007). A risk-assessment approach to evaluating the significance of seal bycatch in two Australian fisheries. *Biol. Conserv.* 139, 269–285. doi: 10.1016/j.biocon.2007.07.010

Goldsworthy, S. D., Hodgson, J., and Holman, D. (2020). *Australian Sea Lion Investigations: 2018-19. SARDI Aquatic Sciences Pubblication No. F2020/000052–1, SARDI Research Report Series No. 1051*. Adelaide, SA, 88. Available online at: https://www.pir.sa.gov.au/__data/assets/pdf_file/0004/358285/Australian_sea_lion_investigations_2018-19.pdf (accessed September 15, 2021).

Goldsworthy, S. D., Page, B., Shaughnessy, P. D., and Linnane, A. (2010). *Mitigating Seal Interactions in the SRLF and the Gillnet Sector SESSF in South Australia. SARDI Aquatic Sciences Publication Number F2009/000613-1, SARDI Research Report Series Number 405*. Adelaide, SA, 213. Available online at: https://www.pir.sa.gov.au/__data/assets/pdf_file/0013/232006/No_405_Mitigating_Seal_Interactions_in_the_SRLF_and_the_Gillnet_Sector_SESSF_in_SA.pdf (accessed September 15, 2021).

Goldsworthy, S. D., Page, B., Shaughnessy, P. D., Hamer, D., Peters, K. D., McIntosh, R. R., et al. (2009). *Innovative Solutions for Aquaculture Planning and Management: Addressing Seal Interactions in the Finfish Aquaculture Industry. SARDI Publication Number F2008/000222-1, SARDI Research Report series No. 288*. Adelaide, SA, 287. Available online at: https://www.pir.sa.gov.au/__data/assets/pdf_file/0007/231793/No_288_Part_A_Innovative_solutions_for_aquaculture_planning_and_management_addressing_seal_interactions_in_the_finfish_aquaculture_industry.pdf and https://www.pir.sa.gov.au/__data/assets/pdf_file/0008/231794/No_288_Part_B_Innovative_solutions_for_aquaculture_planning_and_management_addressing_seal_interactions_in_the_finfish_aquaculture_industry.pdf (accessed September 15, 2021).

Goldsworthy, S. D., Shaughnessy, P. D., and Page, B. (2014). "Seals in Spencer Gulf," in *Natural History of Spencer Gulf*, eds S. A. Shepherd, S. Madigan, B. M. Gillanders, S. Murray-Jones, and D. Wiltshire (Adelaide, SA: Royal Society of South Australia), 254–265.

Goldsworthy, S. D., Shaughnessy, P. D., Mackay, A. I., Bailleul, F., Holman, D., Lowther, A. D., et al. (2021). Assessment of the status and trends in abundance of a coastal pinniped, the Australian sea lion *Neophoca cinerea*. *Endang. Spec. Res.* 44, 421–437. doi: 10.3354/esr01118

Gray, C. A., and Kennelly, S. J. (2018). Bycatches of endangered, threatened and protected species in marine fisheries. *Rev. Fish Biol. Fisher.* 28, 521–541. doi: 10.1007/s11160-018-9520-7

Gulland, F., Danil, K., Bolton, J., Ylitalo, G., Okrucky, R. S., Rebolledo, F., et al. (2020). Vaquitas (*Phocoena sinus*) continue to die from bycatch not pollutants. *Vet. Rec.* 20:187. doi: 10.1136/vr.105949

Hamer, D. J., Goldsworthy, S. D., Costa, D. P., Fowler, S. L., Page, B., and Sumner, M. D. (2013). The endangered Australian sea lion extensively overlaps with and regularly becomes by-catch in demersal shark gill-nets in South Australian shelf waters. *Biol. Conserv.* 157, 386–400. doi: 10.1016/j.biocon.2012.07.010

Hamer, D. J., Ward, T. M., Shaughnessy, P. D., and Clark, S. R. (2011). Assessing the effectiveness of the Great Australian Bight Marine Park in protecting the endangered Australian sea lion *Neophoca cinerea* from bycatch mortality in shark gillnets. *Endang. Spec. Res.* 14, 203–216. doi: 10.3354/esr00353

Hamilton, S., and Baker, G. B. (2019). Technical mitigation to reduce marine mammal bycatch and entanglement in commercial fishing gear: lessons learnt and future directions. *Rev. Fish Biol. Fisher.* 29, 223–247. doi: 10.1007/s11160-019-09550-6

Jounela, P., Sipila, T., Koskela, J., Tiilikainen, R., Auttila, M., Niemi, M., et al. (2019). Incidental bycatch mortality and fishing restrictions: impacts on juvenile survival in the Endangered Saimaa ringed seal *Pusa hispida saimensis*. *Endang. Spec. Res.* 38, 91–99. doi: 10.3354/esr00939

Kailola, P. J., Williams, M. J., Stewart, P. C., Reichelt, R. E., Mcnee, A., and Grieve, C. (1993). *Australian Fisheries Resources*. Canberra, ACT: Bureau of Resource Sciences, and the Fisheries Research and Development Corporation.

Kirkwood, R., and Goldsworthy, S. (2013). *Fur Seals and Sea Lions*. Clayton, VIC: CSIRO Publishing.

Knuckey, I., Ciconte, A., Koopman, M., Hudson, R., and Rogers, P. (2014). *Trials of Longlines to Target Gummy Shark in SESSF Waters off South Australia. FRDC Project 2011/068*. Queenscliff, VIC: Fishwell Consulting, 105.

Komoroske, L. M., and Lewison, R. L. (2015). Addressing fisheries bycatch in a changing world. *Front. Mar. Sci.* 2:83. doi: 10.3389/fmars.2015.00083

Kovacs, K. M., Aguilar, A., Aurioles, D., Burkanov, V., Campagna, C., Gales, N., et al. (2012). Global threats to pinnipeds. *Mar. Mam. Sci.* 28, 414–436. doi: 10.1111/j.1748-7692.2011.00479.x

Kratzer, I. M. F., Brooks, M. E., Bilgin, S., Ozdemir, S., Kindt-Larsen, L., Larsen, F., et al. (2021). Using acoustically visible gillnets to reduce bycatch of a small cetacean: first pilot trials in a commercial fishery. *Fisher. Res.* 21:243. doi: 10.1016/j.fishres.2021.106088

Larcombe, J., and McLoughlin, K. (2007). *Fishery Status Reports 2006: Status of Fish Stocks Managed by the Australian Government*. Canberra, ACT: Australian Government Department of Agriculture, Fisheries and Forestry, Bureau of Rural Sciences.

Lewison, R. L., Soykan, C. U., and Franklin, J. (2009). Mapping the bycatch seascape: multispecies and multi-scale spatial patterns of fisheries bycatch. *Ecol. Appl.* 19, 920–930. doi: 10.1890/08-0623.1

Lowther, A. D., and Goldsworthy, S. D. (2011). Detecting alternate foraging ecotypes in Australian sea lion (*Neophoca cinerea*) colonies using stable isotope analysis. *Mar. Mam. Sci.* 27, 567–586. doi: 10.1111/j.1748-7692.2010.00425.x

Lowther, A. D., Harcourt, R. G., Page, B., and Goldsworthy, S. D. (2013). Steady as he goes: at-sea movement of adult male Australian sea lions in a dynamic marine environment. *PLoS One* 8:e74348. doi: 10.1371/journal.pone.0074348

Lowther, A., and Goldsworthy, S. (2012). Head start: Australian sea lion pups gain experience of adult foraging grounds before weaning. *Mar. Biol.* 159, 2687–2696. doi: 10.1007/s00227-012-2026-2

Lowther, A., Harcourt, R. G., Hamer, D., and Goldsworthy, S. (2011). Creatures of habit: foraging habitat fidelity of adult female Australian sea lions. *Mar. Ecol. Progr. Ser.* 443, 249–263. doi: 10.3354/meps09392

Lowther, A., Harcourt, R., Goldsworthy, S., and Stow, A. (2012). Population structure of adult female Australian sea lions is driven by fine-scale foraging site fidelity. *Anim. Behav.* 83, 691–701. doi: 10.1016/j.anbehav.2011.12.015

Luck, C., Jessopp, M., Tully, O., Cosgrove, R., Rogan, E., and Cronin, M. (2020). Estimating protected species bycatch from limited observer coverage: a case study of seal bycatch in static net fisheries. *Glob. Ecol. Conserv.* 2:24. doi: 10.1016/j.gecco.2020.e01213

Machado, R., Ott, P. H., Moreno, I. B., Danilewicz, D., Tavares, M., Crespo, E. A., et al. (2016). Operational interactions between South American sea lions and gillnet fishing in southern Brazil. *Aquat. Conserv. Mar. Freshw. Ecosyst.* 26, 108–120. doi: 10.1002/aqc.2554

McConnell, B. J., Chambers, C., Nicholas, K. S., and Fedak, M. A. (1992). Satellite tracking of grey seals (*Halichoerus grypus*). *J. Zool.* 226, 271–282.

McIntosh, R. R. (2007). *Life History and Population Demograhics of the Australian Sea Lion.* Ph.D. thesis. Melbourne, VIC: La Trobe University.

Moore, J., Heinemann, D., Francis, T., Hammond, P., Long, K. J., Punt, A. E., et al. (2021). Estimating bycatch mortality for marine mammal stock assessment: concepts and best practices. *Front. Mar. Sci* 8:752356. doi: 10.3389/fmars.2021.752356

Owen, K., Sköld, M., and Carlström, J. (2021). An increase in detection rates of the critically endangered Baltic Proper harbor porpoise in Swedish waters in recent years. *Conserv. Sci. Pract.* 2:e468. doi: 10.1111/csp2.468

Page, B., Mckenzie, J., McIntosh, R., Baylis, A., Morrissey, A., Calvert, N., et al. (2004). Entanglement of Australian sea lions and New Zealand fur seals in lost fishing gear and other marine debris before and after government and industry attempts to reduce the problem. *Mar. Pollut. Bull.* 49, 33–42. doi: 10.1016/j.marpolbul.2004.01.006

Peltier, H., Authier, M., Caurant, F., Dabin, W., Daniel, P., Dars, C., et al. (2021). In the wrong place at the wrong time: identifying spatiotemporal co-occurrence of bycaught common dolphins and fisheries in the bay of biscay (NE Atlantic) From 2010 to 2019. *Front. Mar. Sci.* 8:617342. doi: 10.3389/fmars.2021.617342

Peltier, H., Authier, M., Deaville, R., Dabin, W., Jepson, P. D., Van Canneyt, O., et al. (2016). Small cetacean bycatch as estimated from stranding schemes: the common dolphin case in the northeast Atlantic. *Environ. Sci. Policy* 63, 7–18. doi: 10.1016/j.envsci.2016.05.004

Ramos, K. L., Machado, R., and Schiavetti, A. (2020). Operational interactions between sea lion species (Otariinae) and commercial fisheries. *Ethnobiol. Conserv.* 9:22. doi: 10.15451/ec2020-06-9.22-1-34

Read, A. J., Drinker, P., and Northridge, S. (2006). Bycatch of marine mammals in US and global fisheries. *Conserv. Biol.* 20, 163–169. doi: 10.1111/j.1523-1739.2006.00338.x

Reeves, R. R., Mcclellan, K., and Werner, T. B. (2013). Marine mammal bycatch in gillnet and other entangling net fisheries, 1990 to 2011. *Endang. Spec. Res.* 20, 71–97. doi: 10.3354/esr00481

Sequeira, A. M. M., Hays, G. C., Sims, D. W., Eguíluz, V. M., Rodríguez, J. P., Heupel, M. R., et al. (2019). Overhauling ocean spatial planning to improve marine megafauna conservation. *Front. Mar. Sci.* 6:639. doi: 10.3389/fmars.2019.00639

Shaughnessy, P., Kirkwood, R., Cawthorn, M., Kemper, C., and Pemberton, D. (2003). "Pinnipeds, cetaceans and fisheries in Australia: a review of operational interactions," in *Marine Mammals: Fisheries, Tourism and Management Issues*, eds N. Gales, M. Hindell, and R. Kirkwood (Clayton, VIC: CSIRO Publishing), 136–152.

Sims, M., Cox, T., and Lewison, R. (2008). Modeling spatial patterns in fisheries bycatch: Improving bycatch maps to aid fisheries management. *Ecol. Appl.* 18, 649–661. doi: 10.1890/07-0685.1

Steer, M. A., Fowler, A. J., Rogers, P. J., Bailleul, F., Earl, J., Matthews, D., et al. (2020). *Assessment of the South Australian Marine Scalefish Fishery in 2018. Report to PIRSA Fisheries and Aquaculture. South Australian Research and Development Institute (Aquatic Sciences), Adelaide. SARDI Publication No. F2017/000427-3. SARDI Research Report Series No. 1049*. 214. Available online at: https://www.pir.sa.gov.au/__data/assets/pdf_file/0004/360589/Assessment_of_the_South_Australian_Marine_Scalefish_Fishery_in_2018._Report_to_PIRSA_Fisheries_and_Aquaculture.pdf (accessed September 15, 2021).

Sterling, J., and Ream, R. R. (2004). At-sea behavior of juvenile male northern fur seals (*Callorhinus ursinus*). *Canad. J. Zool.* 82, 1621–1637. doi: 10.1139/z04-136

Thorne, L. H., Baird, R. W., Webster, D. L., Stepanuk, J. E., and Read, A. J. (2019). Predicting fisheries bycatch: a case study and field test for pilot whales in a pelagic longline fishery. *Diver. Distribut.* 25, 909–923. doi: 10.1111/ddi.12912

White, G. C., and Burnham, K. P. (1999). Program MARK: survival estimation from populations of marked animals. *Bird Study* 46, 120–139. doi: 10.1080/00063659909477239

Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.