

Biological Conservation

Estimates and drivers of protected species bycatch in the California set gillnet fishery

--Manuscript Draft--

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Abstract:	The identification of efficient management strategies that reduce protected species bycatch while also minimizing impacts on fishing livelihoods is a global conservation challenge. Identifying such strategies requires understanding levels of bycatch relative to management targets as well as the relationship between bycatch risk and potential management levers. In this study, we use ratio estimators to reconstruct bycatch of select marine mammal and seabird species in the California ≥3.5" set gillnet fishery from 1981-2022 and random forest models to identify drivers and hotspots of bycatch risk. We find that bycatch has dropped precipitously since the 1980s as a result of management, but at significant costs to fisheries participation and revenues. Recent marine mammal bycatch ranges from 0.1% to 4.0% of the potential biological removal and marine mammal populations are recovering. Spatial-temporal drivers of bycatch risk were more important than fishing-related drivers of risk, suggesting that spatial-temporal closures would be more effective than mesh size or soak time restrictions at limiting bycatch. For each species, we identified 2-5 hotspots of elevated bycatch risk as candidates for temporary seasonal closures. Bycatch risk for harbor seal (<i>Phoca vitulina</i>) and California sea lion (<i>Zalophus californianus</i>), the species with the greatest bycatch risk, is especially high from April 1st to June 15th, suggesting that hotspot closures during this 2.5-month time period could be particularly efficient. Our study also highlights the value of competing multiple modeling approaches to identify methods that best predict rare bycatch events.
Response to Reviewers:	Please see the attached response to reviewers document

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February 20, 2025

Dear Dr. Bates:

We are grateful for the opportunity to submit our revised manuscript, “Estimates and drivers of protected species bycatch in the California set gillnet fishery”, for consideration as an Article in *Biological Conservation*.

We carefully reviewed the comments from you and the reviewers and appreciate this thoughtful feedback. We address each comment individually below with the original comment shown in black text and the response shown in indented blue text. This feedback and the associated revisions have greatly improved the manuscript.

Briefly, we made the following notable changes to the manuscript text:

1. We added an “Overview” paragraph at the top of the methods to better orient the reader to the study objectives, approach, and a few key choices.
2. We added a paragraph before introducing the logbook and observer datasets to orient the reader to their differences and their role in the analysis.
3. We moved the methods and results text describing the estimation of bycatch used the random forest models to the supplemental information to reduce confusion.
4. We substantially edited the paper to clarify how our analysis of drivers of bycatch can help to refine bycatch management to more efficiently meet bycatch objectives while minimizing impacts on fishing opportunities. In particular, we better explained why spatial-temporal hotspot closures could lead to improved efficiency.
5. We clarified why we used a random forest modeling approach and how we selected and categorized the variables assessed as bycatch drivers using this approach.

Thank you for your consideration and please let us know if you have any questions.

On behalf of all authors,
Sincerely,

Yutian Fang

Reviewer #1

This is a comprehensive and well-written manuscript on an important topic of relevance to the readership of Biological Conservation.

I liked the comprehensive multi-species, multi-order approach. Most by-catch papers are inappropriately taxonomically restricted.

We are grateful for your close review of our manuscript, for this kind acknowledgement, and for your thoughtful and constructive feedback. We agree with all of your comments and suggestions and detail how we addressed them in the indented blue text below.

I have only a few minor suggestions to improve the manuscript:

1. The authors make a large number of assumptions scattered through the methods. I suggest adding a table of assumptions, an assessment of the likelihood of each being correct and a comment on the implications for the results if each assumption is not correct to the Supplementary Materials

We added a new table to the supplement (**Table S4**) to detail the assumptions made in the analysis and their implications for the results. A snapshot is shown below.

Table SX. Assumptions made throughout the analysis, their likelihood, and their potential impact on the results (*assumptions that are expected to be valid or true on average are not likely to impact the results).

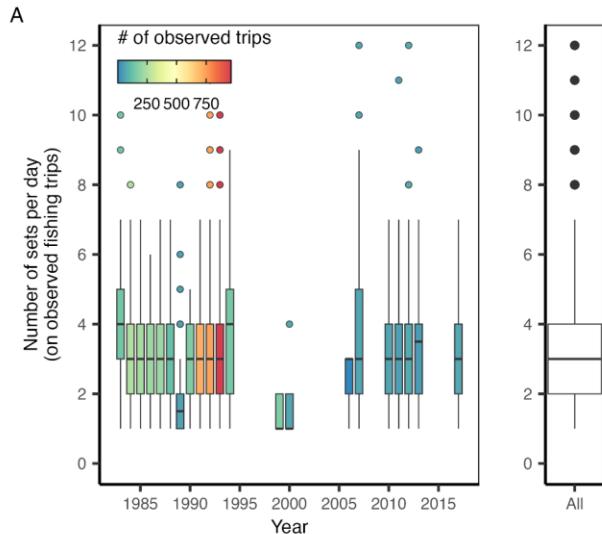
Assumption	Likelihood and potential impact of assumption
<i>Data analysis</i>	
<i>Ratio estimation</i>	
1. Ratio estimation assumes that the rate of bycatch for observed fishing trips equals the rate for all fishing trips in a given stratum.	This assumption appears valid as the traits of observed trips are representative of the unobserved trips (Fig. S7) and we stratified our analysis using the 7-region scheme recommended in other studies (Fig. S8).
2. We assumed that the bycatch rate in years without observer data was equal to the bycatch rate in the closest year with data.	The validity of this assumption is difficult to assess given the inconsistency in the observer program (Fig. 2BC); however, it is this inconsistency that also makes the assumption necessary. Ultimately, our finding that recent bycatch is low is not likely sensitive to this assumption, as the effects of substantial decreases in fishing effort overwhelm the effects of changes in bycatch rates.
<i>Bycatch estimation using the random for models*</i>	
1. We assumed that a “pseudo-set” (roughly equivalent to a trip) is equivalent to 3 sets (Fig. S4AB).	This assumption is likely to be true on average since it reflects the median value, which has been consistent through time (Fig. S4AB).
2. We assumed the number of captures per set is the	This assumption is likely to be true on average since it reflects

2. Moving Table 1 to the Supplementary Materials to reduce the number of Tables and Figures in the main paper

We moved **Table 1** to the Supplementary Materials (now **Table S1**) and updated table numbers throughout the manuscript.

3. Clarifying whether all the trips were the same length and involved the same number of sets. If not, does that affect the validity of the analyses?

We appreciate the question and refer the reviewer to **Fig. S4A** (provided below for ease), which shows that there is a consistent median of three sets per observed trip over time. The level of variability around this central measure is also consistent over time.



The validity of the ratio estimation analysis, which treats trips and not sets as the sampling units, would be compromised if there were temporal trends in these values; fortunately, the lack of trend means that the analysis is appropriate and valid. We added the following underlined sentence to the methods to clarify this point:

"We used trips rather than sets as the sampling unit given the inability to identify unique sets in the logbook data (**Fig. S4B**). This is valid because the number of gillnet sets per fishing trip (median: 3 sets/trip; IQR: 2-4 sets/trip) has been consistent through time (**Fig. S4A**)."

This is also true for the estimation of bycatch using the random forest models, where we assumed that the average number of sets per trip is time invariant. This assumption is supported by **Fig. S4A**, which was referenced in this methods sentence:

"We summed the number of pseudo-sets predicted to have bycatch each year, converted this sum to "true sets" assuming three sets per pseudo-set (**Fig. S4AB**), and multiplied this sum by the median number of captures when a capture occurs to generate estimates of the total number of captured animals (**Fig. S4C**)."

The evaluation of drivers of bycatch through the random forest analysis is not sensitive to this variability because the sampling unit for this analysis is gillnet sets not trips.

4. Clarifying how you selected the attributes for inclusion in the forest models.

We added the following underlined text to describe how we selected these variables:

"We considered nine attributes of fishing as potential drivers of bycatch risk: haul depth (fathoms), mesh size (inches), soak time (hours), latitude ($^{\circ}$ N), longitude ($^{\circ}$ W), distance from shore (km), Julian day, sea surface temperature ($^{\circ}$ C), and whether the fishing occurs near an island (i.e., within 10 km of island coast).
These attributes were selected based on their demonstrated relationship to bycatch risk in other papers (e.g., Bettoli et al. 2006, Bjørge et al. 2013, Kroetz et al. 2020) and their availability in the observer data or their ability to be derived through remote sensing (i.e., distance from shore, temperature, island area).
They represent a range of spatial (latitude, longitude, distance from shore, depth, island area), temporal (Julian day), environmental (temperature), and fishing-related (soak time, mesh size) attributes."

- Kroetz et al. 2020: region, latitude, season, sea surface temperature, depth, net depth, net length, mesh size, gear type, target, and soak time
- Bettoli et al. 2006: mesh size, fish length, water temperature, soak time, and twine type
- Bjørge et al. 2013: Season, areas, depth, soak time
- Kroetz, A.M., Mathers, A.N. and Carlson, J.K., 2020. Evaluating protected species bycatch in the US Southeast Gillnet Fishery. *Fisheries research*, 228, p.105573.
- Bettoli, P.W. and Scholten, G.D., 2006. Bycatch rates and initial mortality of paddlefish in a commercial gillnet fishery. *Fisheries Research*, 77(3), pp.343-347.
- Bjørge, A., Skern-Mauritzen, M. and Rossman, M.C., 2013. Estimated bycatch of harbour porpoise (*Phocoena phocoena*) in two coastal gillnet fisheries in Norway, 2006–2008. Mitigation and implications for conservation. *Biological Conservation*, 161, pp.164-173.

5. Checking the use of 'only' which I think is misplaced in several sentences.

We deleted "only" in two places in the manuscript where it was used more as a colorful adjective stressing a small value than as an expression of a precise or single value:

"Since then, participation has continued to decline, with ~~only~~ ~40 vessels active in 2022, and the vast majority (>90%) of landings coming from just 13 vessels (CDFW, 2023) (**Fig. 1B**)."

"Fleetwide revenues decreased from US\$15 million in 1987 to ~~only~~ US\$1 million in 2022 (**Fig. 1C**; both values in 2022 dollars)."

Reviewer #2

Bycatch estimation and the identification of factors affecting bycatch are essential for developing effective management measures to reduce bycatch. The aim of the present study was to estimate bycatch in the California set gillnet fishery and identify hotspot areas of bycatch. Despite the relevance of this work, the article requires significant improvement. The main aspects that need attention include a clearer explanation of the dataset used and the statistical methods employed for bycatch estimation. Additionally, a restructuring of all sections is necessary to enhance the overall clarity and flow of the manuscript.

We are grateful for your close review of our manuscript and for your thoughtful and constructive feedback. We agree with all of your comments and suggestions and detail how we addressed them in the indented blue text below.

Above, I offer the following comments as suggestions to further improve the manuscript:

Introduction

Lines 33-37: The authors highlight that management disruptions can have serious social, cultural, and economic implications. This point is reiterated in another part of the introduction, leading the reader to believe that the present study will propose a management approach that addresses both ecological and social concerns. However, this was not actually presented in the article. I suggest that the authors reconsider how they introduce this topic.

In response to many of the comments below, we revised the paper to clarify how our analysis of drivers of bycatch can be used to design more efficient management that expands fishing opportunities while keeping bycatch below management targets. We hope that these revisions better show how our paper can support management that addresses both ecological and social concerns. In particular, see the changes we made in response to comments about the spatial-temporal hotspot closures.

Lines 35-37: I am not sure that fishery bycatch, particularly in gillnet fisheries, constitutes an economic constraint for the fishing industry. Bycatch becomes a problem when fishing effort needs to be reduced to mitigate bycatch levels. A better contextualization of this issue is needed here.

We replaced the word “impacts” with “consequences” to better connect this sentence to the previous two sentences, which articulate that bycatch becomes a problem when bycatch management reduces fishing opportunities:

“As a result, many countries have established strict mandates to limit bycatch of vulnerable species, which can result in fisheries closures and other severe restrictions (Crowder and Murawski, 1998; Senko et al., 2014). These

management disruptions can have serious social, cultural, and economic impacts on fishing communities (Smith et al., 2020). Due to the negative ecological, economic, and social consequences of fishery bycatch, bycatch avoidance is an important objective for global fishery management.”

The cited papers all show that bycatch management can result in reduced fishing opportunities that result in economic impacts. Further in the introduction, we show how historical bycatch management of the California set gillnet fishery has dramatically reduced fishing opportunities and revenues.

Lines 64-66: To predict bycatch on unobserved trips, the bycatch estimates from observed trips must be representative, even when using a model-based approach.

This is true. If observed trips are not representative of unobserved trips, a model risks being asked to make out-of-sample (OOS) predictions. We added the following underlined text to soften the unintended implication that model-based approaches are not also vulnerable to the challenges faced by ratio estimation:

“Model-based approaches, which use statistical models to estimate bycatch, can overcome many of these limitations by incorporating a wider suite of covariates and allowing for non-linear impacts and are generally thought to produce better bycatch estimates (Stock et al., 2019).”

We opt not to make more drastic changes given our confirmation that, in our case, the observed trips were representative of the unobserved trips (see **Fig. S7**).

Lines 100-111: A drastic reduction in fishing effort, as well as bycatch, was observed in the study area. This reduction corresponds to a 15-fold decrease in fishing days and a 10-fold decrease in fleet size, representing a significant decline in fishing effort. However, despite these decreases, the authors highlight ongoing calls for further reductions. To support additional reductions in fishing effort, it is important to present evidence, including examples, demonstrating the real need for further limiting fishing activities.

We added the following underlined text to clarify that the ongoing calls are from one of the U.S.’s more agenda-based conservation groups (Oceana). As a result, these calls are less based on empirical scientific evidence and more on organizational mission. Our paper serves to provide the scientific basis for either refuting or supporting Oceana’s claims that bycatch is a significant problem in the fishery. After conducting the research, we found no support for Oceana’s claims, as current bycatch mortality is below NOAA’s “zero mortality rate goal” (ZMRG). To avoid being overly provocative, we are intentionally vague about the specific conservation group, though both of the cited documents are Oceana lobbying materials.

"Despite declining fishing effort and bycatch, conservation groups are lobbying for additional restrictions, including permanent closure, to further avoid bycatch (Birch et al., 2023; Birch and Shester, 2023). There is thus great need for scientific guidance on management regulations that are likely to provide conservation benefits while also avoiding unnecessary burdens on the fishing industry."

In addition, I suggested that the authors reconsider and restructure the introduction. Below, I provide an example of a potential revised structure and emphasize the need for a more detailed statistical explanation regarding bycatch estimation and bycatch predictors:

- 1- Fishery bycatch concerns and management actions to reduce bycatch.
- 2- Description of the management actions implemented to reduce bycatch in the California set gillnet fishery, along with the consequent reduction in fishing effort in the region.
- 3- Despite the reduction in fishery bycatch, there is substantial evidence suggesting that fishing remains a growing impact.
- 4- To guide effective bycatch reduction policies, it is crucial to understand the magnitude of both historical and recent bycatch, as well as the key drivers of bycatch in fisheries.
- 5- Explanation of the models used: Why choose ratio estimation over model-based predictions (e.g., regression models)? Unlike ratio estimators, which are commonly used to extrapolate bycatch estimates by multiplying an observed bycatch rate by the total fishing effort, model-based predictions offer greater efficiency and precision by incorporating a statistical model of bycatch. Additionally, why choose Random Forest over regression models? Regression models can provide both bycatch predictions and identify predictors simultaneously. Please explain the advantages of using different approaches for estimating bycatch and identifying predictors.
- 6- Objectives.

We took several steps to address this comment.

We incorporated the requested information -- i.e., (1) why we used both design- and model-based approaches to estimate bycatch and (2) why we used random forests, a machine learning approach, instead of classical regression approaches like GLMs or GAMs to evaluate predictors of bycatch -- into the methods section. We opted to put this information in the methods rather than in the introduction to keep the introduction broadly focused on the roles and tradeoffs of the approaches and the methods focused on the technical details and decisions. This information was added to a new methods "Overview" section that address both points, a rewrite of the methods section on estimating bycatch using the random forests that explains both the reason for and problems with this approach (point #1), and small changes to the random forests section to articulate why random forests were used instead of classical regression techniques (point #2).

We also added the following underlined text to the introduction to better introduce the types of classical regression techniques and machine learning techniques that can be used to both estimate bycatch and evaluate drivers:

"Model-based approaches, which use either statistical (e.g., generalized linear models, generalized additive models) or machine learning (e.g., random forests, boosted regression trees) models to estimate bycatch, can overcome many of these limitations by incorporating a wider suite of covariates and allowing for non-linear impacts and are generally thought to produce better bycatch estimates (Stock et al., 2019). Additionally, these approaches can support management by identifying drivers of bycatch risk and by predicting detailed hotspots of risk (Long et al., 2024; Lopez et al., 2024; Stock et al., 2019)."

We note that this text previously explained that statistical models can both estimate bycatch and evaluate drivers.

We appreciate the suggested outline for a potential restructured introduction but opted not to restructure the introduction because (1) the current structure is fairly close to the suggested structure and (2) the current structure better flows from broad (general bycatch issues) to narrow (specific case study). The current structure is as follows:

1. Fishery bycatch concerns and impacts on fishing communities
2. Bycatch management actions and the information needed to guide them
3. Approaches for quantifying the scale/drivers of bycatch to guide management
4. Introduction of the study fishery and its bycatch concerns
5. History of bycatch management in the study fishery
6. Consequences of management on communities and need for greater efficiency
7. Study objectives

The current introduction starts with 3 broad paragraphs followed by 4 narrow paragraphs. The suggested introduction goes 1 broad paragraph then 2 specific paragraphs then 2 broad paragraphs then 1 specific paragraph.

Methods

I missed the "Study Area" subsection. I agree that describing the area in terms of oceanographic conditions and productivity is crucial for a better understanding of the spatial and temporal patterns of both fishery activity and the target bycatch species.

We added a new "Study area" subsection that briefly introduces the section of California coast evaluated in the study and its oceanography, fisheries, and protected species. The new subsection reads as follows:

"Our study area spans southern and central California, U.S.A. from the U.S.-Mexico border (32°N) to Point Reyes (38°N), which is just north of San Francisco Bay (**Fig. 1A**). The study area falls within the California Current Ecosystem, a highly productive eastern boundary upwelling system spanning from the southern

tip of the Baja Peninsula, Mexico (28°N) to the U.S.-Canada border (48°N). Seasonal upwelling of cold, nutrient-rich water fuels populations of krill, squid, sardines, and other low trophic species that are fed upon by larger fishes, sea turtles, seabirds, and marine mammals. The ecosystem also supports important commercial and recreational fisheries and other human uses that benefit tens of millions of people living along the U.S. West Coast. We assessed bycatch within seven regions of the study area, which are shown in **Fig. 1A** and explained in detail in section 2.5.”

Moreover, the subsections about the fishery dataset are confusing. It is unclear whether the authors used two data sources: one from observer programs and another from logbooks. Is the logbook dataset merely complementary information, or does it represent new fishing sets? Additionally, these subsections are lengthy and repetitive, and should be condensed.

We added a new “Overview” subsection to the methods where we now provide an overview of both the analytical approach and the datasets used to support this approach.

This new “Overview” subsection introduces the differences between the logbook and observer datasets and explains their role in the analysis. Briefly, all gillnet vessels are required to submit logbooks that describe ***all fishing trips*** and retained catch. Logbooks do not contain information about protected species bycatch but they do document all fishing trips for which we then predict the bycatch associated with the trip. Observers are trained scientists placed on ***a sample of all fishing trips*** that collect information about discarded catch including catch of protected species. The random forest models are trained on the observer data. The ratio estimators use bycatch rates on observed trips (observer data) to estimate bycatch for all trips (logbook data).

We also added a new opening paragraph to the “Dataset” section to explain the differences and role of the two datasets before they are introduced in detail:

“Our analysis relies on two fisheries-dependent datasets: logbook data and observer data. All gillnet vessels are required to submit logbooks documenting when, where, and how they fished and how much catch was retained. Thus, logbooks characterize all fishing trips. However, because logbooks are self-reported and reported discards are unverifiable, logbooks likely underreport discarded bycatch, especially the bycatch of protected species. As a result, information from observer programs, which place trained observers on a sample of fishing trips (0-16.7% in this fishery; **Fig. 2B**), are required to inform estimates of bycatch for the unobserved trips recorded in vessel logbooks, which constitute the majority of fishing effort (83.3-100% in this fishery; **Fig. 2B**). Thus, the estimation of total bycatch through ratio estimation depends on both the observer and logbook data. In contrast, the evaluation of bycatch drivers and hotspots with the random forest models uses only the observer data, as these are the only data

to accurately record protected species bycatch when it occurs. These datasets are described in detail below.”

We reduced the length and repetitiveness of the dataset subsections by moving the details on how missing values were imputed to the supplemental methods. We added the following two sentences to refer the reader to this text:

“We developed a series of simple assumptions to impute missing values for a few key variables (GPS coordinates, fishing depth, soak hour, mesh size) used to describe gillnet sets documented in the observer data (**Fig. S3**; see supplemental methods for details).”

“We developed a series of simple assumptions to impute missing or unrealistic values for a few key variables (fishing depth, soak hour, mesh size) used to describe gillnet pseudo-sets documented in the logbook data (**Fig. S5 & S6**; see supplemental methods for details).”

Lines 149-154: It is not necessary to explain how the fishing sets were identified in the federal and state datasets. Furthermore, the distinction between the state and federal datasets is not clearly explained.

We removed the text explaining how unique identifiers for fishing sets in the observer data were constructed. By deleting this text, we also removed the only reference to the “state” and “federal” observer data, thereby preventing the need to clarify this distinction. For reference, CDFW is a state agency and NOAA is a federal agency; this nomenclature was originally intended to distinguish between the data collected by the state-run (CDFW) and federally-run (NOAA) observer programs.

Lines 157-160: I don't think it's necessary to explain that some pieces of the dataset were digitized from sheets. Simply stating the origin of the data should suffice.

We replaced the word “digitized” with the word “recovered” in the following sentence:

“We recovered a small portion of the missing raw data – observations from Monterey Bay from 1987-1989 (**Fig. 2**) – from original CDFW data sheets that were given to a colleague at the Southwest Fisheries Science Center during the late 1990s for a reanalysis of historic bycatch rates in that region (Forney et al., 2001).”

This is helpful because it better aligns the text with the language used in **Fig. 2**.

Line 164: What missing values are being referred to? Are the authors referring to a fishing net set?

We added the following underlined text to clarify that we imputed missing values that describe traits of observed or logged gillnet sets:

“We developed a series of simple assumptions to impute missing values for a few key variables (GPS coordinates, fishing depth, soak hour, mesh size) used to describe gillnet sets documented in the observer data (**Fig. S3**; see supplemental methods for details).”

“We developed a series of simple assumptions to impute missing or unrealistic values for a few key variables (fishing depth, soak hour, mesh size) used to describe gillnet pseudo-sets documented in the logbook data (**Fig. S5 & S6**; see supplemental methods for details).”

Lines 167-168: What does 'block' mean? This term is introduced for the first time, so a definition is required. Additionally, the authors state that if week-level information is missing, they use month-level data, and if month-level data is unavailable, they use year-level data to determine the position of the fishing net set. Is this correct? If so, it seems complicated to impute missing values based on monthly or yearly averages. I suggesting excluding fishing net sets where no week information is available, as this could lead to more accurate data.

In California, fishing catch and effort is reported to 10x10 minute (~18x18 km) statistical reporting blocks, which we refer to as “blocks” for brevity. We added a definition of these statistical blocks at first use and in the caption of **Fig. 1**, where the blocks are mapped.

First use: “These data describe vessel information (vessel name, unique identifier, permit number); when (date), where (statistical reporting “block”; Fig. 1A), and how long (hours) a vessel fished;”

Fig. 1: “Panel A shows the spatial history of fishing effort during four regulatory periods. Trips are reported by the 10 x 10 minute (~18 x 18 km) statistical blocks used for fisheries catch and effort reporting. The horizontal lines delineate geographical strata used in the ratio estimation analysis; strata are labeled in the first plot.”

Line 183: The distinction between the logbook and observer coverage datasets is unclear. Both datasets cover the same period (1983-2017). Does the information obtained from logbooks complement the observer data, or does it represent new trips that were not monitored by the observer program? Clarification on the relationship between these two datasets is needed.

We added a new “Overview” subsection to the methods where we now provide an overview of both the analytical approach and the datasets used to support this approach.

We also added a new opening paragraph to the “Dataset” section to explain the differences and role of the two datasets before they are introduced in detail.

See our response to the comment where this point is first made and addressed above.

Line 227: What characteristics were evaluated?

We added the following underlined text to list the evaluated characteristics, which we note are also all shown in Fig. S7:

“This assumption requires that the characteristics of observed trips do not systematically differ from the characteristics of all trips, which was confirmed by a two-sided Kolmogorov-Smirnov test for six key traits (i.e., day of year, depth, latitude, mesh size, distance from shore, soak time) (**Fig. S7**).”

Line 228: The unit used in the article was fishing trips rather than fishing net sets. However, for some imputed data obtained from historical records, the unit was fishing net sets, as shown in Table 1. It is not clear how the authors standardized all the information to the same unit.

We took several actions to address this comment.

First, we added the following underlined text to the methods to clarify that we converted set-level bycatch rates from historical reports to trip-level bycatch rates assuming 3 sets per trip, as indicated by the observer data:

“We extracted summaries of set-level bycatch rates from historical reports (**Table S1**) for years and regions missing raw data to support the ratio estimation analysis. We converted set-level bycatch rates to trip-level bycatch rates assuming an average of 3 sets per trip (Table S3), as indicated by the observer data (Fig. S4).”

Second, we added a new supplemental table (**Table S3**) to explicitly provide the bycatch rates extracted from historical reports. A snippet is shown below:

1029 Table S3. Bycatch rates extracted from historical reports (* mark years with raw observer data where the
1030 summary values from the historical reports are not needed).

Reference	Species	Region	Year	Estimated #			Catch per trip	Raw data available?
				# sets observed	trips observed	Catch		
Hanan et al. 1988	California sea lion	Channel Islands	1985	180	60.00	44	0.733	*
Hanan & Diamond 1989	California sea lion	Channel Islands	1986	66	22.00	54	2.455	*
Hanan et al. 1988	California sea lion	Monterey Bay	1983	22	7.33	12	1.636	
Hanan et al. 1988	California sea lion	Monterey Bay	1984	126	42.00	19	0.452	
Hanan et al. 1988	California sea lion	Monterey Bay	1985	49	16.33	5	0.306	
Hanan & Diamond 1989	California sea lion	Monterey Bay	1986	36	12.00	4	0.333	
Hanan et al. 1988	California sea lion	Morro Bay	1983	288	96.00	41	0.427	
Hanan et al. 1988	California sea lion	Morro Bay	1984	374	124.67	22	0.176	
Hanan et al. 1988	California sea lion	Morro Bay	1985	317	105.67	25	0.237	
Hanan & Diamond 1989	California sea lion	Morro Bay	1986	137	45.67	13	0.285	*
Hanan et al. 1988	California sea lion	San Francisco	1983	158	52.67	4	0.076	
Hanan et al. 1988	California sea lion	San Francisco	1984	300	100.00	8	0.080	
Hanan et al. 1988	California sea lion	San Francisco	1985	348	116.00	3	0.026	
Hanan & Diamond 1989	California sea lion	San Francisco	1986	419	139.67	2	0.014	
Hanan et al. 1988	California sea lion	Southern California	1983	430	143.33	16	0.112	*
Hanan et al. 1988	California sea lion	Southern California	1984	571	190.33	13	0.068	*
Hanan et al. 1988	California sea lion	Southern California	1985	339	113.00	5	0.044	*
Hanan & Diamond 1989	California sea lion	Southern California	1986	425	141.67	15	0.106	*
Hanan et al. 1988	California sea lion	Ventura	1983	430	143.33	16	0.112	*

Finally, we modified **Figures 2 and S9** to explicitly mark values that were extracted from the historical reports. See the revised figures in a response to a comment further below.

Line 230: What do the authors mean by "stratum i"? Is it related to the "seven-region" division? If so, I suggest modifying the text as follows: "Where $k_{s,i}$ is the total number of individuals of species s captured in observed trips occurring in each region i and...". Additionally, I recommend that the authors include a "Study Area" subsection and describe how the study area was divided into different regions.

We added the following underlined text to tease up that, in this case, strata are defined by years and regions:

"Under this approach, the bycatch rate for species s in stratum i ($r_{s,i}$) – where, in this case, strata are defined by years and regions (see next paragraph) – is thus calculated as:"

The next paragraph explains the derivation of the regional strata.

Line 234: What is the number of trips observed and the number of trips not observed?

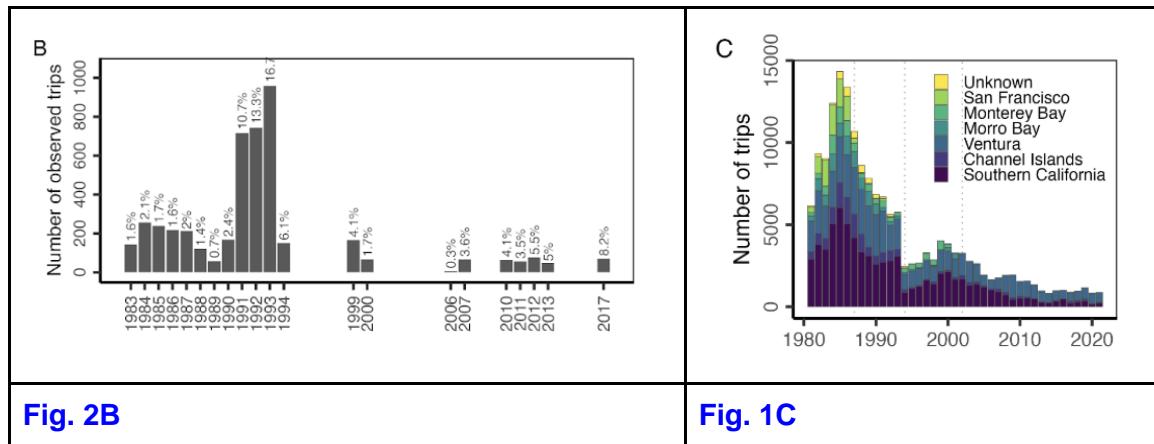
We added a new supplemental table (**Table S2**) to provide the number of total (observed+unobserved) trips, observed trips, and unobserved trips per year.

1026 Table S2. Number of fishing trips in the California 3.5 inch mesh set gillnet fishery by year.

1027

Year	# of vessels	# of trips		Percent of trips	
		Total	Observed	Unobserved	Observed
1981	162	6139	0	6139	0.0%
1982	222	9218	0	9218	0.0%
1983	262	9012	143	8869	1.6%
1984	357	12374	255	12119	2.1%
1985	378	14314	238	14076	1.7%
1986	379	13336	217	13119	1.6%
1987	290	10667	213	10454	2.0%
1988	211	8585	120	8465	1.4%
1989	198	7811	58	7753	0.7%
1990	182	6836	167	6669	2.4%
1991	158	6668	716	5952	10.7%
1992	117	5611	744	4867	13.3%
1993	115	5754	959	4795	16.7%
1994	74	2455	150	2305	6.1%
1995	70	2616	0	2616	0.0%
1996	59	2654	0	2654	0.0%
1997	66	3310	0	3310	0.0%

We also note that this information can be visually assessed from **Fig. 2B**, which already plots the number and percentage of trips that are observed trips. **Fig. 1C** provides additional helpful context because it shows the total number of trips over time.



We added the following sentence to the “Observer data” subsection to highlight this new table and better highlight the existing figures showing this information:

“The percentage of fishing trips with onboard observers has varied over time, ranging from 0.3% of trips in 2006 to 16.7% of trips in 1993 (**Fig. 2B; Table S2**).”

Line 241: Did the authors assess the logbooks of all boats? This should be clearly stated in the description of the logbook dataset.

We added the following sentence to the “Logbook data” section to clarify that we assessed all logbooks, which means that we assessed all fishing effort associated with the fishery.

“All California gillnet vessels are required to submit logbooks that document all of their fishing trips; as a result, these logbooks represent all fishing effort associated with the fishery.”

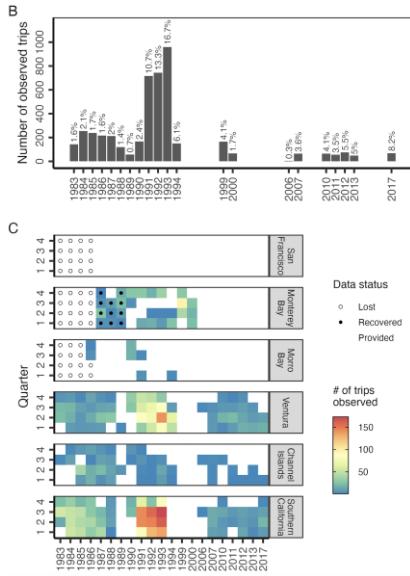
We also added a new “Overview” subsection to the methods and a new opening paragraph to the “Dataset” section that we hope further clarified this point. See our response to related comments above for details.

Lines 248-250: For missing observed coverage, the authors mention that bycatch rates were obtained from historical reports, correct? Are there years where bycatch rates are unavailable from both historical records and observed/logbook records? If so, imputing missing bycatch values by averaging previous and subsequent years may not be appropriate. I suggest the authors verify the best procedure for handling missing values in a time series.

In this response, we (1) answer the two questions posed in the comment; (2) explain how we made the answers to these questions more clear in the revised manuscript; (3) clarify the procedure used to impute missing bycatch rates; and (4) explain why this is the best possible imputation procedure.

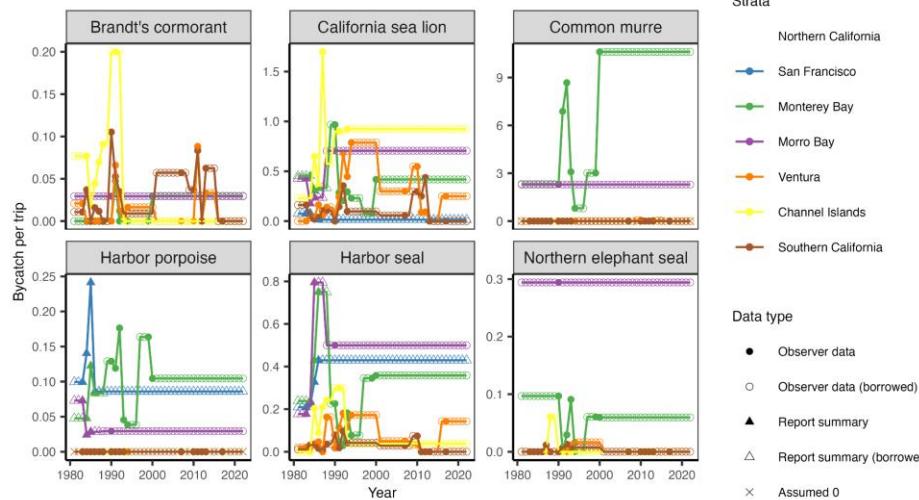
(1) Yes, bycatch rates were obtained from historical reports for strata (regions and years) not covered by the raw observer data and yes, there are strata for which bycatch rates are not available from either historical reports or raw data. These are the strata for which we had to impute bycatch rates. Strata missing bycatch rates can be seen in **Figures 2BC** and **Figure S9** (revised version shown below).

Figure 2BC for reference:



(2) We made this more clear by modifying **Figures S9** to mark which years have bycatch rates from historical reports. Please see the modified figure below.

Figure S9: We added symbols to differentiate between observer/historical report data.



(3) We did not impute missing bycatch rates by averaging previous and subsequent years. This is not possible because, as seen in **Figures 2** and **S9**, observer data ends in 2017 for the southern strata and even earlier for the northern strata; there are no data for “subsequent years” after the observer programs end. This is why we, instead, used the closest available data for the strata, as stated in this sentence in the methods:

“Stratum-specific bycatch rates for years without observer coverage in the stratum are borrowed from the closest year (forwards or backwards) with observer coverage in the stratum (**Fig. 2C & S9**), as has been the practice in previous studies.”

(4) This was our approach because no practicable alternative exists and the use of the most recent year with data is the standard used in the historical reports. We clarified that this has been the approach of current studies by adding the underlined text above.

Lines 252-254: Why was uncertainty not provided for the years in which a raw dataset is available?

We appreciate the encouragement to estimate uncertainty where possible as this is a critical part of rigorous science. However, this is only possible for 6 of the 42 evaluated years (1981-2022): 2006, 2007, 2010, 2011, 2012, and 2017. These are the only years during which the observer program operated in all of the fished strata (see **Fig. 1AC**; **Fig. 2BC**). 2013 almost qualified, except that fishing occurred in the Morro Bay strata that year (and there was no observer coverage there). We feel that estimating uncertainty for such a small, scattered, and non-continuous number of years would add more confusion than value and opted not to estimate uncertainty for these 6 years.

We added the following underlined text to clarify this in the text:

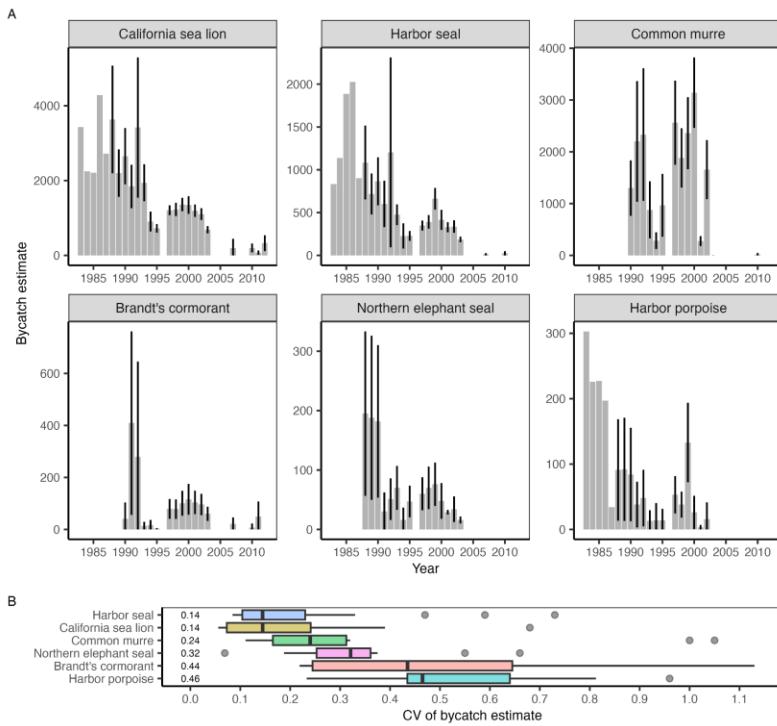
"Because these procedures require raw observer data, we cannot use them for (1) years where summary values from historical reports are used because the raw data have been lost or (2) years without observer data from within one of the fished strata. As a result, only 6 of the 42 evaluated years had the data required to estimate uncertainty: 2006, 2007, 2010, 2011, 2012, and 2017 (Fig. 1AC; Fig. 2BC). An exploration of the uncertainty estimates generated in historical reports with access to the lost data (Table S1) suggests that the median coefficient of variation for estimates of annual bycatch estimates ranges from a low of 0.14 for harbor seal to a high 0.47 for harbor porpoise (**Figure S10B**)."

Lines 257-259: It is not clear what the authors intend to convey with this sentence. Why is the estimate provided for only two species?

The original sentence contained a few typos that made it difficult to understand. We corrected these typos and the sentence now reads:

"An exploration of the uncertainty estimates generated in historical reports (**Table 1**) suggests that the median coefficient of variation for estimates of annual bycatch estimates ranges from a low of 0.14 for harbor seal to a high 0.47 for harbor porpoise (**Figure S10B**)."

In addition to fixing the typos, we added the underlined text to clarify that we were bookending the range of values (i.e., providing the smallest and largest values), rather than listing all five values, which is why only two values were provided. To further clarify this, we printed the median coefficient of variation (CV) for all five species in **Fig. S10B**. Rather than listing all five values, an interested reader can now refer to the figure.



We added the following text to the Fig. S10 caption to explain this addition: “In (B), the median CV of the bycatch estimates for each species is printed on the far left.”

Lines 325-327: The authors state that the ratio estimation method was used to estimate bycatch rates, while Random Forest was employed to evaluate the drivers of bycatch. However, the text also suggests that bycatch estimation was derived using Random Forest, which requires clarification. Additionally, it is unclear how bycatch was estimated using Random Forest with a Bernoulli distribution (0 or 1) rather than a count distribution. Could the authors provide more details on this methodology and clarify how it aligns with the stated objectives?

We took multiple steps to address this comment.

It is correct that we estimated total bycatch using both the design-based ratio estimation approach and the model-based random forest approach; however, we ultimately found the model-based random forest approach to be unsuitable in this particular case study. In the original submission, we documented both approaches in the main text but focused largely on the ratio estimation results, which created understandable confusion.

As a result, we moved the methods and results pertaining to bycatch estimation using the random forest models to the supplemental information. This makes the paper cleaner and easier to understand as only the method used to generate reliable bycatch estimates is now described in the main text. However, we felt it was important not to entirely remove the model-based approach methods and results from the paper, as this

approach is generally thought to perform better than the design-based approach, and it would have been a scientific oversight not to pursue the approach that is generally thought to perform best. Thus, moving this work to the supplemental information retains the work while also making the paper less busy and confusing.

In doing this, we added text to the main text to notify the reader that we tried to estimate total bycatch using the random forest models but that the results were unreliable, and we refer the reader to the supplemental information for details. Specifically, we included the following underlined text in the new methods “Overview” section:

“We used a design-based ratio estimation approach to estimate bycatch of select marine mammal and seabird species in the California set gillnet fishery from 1981-2022. Briefly, the ratio estimation approach estimates total bycatch by applying the bycatch rates (i.e., bycatch per trip) on the sample of fishing trips with trained observers (*section 2.4.1*) to all fishing trips, which are documented in the logbooks maintained by all fishing vessels (*section 2.4.2*). Although model-based approaches, including the random forest approach used in the second portion of our analysis, are generally thought to produce better estimates of bycatch than design-based approaches (Stock et al., 2019), we found model-based approaches to be unsuitable for this specific case study due to data challenges (see *section 2.6.4* and the supplemental information for more details). However, the random forest models were successfully used to evaluate drivers of bycatch and map hotspots of bycatch risk for four of the six evaluated species.”

We also replaced the paragraph describing the methods for estimating bycatch using the random forests, which was moved to the supplement, with this paragraph explaining why the method produced unreliable results and is not featured in the main text:

“We explored using the random forests models to estimate annual bycatch (1981-2022) in the fishery but found them to be unsuitable for this specific case study. Although model-based approaches generally perform better than design-based approaches at estimating bycatch (reference), we found that our random forest models underpredicted bycatch risk in the 1980s and 1990s relative to the ratio estimator (**Fig. S14**). This is most likely because of the loss of observer data from the northern strata during this time period, fishery largely operated in the northern portion of the region. See the supplemental information for details on the estimation of bycatch using the random forest models.”

The following text in the now supplemental methods explains how bycatch was estimated using a random forest model that predicts bycatch presence/absence:

“We summed the number of pseudo-sets predicted to have bycatch each year, converted this sum to “true sets” assuming three sets per pseudo-set (**Fig. S4AB**), and multiplied this sum by the median number of captures when a

capture occurs to generate estimates of the total number of captured animals (**Fig. S4C**)."

Lines 327-328: It is not clear how bycatch can be estimated when the number of fishing sets is under estimation. What are the implications of this for the reliability of bycatch estimation?

We added the following underlined text to tease the answer to this question, which is addressed three sentences later in the paragraph:

Topic sentence with added text: "We used the best fitting model to generate annual estimates of protected species bycatch from 1981 to 2022 by predicting whether "pseudo-sets" recorded in logbooks were likely to have captured each study species and assuming median numbers of sets and caught animals for "pseudo-sets" with bycatch."

Details occurring later in paragraph: "We summed the number of pseudo-sets predicted to have bycatch each year, converted this sum to "true sets" assuming three sets per pseudo-set (**Fig. S4AB**), and multiplied this sum by the median number of captures when a capture occurs to generate estimates of the total number of captured animals (**Fig. S4C**)."

We note that this text now occurs in the supplemental information and not the main text.

Results

Some results presented here were not explicitly stated in the objectives or in the material and methods sections. Additionally, in many instances, the authors are not simply presenting their results but also providing interpretations, which is more appropriate for the discussion section. In the results section, authors typically report findings without offering detailed interpretation. Moreover, the bycatch estimation obtained from Random Forest also requires further explanation to clarify the methodology and its role in the analysis.

We took multiple steps to address this comment.

First, we added the following underlined text to the last paragraph of the introduction to clarify upfront that comparing bycatch levels to management targets is an objective of the study:

"We use ratio estimation methods to reconstruct historical bycatch levels and compare recent bycatch levels to management targets."

Second, we added the methods for comparing recent bycatch levels to management targets to the methods section. See a response to a comment below for details.

Third, we moved the methods and results related to estimating bycatch using the random forest model to the supplemental information. This negates the need to state the use of this method and the comparison of its results to those from the ratio estimation approach as an explicit objective of the paper.

Finally, we carefully reviewed the results and removed many sentences that are more appropriate for the discussion (i.e., they interpret the results).

Many of these points are corrected by the helpful comments below.

Lastly, I think that in the first part of the results, the authors could provide some basic information on bycatch and fishing effort before presenting the bycatch trends. For example, it is only in line 417 that the authors mention that the common murre was the most common bycatch species.

We added the following underlined text to better highlight the most common bycatch species in the 1980s:

"In general, estimated bycatch peaked in the mid-1980s, steadily declined following the 40-fathom depth restriction implemented in 1987, with a temporary increase in the late-1990s followed by continued decline (**Fig. 3**). Bycatch peaked, in order of decreasing magnitude, at 5,059 common murre in 1984, 3,437 California sea lion in 1987, 2,605 harbor seal in 1986, 560 harbor porpoise in 1985, 453 Brandt's cormorant in 1991, and 432 northern elephant seal in 1986 (**Fig. 3**). This pattern reflects trends in fishing effort, which also declined after a peak in 1985, with a brief expansion in the late-1990s followed by continued decline (**Fig. 1C**)."

The last sentence provides a brief summary of the fishing effort trends shown in Figure 1, which was presented in detail in the introduction.

Lines 359-360: This information could be included in the discussion section "40-fathom depth restriction implemented in 1987".

We retained the first sentence of the results as originally written because it is a simple reminder of what happened in 1987 that coincides with this decline.

Line 364: Looking at Figure 3, the peak occurred in 1990, not in 1989 as indicated by the authors.

We deleted these sentences to eliminate confusion.

Lines 362-366: The issue lies in the timing of the changes in bycatch estimation described in the two sentences.

1. First sentence: "Estimated bycatch precipitously declines after a peak in 1989 with a temporary increase occurring in the early-2000s." This indicates that after a peak in 1989, bycatch rates declined until the early 2000s.
2. Second sentence: "This pattern is driven by a steep increase in Brandt's cormorant bycatch rates in the Channel Islands region in 1990 that is assumed to have persisted to today." This implies a steep increase in bycatch rates starting in 1990 and continuing to the present. The contradiction arises because the first sentence describes a decline in bycatch rates after 1989, while the second sentence suggests a steep increase starting in 1990. If there was a decline in bycatch rates from 1989 to 2000 (as the first sentence suggests), it seems inconsistent with the second sentence, which states that there was a steep increase in bycatch rates starting in 1990.

We deleted these sentences to eliminate confusion.

Lines 366-367: Observing Fig. S9, it is also possible to see significant bycatch variability for the California sea lion (yellow line), harbor porpoise (green line), and common murre (green line)

We deleted these sentences to eliminate confusion.

Lines 367-369: A sharp decline is also observed for the elephant seal (Figure 3).

This is a good observation. We added northern elephant seal to this sentence:

"Estimated bycatch of harbor porpoise, northern elephant seal, and common murre declined especially sharply following the 2002 exclusion of fishing from waters shallower than 60 fathoms."

Lines 369-371: I suggest that the authors present only the results in this section and reserve the discussion of the causes behind these patterns for the Discussion section.

We moved this sentence to the discussion.

Lines 373-378: Until this point in the manuscript, the authors have not stated that one of the objectives of the article is to compare the bycatch estimation with potential biological removal. This objective should be clearly outlined in both the Introduction and Materials and Methods sections.

We added the following underlined text to the last paragraph of the introduction to clarify upfront that comparing bycatch levels to management targets is an objective of the study:

"We use ratio estimation methods to reconstruct historical bycatch levels and compare recent bycatch levels to management targets."

We added the following paragraph to the ratio estimation methods to explain that we compare recent levels of bycatch to management targets and to explain the reference points used to judge sustainability:

"We evaluated the sustainability of recent estimated marine mammal bycatch by comparing it to the potential biological removal (PBR) for each stock, which is defined under the MMPA as the maximum number of animals, not including natural mortalities, that may be removed from a marine mammal stock while allowing that stock to reach or maintain its maximum sustainable population. We extracted each PBR from its most recent stock assessment (**Fig. 7**) and compared it to the average estimated catch over the last 10 years (2013-2022). A fishery is managed based on its classification into one of three categories: Category 1 fisheries cause annual mortality and serious injury (M/SI) greater than 50% of the PBR, Category II fisheries cause annual M/SI between 1 and 50% of the PBR, and Category III fisheries cause annual M/SI less than 1% of the PBR. A fishery is considered to be approaching the MMPA's "zero mortality rate goal" (ZMRG) when annual M/SI is below 10% of the PBR."

Lines 390-406: This sentence is more suited for the discussion section, not the results.

We moved this paragraph to the supplemental information.

We added the following text to the end of the first results paragraph to refer the reader to these results in the supplemental info:

"Slight differences between our estimates of annual bycatch and those from historical studies (**Fig. S11**) are driven by a mixture of differences in our methods and input data (**Fig. S12**; see supplemental information for more details)."

Lines 411-412: Move this sentence: 'The models for two species, Brandt's cormorant and harbor porpoise, were excluded from further consideration because of their poor performance (Table 2)' to after the first sentence of the paragraph.

We moved this sentence to the suggested location. This is much improved.

Line 414-415: This sentence should be removed: "This highlights the importance of evaluating multiple modeling approaches when predicting rare bycatch events."

We agree that this is more appropriate for the discussion and deleted this sentence.

Line 416: What do the authors mean by "strongly correlated".

We rewrote this sentence to clarify the correlation, its magnitude, and implications:

"Model performance was positively correlated with the frequency of bycatch observations, i.e., species with more observed bycatch events produced models with greater skill (**Table 1**; $r^2 = 0.64$ for Cohen's kappa for training data)."

See the reply to the comment relating to Lines 423-425 for a graphical representation of this correlation. We did not add this visual to the supplement because the supplement is already long and we judged **Table 1** to be sufficient for conveying this minor result.

Lines 420-422: This sentence should be removed, as the threshold value indicating when a model was not used is already provided in the Materials and Methods section and should not be repeated in the Results.

We removed this sentence and adopted the language proposed below.

Lines 422-423: The sentence should be removed, as the Materials and Methods section already states that fair performance models were also used for prediction.

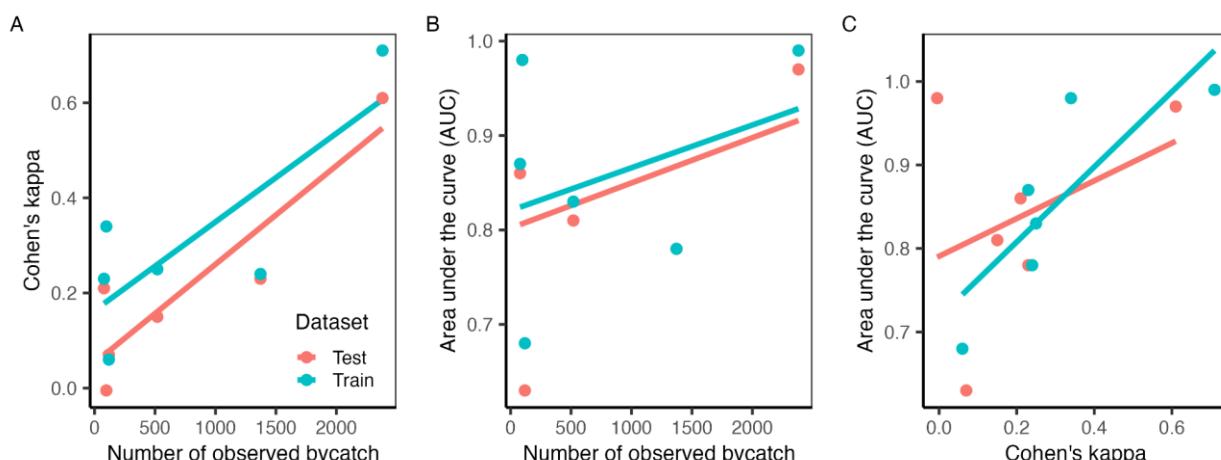
We removed this sentence and adopted the language proposed below.

Lines 423-425: What do the authors mean by 'Cohen's kappa values were correlated with the area under the receiver operator curve (AUC)?' How can the reader identify this correlation?

We added the following underlined text to clarify the correlation and its magnitude:

"Cohen's kappa was positively correlated with the area under the receiver operator curve (AUC) ($r^2=0.68$ for the training dataset), indicating minimal tradeoffs in using this metric for model selection (**Table 1**)."

It can be tabularly assessed in **Table 1** but we provide a figure below for graphical reference. We did not add this visual to the supplement because the supplement is already long and we judged **Table 1** sufficient for conveying this minor result.



Here, I've provided a restructured version of the paragraph (409-425):

"The best-fitting model performed well for all species, except for Brandt's cormorant and harbor porpoise, where poor performance was observed (Cohen's kappa values of less than 0.2). Weighted random forest models performed best for California sea lion, common murre, northern elephant seal, and harbor seal with case weights of 25, 25, 25, and 75, respectively (Table 2). For common murre, the Cohen's kappa value was 0.71, indicating 'good' performance, while for harbor seal, California sea lion, and northern elephant seal, Cohen's kappa values were 0.25, 0.24, and 0.23, respectively, indicating 'fair' performance."

We appreciate this suggestion as it more succinctly describes our results. We adopted this suggested language in the paper with a few small edits. The paragraph now reads:

"The best-fitting model performed well for all species, except for Brandt's cormorant and harbor porpoise, which exhibited poor performance (Cohen's kappa less than 0.2) and were therefore excluded from further consideration. Weighted random forest models performed best for California sea lion, common murre, northern elephant seal, and harbor seal with case weights of 25, 25, 25, and 75, respectively (**Table 1; Fig. S.13**). For common murre, Cohen's kappa was 0.71, indicating "good" performance, while for harbor seal, California sea lion, and northern elephant seal, Cohen's kappa was 0.25, 0.24, and 0.23, respectively, indicating "fair" performance. Model performance was positively correlated with the frequency of bycatch observations, i.e., species with more observed bycatch events produced models with greater skill (**Table 1**; $r^2 = 0.64$ for Cohen's kappa for training data). Cohen's kappa was positively correlated with the area under the receiver operator curve (AUC) ($r^2 = 0.68$ for the training dataset), indicating minimal tradeoffs in using this metric for model selection (**Table 1**)."

Lines 427-430: I suggest the authors explicitly include the categorization of the driver predictors as environmental, fishing-related, temporal, and spatial in the Materials and Methods section (paragraphs 262-268). Please, the process of predictor selection

We added the following underlined text to explain variable selection and categorization:

"We considered nine attributes of fishing as potential drivers of bycatch risk: haul depth (fathoms), mesh size (inches), soak time (hours), latitude ($^{\circ}$ N), longitude ($^{\circ}$ W), distance from shore (km), Julian day, sea surface temperature ($^{\circ}$ C), and whether the fishing occurs near an island (i.e., within 10 km of island coast). These attributes were selected based on their demonstrated relationship to bycatch risk in other papers (e.g., Bettoli et al. 2006, Bjørge et al. 2013, Kroetz et al. 2020) and their availability in the observer data or their ability to be derived through remote sensing (i.e., distance from shore, temperature, island area). They represent a range of spatial (latitude, longitude, distance from shore, depth,

island area), temporal (Julian day), environmental (temperature), and fishing-related (soak time, mesh size) attributes.”

- Kroetz et al. 2020: region, latitude, season, sea surface temperature, depth, net depth, net length, mesh size, gear type, target, and soak time
- Bettoli et al. 2006: mesh size, fish length, water temperature, soak time, and twine type
- Bjørge et al. 2013: Season, areas, depth, soak time
- Kroetz, A.M., Mathers, A.N. and Carlson, J.K., 2020. Evaluating protected species bycatch in the US Southeast Gillnet Fishery. *Fisheries research*, 228, p.105573.
- Bettoli, P.W. and Scholten, G.D., 2006. Bycatch rates and initial mortality of paddlefish in a commercial gillnet fishery. *Fisheries Research*, 77(3), pp.343-347.
- Bjørge, A., Skern-Mauritzen, M. and Rossman, M.C., 2013. Estimated bycatch of harbour porpoise (*Phocoena phocoena*) in two coastal gillnet fisheries in Norway, 2006–2008. Mitigation and implications for conservation. *Biological Conservation*, 161, pp.164-173.

Lines 428-430: How did the authors address potential collinearity between the variables? For example, three spatial variables were included in the same model, which may lead to high collinearity.

We added the following underlined text to clarify that a strength of the random forest approach is that, compared to classical regression techniques, it is insensitive to collinear and unimportant variables:

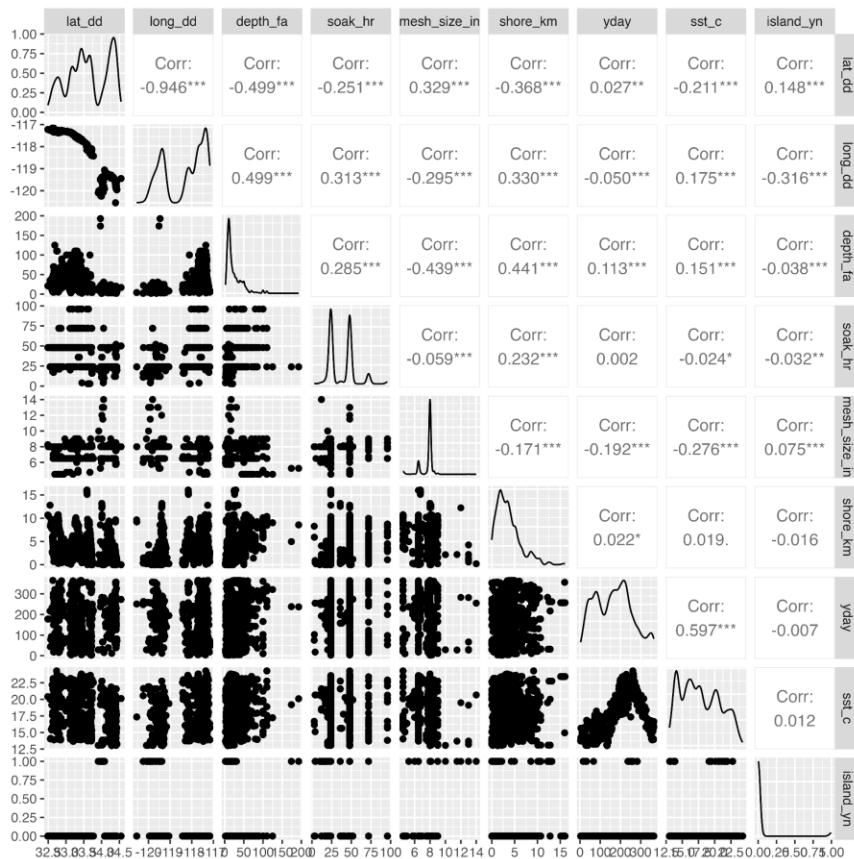
“We used random forest classification models trained on the observer data to identify drivers of bycatch risk for each of the six study species. We used a random forest approach, a machine learning method that ensembles predictions from hundreds of decision trees, rather than a classical regression method (e.g., generalized linear or additive models) because of their comparatively high predictive skill to rare events, ability to model non-linear relationships, and insensitivity to collinear or unimportant predictor variables (Cutler et al., 2007; Prasad et al. 2006).”

Additionally, among the three spatial variables considered, only latitude and longitude showed a strong correlation with each other ($r^2 > -0.9$) (**see figure below**). The correlation between latitude and distance to shore was -0.368, while longitude had a correlation of 0.33 with distance to shore. Despite the collinearity between longitude and latitude, we chose to include both variables for several reasons: (1) random forest models are generally less sensitive to collinearity issues compared to traditional regression techniques, as noted above; (2) including longitude resulted in higher model skill and improved the model's ability to predict bycatch hotspots observed around the Channel Islands; and (3) other studies using random forest for spatiotemporal bycatch predictions typically incorporate both latitude and longitude (Stock et al., 2020).

Stock, B. C., Ward, E. J., Eguchi, T., Jannot, J. E., Thorson, J. T., Feist, B. E., & Semmens, B. X. (2020). Comparing predictions of fisheries bycatch using multiple spatiotemporal species

distribution model frameworks. *Canadian Journal of Fisheries and Aquatic Sciences*, 77(1), 146–163. <https://doi.org/10.1139/cjfas-2018-0281>

A correlogram showing the correlation between the predictor variables.



Lines 433-436: This sentence should be moved to the discussion section.

We deleted this sentence. It is already discussed in the discussion.

Lines 438-439: What do the authors mean by 'correlated' and 'no-correlated'?

We replaced “correlated” and “non-correlated” with “similar” and “dissimilar”. We meant that species sometimes show similar responses to drivers and that they sometimes show dissimilar responses:

“The species exhibit a mixture of similar and dissimilar responses to the explanatory variables (**Fig. 5**).”

Lines 439-441. Simply stating that a peak was observed at 34° latitude does not necessarily indicate that it represents nearshore areas. To confirm this, additional longitude information

would be needed to pinpoint the location more accurately and determine its proximity to the shore.

We added the following underlined text to clarify that the spike at 34° included some deeper offshore areas:

"California sea lion and harbor seal exhibit similar responses in bycatch risk. Both species have higher bycatch risk in shallower depths in nearshore areas with a spike in risk occurring around 34°N latitude, including some deeper offshore areas (Fig. 6)."

We also refer the reader to **Figure 5**, which shows the marginal effects of depth, distance to shore, latitude, and longitude on bycatch risk and **Figure 6**, which shows maps of bycatch risk. Together, these figures substantiate this statement.

Lines 446-448: The dome-shaped relationship is more apparent for the California sea lion and harbor seal, whereas for the elephant seal, a decrease in bycatch risk is observed with increasing temperature.

We rewrote this sentence, which now reads as follows:

"For all four species, bycatch risk exhibits an asymptotic relationship with soak time, though the shapes of these relationships differ by species. The species exhibit complex and variable relationships to temperature."

Line 449: The 'Maps of Bycatch Risk' section is lengthy and repetitive. A reduction is required.

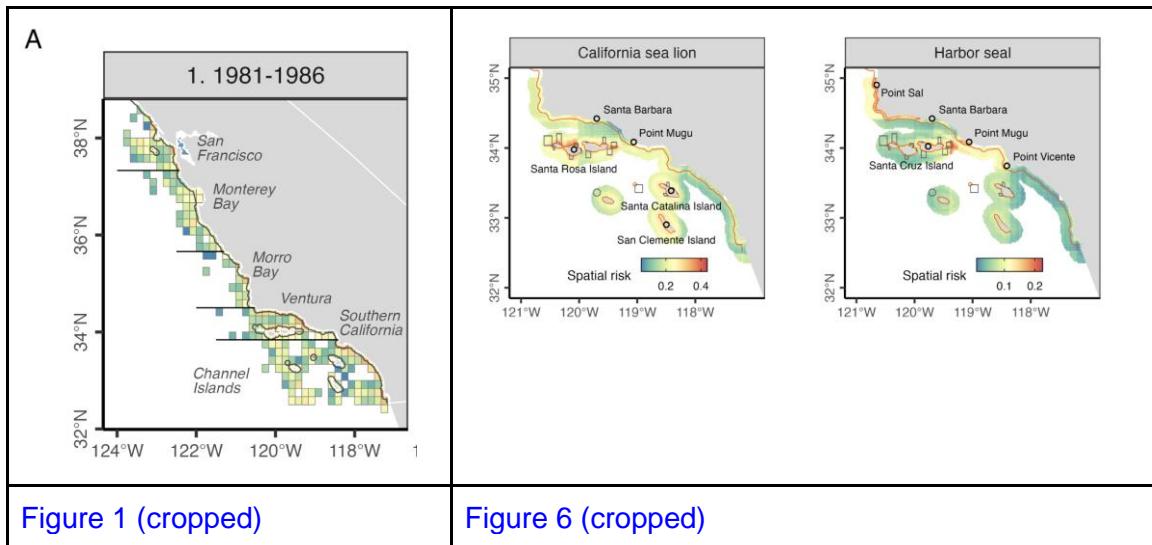
We shortened this section to a single paragraph by moving the sentences that were more related to results interpretation to the discussion. The paragraph now reads:

"The species exhibited different patterns of spatial bycatch risk. California sea lion bycatch risk is predicted to be highest in four areas: (1) on the northern coasts of the northern Channel Islands, especially on the northern coast of Santa Rosa; (2) a small nearshore area west of Santa Barbara; (3) the eastern coast of Santa Cruz Island; and (4) the northwestern shores of Santa Catalina and San Clemente Islands; (**Fig. 6**). Harbor seal bycatch risk is predicted to be highest in four areas: (1) the sliver of nearshore area stretching from Santa Barbara to Point Sal; (2) the eastern coasts of Santa Cruz Island; (3) a broad coastal area near Point Mugu; and (4) the sliver of nearshore area stretching from Point Mugu to Point Vicente (**Fig. 6**). Common murre bycatch risk is predicted to be negligible throughout most of southern California (**Fig. 6**). It is only predicted to be high in a small patch near Point Sal and even there, the maximum risk index is much lower than for the other evaluated species. Like common murre, northern

elephant seal bycatch risk is also predicted to be negligible throughout most of southern California except in the region near Point Sal (**Fig.6**)."

Line 451: Where are the Channel Islands and Santa Cruz located?

We added a label for the Channel Islands, the name for the group of islands in southern California, in **Figures 1 and 6**. We added the word "Island" to the end of "Santa Rosa", "Santa Catalina", "Santa Cruz", and "San Clemente" in **Figure 6** to clarify that these are the names of islands (thereby clarifying where Santa Cruz Island is).



Lines 553-554: This sentence should be moved to the discussion section.

Lines 459-460: This sentence should be moved to the discussion section.

Lines 464-467: This sentence should be moved to the discussion section.

Lines 470-474: This sentence should be moved to the discussion section.

We addressed all four of these comments, which all relate to moving the interpretation of the spatial patterns in bycatch risk to the discussion section, by merging these sentences into a new paragraph in the discussion. The paragraph reads:

"Hotspots of bycatch risk are aligned with the location of known haulouts, breeding colonies, and foraging grounds. High California seal lion bycatch risk around the northern Channel Islands is likely related to the large haulouts of sea lions in that area (**Fig. S15**). Similarly, hotspots of harbor seal bycatch risk correspond to the locations of large harbor seal haulouts on Santa Cruz Island and near Point Mugu (**Fig. S15**). The absence of common murre bycatch risk in southern California is consistent with the distribution of the species, which has no breeding colonies or permanent foraging grounds in southern California (**Fig. S15**). Similarly, the negligible risk for northern elephant seals is consistent with

the phenology of their migrations. Although northern elephant seals breed on the Channel Islands and near San Simeon/Cambria (**Fig. S15**) from December to March, they disperse to their distant foraging grounds (males to Alaska and females to oceanic waters far West of California) before the fishing season peaks from April to June, significantly reducing their vulnerability to the gillnet fishery.”

We believe that the comment referring to lines 553-554 intended to refer to lines 453-454 based on its order in the review and its similarity to the following comments. Lines 553-554 were in the fourth paragraph of the discussion in the original submission.

Lines 476-477: If random forest models also estimate bycatch, why did the authors choose to use ratio estimation methods for bycatch estimation? What is the rationale for using both methods? This reasoning should be explicitly addressed in the Materials and Methods section

Please see our detailed response where this comment is made for the first time. Briefly, we used both methods because (1) the design-based ratio estimation approach has been used to estimate bycatch in all historical assessments (**Table S1**) but (2) model-based approaches are generally thought to perform better and were therefore worth exploring here. Ultimately, we found the model-based approach to be unsuitable for this case study due to the loss of raw observer data and the inability of this approach to use summarized data. In contrast, the ratio estimation approach was less sensitive to the loss of raw observer data because it can use summarized data. We moved the methods and results associated with the estimation of bycatch using the random forests to the supplemental information to avoid confusion.

Lines 476-493: The authors have conflated results with discussion in this paragraph, as well as throughout the results section. A restructuring of the results section is necessary to clearly distinguish between these components. Additionally, this paragraph repeats information already mentioned in the first paragraph of the results section. Similar topics, such as bycatch trends, should be grouped together to enhance coherence and avoid redundancy

This entire paragraph, which presented the results of bycatch estimation using the random forest models, was moved to the supplemental information. As a result, the blended results and discussion is more appropriate because it now occurs in the supplemental information. However, we have reviewed and edited the rest of the results, including the sentences flagged above, to ensure that results interpretation occurs in the discussion and not in the results section.

Lines 477-479: Agreement between two methods does not inherently validate the observed trend—it only demonstrates consistency between the results produced by the methods. The validity of the trend depends on the robustness of the methodologies, the quality of the data, and the assumptions underlying the analysis.

We deleted this sentence. Furthermore, this entire paragraph, which presents results and discussion related to the use of the random forest models to estimate bycatch, was moved to the supplemental information.

Lines 488-490: What does the author mean by this sentence? It is very confusing, and clarification is needed.

We entirely rewrote the underlined sentence to clarify its meaning. It now reads:

"Unlike the random forest models, the ratio estimators are able to use summarized observer data for this region and time period from old reports. As a result, the ratio estimators can learn from observations from this region and time period while the random forest models are blind to data from this region and period. Thus, the random forest models are likely to underpredict risk in early years in northern strata because they largely learned from late years in southern California, where risk was lower."

Furthermore, this entire paragraph, which presents results and discussion related to the use of the random forest models to estimate bycatch, was moved to the supplemental information.

Discussion

Lines 495-496: A reference for this sentence is not needed. Remove '(Carretta et al., 2014)'.

We removed the reference for this sentence, which now reads:

"Our study provides the first update to total estimates of protected species bycatch in the California set gillnet fishery since 2012."

Lines 500-501: Reference for this sentence is required.

This is a novel result of our study; thus, providing a reference is inappropriate and not possible. We could refer to Fig. 3 but the figure and result was already presented in the results section.

Lines 501-503: What do the authors mean by: 'are more due to reductions in fishing effort than to reductions in bycatch rates (i.e., bycatch per unit effort)', since reducing fishing effort is expected to lead to a decrease in bycatch rates? Please clarify this statement.

We added the following underlined text to clarify this sentence:

"These advances, while directly attributable to management interventions, are more due to reductions in fishing effort (i.e., fewer fishing trips) than to reductions in bycatch rates (i.e., lower bycatch per fishing trip)."

We hope that this clarifies how bycatch *rates* are different from bycatch *amounts*. Reduced fishing effort results in reduced bycatch amounts assuming bycatch rates are unchanged. Reduced fishing effort does not reduce bycatch rates.

$$\begin{aligned} \text{Amount} &= \text{effort} * \text{rate} \\ \text{Amount of total bycatch} &= [\text{number of trips}] * [\text{bycatch per trip}] \end{aligned}$$

Line 505: What do the authors mean by 'sustained declines'? Please clarify this statement.

We meant that fishing opportunities have undergone consistent declines while marine mammal populations have undergone consistent growth. We replaced "sustained" with "prolonged" to clarify this meaning. The sentence now reads as follows:

"This highlights a steep tradeoff between conservation and fisheries objectives under the current management regime: while populations of protected species have undergone sustained growth, fishing opportunities and revenues have undergone prolonged declines."

Lines 508-510: In the first paragraph of the introduction, the authors state: 'In this study, we reconstruct the bycatch of select marine mammals and seabirds in the California set gillnet fishery from 1981-2022 and identify drivers of bycatch that could be used to design management measures that effectively and efficiently reduce bycatch.' However, in the first paragraph of the discussion section, they state: 'Our results indicate that current fishing operations do not pose a threat to the evaluated species, which suggests that current management is sufficient at limiting bycatch.' If current management is already effective, why is there a need to identify drivers to efficiently reduce bycatch?

The need to identify drivers of bycatch risk, despite the low levels of bycatch, could be more clearly justified. It may be helpful to emphasize that understanding these drivers is important for enhancing knowledge and fine-tuning management practices to ensure that bycatch remains at low levels over time, even as conditions change. This would make the identification of drivers less about urgent management needs and more about ongoing refinement of strategies and maintaining low bycatch levels.

The justification for identifying the drivers could be more explicitly linked to improving long-term sustainability or enhancing the effectiveness of current measures, rather than suggesting that further management interventions are immediately necessary.

We did not know that bycatch levels were low before embarking on this study. In fact, conservation groups were claiming that bycatch levels were high and problematic and were therefore lobbying for increased restrictions on the fishery. As a result, the *a priori* objectives of our study were to estimate current bycatch levels and to identify drivers of

bycatch that could be used to inform management strategies that lead to effective and **efficient** bycatch reduction. We bold “efficient” because it is distinct from “effective”. An effective strategy avoids or limits bycatch, but might not consider impacts to fishing opportunities. An efficient strategy achieves bycatch reduction goals while minimizing impacts on fishing opportunities. Although we ultimately found that management has been effective at reducing bycatch (i.e., current bycatch levels are below NOAA’s net zero mortality rate goal), they have caused great hardship for the fishery: as summarized in the introduction, the fishery has shrunk from generating \$15 million annually for 400 vessels in 1987 to generating \$1 million annually for 40 vessels today. As a result, we evaluate drivers of bycatch to propose management refinements that expand fishing opportunities while still achieving bycatch reduction goals (i.e., improve efficiency).

We took several actions to make the above explanation in the paper.

First, as described in response to comments above, we clarified that the study was motivated by claims from conservation organizations that bycatch is high and problematic and that the fishery requires more regulation. To evaluate these claims, we naturally asked the questions: *“how much bycatch is there”* and *“what can be done about it?”* We did not know if bycatch was high or low before conducting our study.

Second, we clarified the definition of efficient management and our goal to support more efficient management by adding/editing the following underlined text to the last paragraph of the introduction, where we introduce the study objectives:

Opening sentence: In this study, we reconstruct the bycatch of select marine mammals and seabirds in the California set gillnet fishery from 1981-2022 and identify drivers of bycatch that could be used to refine management to more efficiently reduce bycatch, where efficient management achieves conservation objectives while minimizing impacts on fishing opportunities.

Final sentence: Based on these results, we make recommendations for how management could more efficiently manage bycatch risk through measures such as seasonal or spatial closures, depth restrictions, or gear restrictions.

We made other small edits throughout the introduction and discussion to clarify this. Collectively, these edits build on other sentences already included in the abstract and introduction including:

Abstract: “The identification of efficient management strategies that reduce protected species bycatch while also minimizing impacts on fishing livelihoods is a global conservation challenge. Identifying such strategies requires understanding levels of bycatch relative to management targets as well as the relationship between bycatch risk and potential management levers.”

Introduction: “Despite declining fishing effort and bycatch, conservation groups are lobbying for additional restrictions, including permanent closure, to further avoid bycatch (Birch et al., 2023; Birch and Shester, 2023). There is thus great need for scientific guidance on management regulations that are likely to provide conservation benefits while also avoiding unnecessary burdens on the fishing industry.”

Third, we made all of the changes described in response to the comment below to substantiate why spatial-temporal hotspot closures could empower efficient management that achieves the same level of bycatch reduction while increasing fishing opportunities.

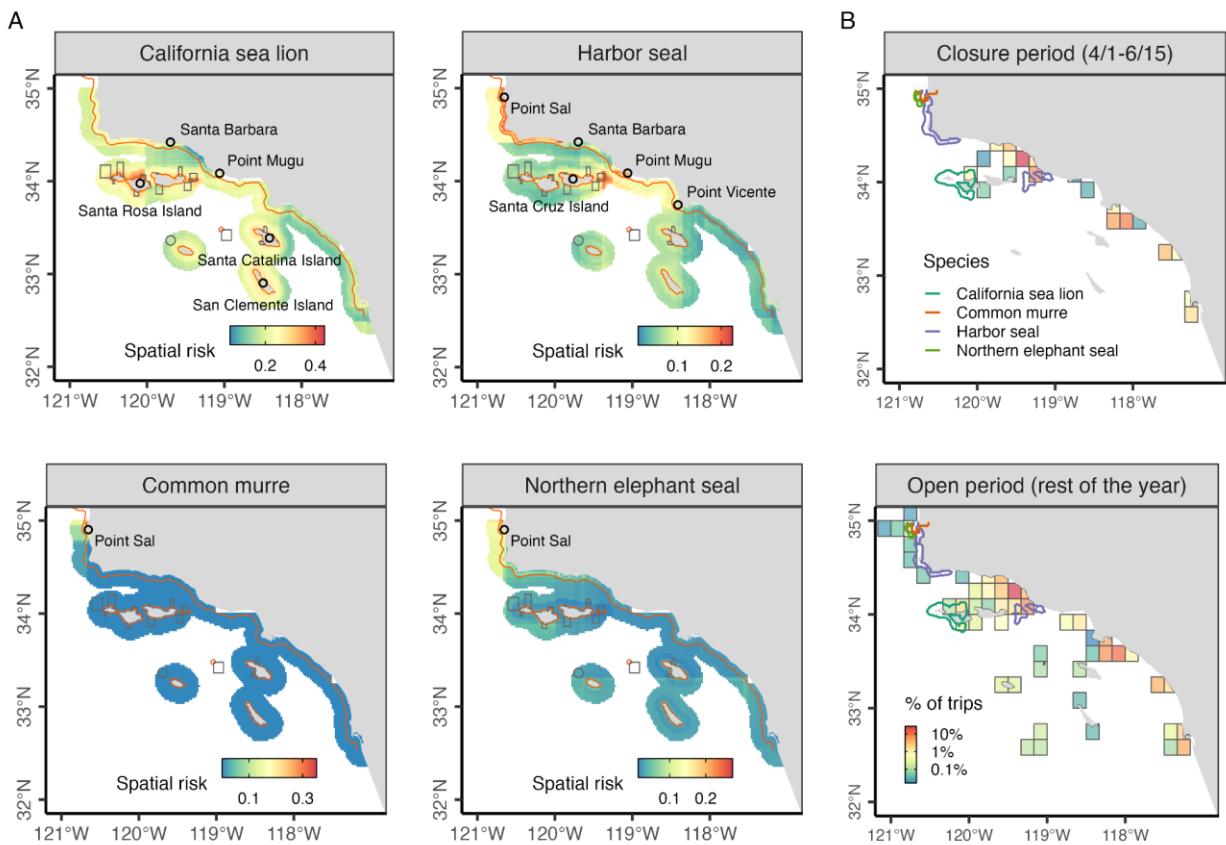
More broadly, these results can be used to anticipate the impact of future changes in management regulations on bycatch, whether they are motivated by bycatch or not.

Lines 508-510: But how could identifying spatial-temporal bycatch hotspots improve economic outcomes? The authors mention this justification in the introduction, but do not explain how.

We took several actions to address this important comment.

First, we expanded **Figure 6** to show the identified bycatch hotspots relative to both the distribution of (1) current closures and (2) recent fishing effort. This illustrates the potential efficiency of seasonal closures by showing that (1) there are areas currently closed to fishing that have low predicted bycatch risk and (2) that bycatch hotspots are located in areas of relatively minor fishing importance, which suggests that the loss of fishing opportunities associated with their brief seasonal closure could be made up by opening areas of low predicted risk that are currently closed to fishing. The figure is

shown below for reference.



Second, we added the following underlined text to the discussion to describe this result:

"Our results suggest that spatial-temporal management could more efficiently and effectively manage bycatch risk than gear modifications or soak time regulations. Specifically, bycatch rates for California sea lions and harbor seals are greatest from April 1 to June 15, suggesting that a 2.5 month seasonal closure of bycatch hotspots for these two species could prevent bycatch while allowing the opening of less risky but currently closed areas to fishing. These hotspots are predicted to disproportionately contribute to bycatch yet are of minor fishing importance (Fig. 6B) suggesting that the loss of fishing opportunities in these areas during brief seasonal closures could be easily made up by opening areas of low predicted bycatch risk. As a result, seasonal closures could broaden fishing opportunities while continuing to meet bycatch avoidance objectives.

However, we caution that such hotspot closures could exacerbate bycatch problems if fishing effort is displaced and concentrated in areas of secondarily high risk (Free et al., 2023). Therefore, monitoring of fishing effort and bycatch rates are important for verifying that seasonal closures achieve their conservation and fisheries objectives. Additionally, any changes in current management strategies must take into account spatial-temporal patterns of bycatch and relative sensitivity of each species."

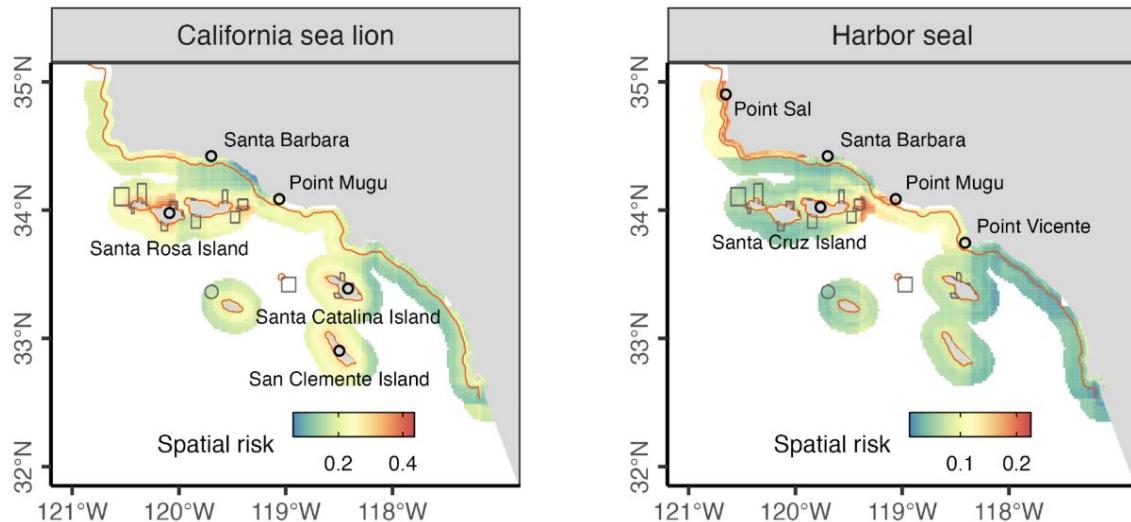
Finally, as described in response to other comments, we revised the introduction to better introduce the concept of efficient management that achieves bycatch objectives while minimizing the impact on fishing opportunities.

Lines 516-524: A figure showing the 0-3 nautical miles offshore area is required, as well as all other restrictions. It is important to visually represent the area using shaded polygons to help the reader understand the coverage of the restricted areas for all fishing regulations.

We added a thin orange line to **Figure 6** to delineate the nearshore areas where gillnet fishing is prohibited: (1) the area 0-3 nautical miles offshore from the mainland, and (2) the area extending 1 nautical mile or 70 fathoms (whichever is closer to shore) around the Channel Islands. Additionally, we mapped Marine Protected Areas, where set gillnet fishing is also excluded, as grey polygons. We added the following sentence to the caption to explain this regulation:

"The thin orange coastal line marks the nearshore areas from which gillnet fishing is excluded: within 3 nautical miles of the mainland and within 1 nautical mile or shallower than 70 fathoms (whichever is closer to shore) from the Channel Islands. The gray polygons indicate the locations of Marine Protected Areas in Southern California, where all set gillnet fisheries are excluded. Spatial bycatch risk is shown only for southern California, as this is the only area where the fishery can operate under current regulations."

Figure 6 (cropped):



Lines 516-524: The authors suggest adjusting the current closed area to target regions identified with higher bycatch risk, justifying that this change could potentially open up more

fishing grounds. However, how can the authors be certain that the current closed area has lower bycatch risk than the areas identified with higher bycatch risk, considering that the 0-3 nautical miles offshore closed area was not sampled? I would expect that areas where fishing is prohibited would not have been sampled. If the current closed area has a higher bycatch risk than the areas identified by the authors, bycatch could increase. Furthermore, even if the current closed area has a lower bycatch risk than the areas identified by the authors, but covers a larger area than the areas identified, bycatch could still increase. The authors need to provide a clearer explanation of how the bycatch risk map was constructed and discuss the implications of changing the current fishing restrictions.

We took several actions to address this comment.

First, we added a new subsection header “2.6.3 Mapping spatial bycatch risk” to better highlight the portion of the methods where the method for constructing the bycatch risk maps is presented.

Second, we expanded **Figure 6** to better illustrate the implications for changing current fishing regulations (i.e., closing high risk areas to fishing and opening low risk areas to fishing) and added text to describe these results to the discussion. See our response to two comments above for details.

Finally, we remind the reviewer that our bycatch risk maps are predicted using a random forest model trained using the full observer data, which spans fishing trips from 1983-1995 (coastwide), 1999-2000 (Monterey Bay), 2010-2013 (South of Point Conception), and 2017 (South of Point Conception). The 0-3 nautical mile closure was implemented after 1994, but prior to that year, set gillnet fishing was still allowed in the current closure area, permitting the observer program to collect samples from these regions (as shown in **Figures S3 and S7**). By including observer data from the period before 1994 in our model training, we account for bycatch events that occurred within the current state water closure. This helps us capture potential high bycatch risks within the 0-3 nautical mile zone in the model predictions. Notably, our model predicts a bycatch hotspot for harbor seals in a nearshore area stretching from Santa Barbara to Point Sal, within the current 0-3 nautical mile closure (**Figure 6**). This prediction provides evidence that our model effectively learns from historical observations to make predictions in areas that may not have been sampled recently.

Lines 528-530: In the introduction, the authors did not mention that a method comparison was intended. An explanation is required to clarify why the comparison of methods is being presented and how it contributes to the study's objectives.

Please see our detailed response where this comment is made for the first time. Briefly, we used both methods because (1) the design-based ratio estimation approach has been used to estimate bycatch in all historical assessments (**Table S1**) but (2) model-based approaches are generally thought to perform better and were therefore worth

exploring here. Ultimately, we found the model-based approach to be unsuitable for this case study due to the loss of raw observer data and the inability of this approach to use summarized data. In contrast, the ratio estimation approach was less sensitive to the loss of raw observer data because it can use summarized data. We moved the methods and results associated with the estimation of bycatch using the random forests to the supplemental information to avoid confusion.

Line 527-530: The authors need to clearly highlight in the Introduction and Materials and Methods sections the rationale for using the ratio estimator to estimate bycatch. Additionally, they should specify the period and region for which bycatch estimation was possible using the random forest model. Furthermore, the authors need to clarify that it was not possible to produce spatial bycatch risk assessments for northern regions due to the absence of data.

Please see our detailed response where this comment is made for the first time. Briefly, we used both methods because (1) the design-based ratio estimation approach has been used to estimate bycatch in all historical assessments (**Table S1**) but (2) model-based approaches are generally thought to perform better and were therefore worth exploring here. Ultimately, we found the model-based approach to be unsuitable for this case study due to the loss of raw observer data and the inability of this approach to use summarized data. In contrast, the ratio estimation approach was less sensitive to the loss of raw observer data because it can use summarized data. We moved the methods and results associated with the estimation of bycatch using the random forests to the supplemental information to avoid confusion.

We clarified that the bycatch estimation using the random forest model was done for all years (i.e., same years as the ratio estimation analysis) in both the main text and supplemental information:

Main text: “We explored using the random forests models to estimate annual bycatch (1981-2022) in the fishery but found them to be unsuitable for this specific case study.”

Supplemental information: “We used the best fitting model to generate annual estimates of protected species bycatch from 1981 to 2022 by predicting whether “pseudo-sets” recorded in logbooks were likely to have captured each study species and assuming median numbers of sets and caught animals for “pseudo-sets” with bycatch.”

We can use the random forest models to make predictions of spatial bycatch risk in the northern regions; we simply chose to limit the predictions to the southern region, the only area where the fishery can operate under current management, to increase the visibility of bycatch hotspots in this priority region. This was stated in the methods section (see underlined text below):

"To generate maps of spatial bycatch risk, we used the best fitting model to predict risk to a 0.02° grid spanning southern California (32-35°N and 117-121°W), the only area where the fishery can operate under current regulations."

However, we added the following underlined text to the caption of **Fig. 6**, which shows the maps of the predicted spatial bycatch risk, to clarify this choice there.

"Fig. 6. Average spatial bycatch risk as estimated by the best fitting random forest model for the four study species with acceptable model performance. The spatial bycatch risk represents the probability of bycatch at a given location under recent (2010-2021) average conditions. Key landmarks for delineating bycatch hotspots are labeled in each panel. Spatial bycatch risk is shown only for southern California, as this is the only area where the fishery can operate under current regulations."

Lines 539-543: It is important to discuss that fishery data obtained from logbooks depends on the trustworthiness of fishers to accurately report bycatch. In some cases, data collected by onboard observers may provide more reliable and accurate information.

This is correct and this is why we used bycatch rates from observed trips (observer data) to predict bycatch from unobserved trips (logbook data).

To clarify this, we added a new “Overview” subsection to the methods where we now provide an overview of both the analytical approach and the datasets used to support this approach. We also added a new opening paragraph in the “Dataset” section to further explain the difference and role between the two datasets.

See responses above for more details.

Lines 545-550: This difference should not be discussed, as it is well-established that a model-based approach to estimate bycatch is preferable to a ratio-based estimate. The use of ratio estimation in this study was merely circumstantial, driven by the absence of raw data. This rationale should be explicitly stated in the Materials and Methods.

This sentence was deleted.

Please see our detailed response where this comment is made for the first time. Briefly, we used both methods because (1) the design-based ratio estimation approach has been used to estimate bycatch in all historical assessments (**Table S1**) but (2) model-based approaches are generally thought to perform better and were therefore worth exploring here. Ultimately, we found the model-based approach to be unsuitable for this case study due to the loss of raw observer data and the inability of this approach to use summarized data. In contrast, the ratio estimation approach was less sensitive to the loss of raw observer data because it can use summarized data. We moved the methods

and results associated with the estimation of bycatch using the random forests to the supplemental information to avoid confusion.

Lines 550-552: Taking into account the authors' conclusion, if both methods produce different results, could this suggest that the reported low levels of bycatch might not be reliable? The agreement or disagreement between methods does not inherently confirm or discredit the reliability of the results. Instead, it underscores the need for a discussion focused on the strengths, limitations, and assumptions of each method. Moreover, the comparison between the two methods was not stated as an objective of the present study. It is essential to clarify whether the authors used the ratio estimator due to the lack of raw data for certain years or whether the intention was to compare the ratio estimator with the random forest approach.

We removed this sentence and concept from the paper. Furthermore, see the responses above detailing how and why we moved the estimation of bycatch using the random forests to the supplemental information.

Lines 577: This figure is not included in the Results section. If the authors choose to include figures, they should be presented consistently within the Results section.

This figure illustrates data from another study and not data generated by our study. Thus, these data are not presented as results; instead, they are presented as useful context for interpreting our results, which is the reason why the figure is referenced in the discussion section. We see no policy against this in the *Biological Conservation* author guidelines but will remove this figure and reference if the editor advises us that this practice is against the journal guidelines.

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1 Estimates and drivers of protected species bycatch in the 2 California set gillnet fishery

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3 Abstract

4 The identification of efficient management strategies that reduce protected species bycatch while also
5 minimizing impacts on fishing livelihoods is a global conservation challenge. Identifying such strategies
6 requires understanding levels of bycatch relative to management targets as well as the relationship
7 between bycatch risk and potential management levers. In this study, we use ratio estimators to
8 reconstruct bycatch of select marine mammal and seabird species in the California $\geq 3.5"$ set gillnet
9 fishery from 1981-2022 and random forest models to identify drivers and hotspots of bycatch risk. We
10 find that bycatch has dropped precipitously since the 1980s as a result of management, but at significant
11 costs to fisheries participation and revenues. Recent marine mammal bycatch ranges from 0.1% to 4.0%
12 of the potential biological removal and marine mammal populations are recovering. Spatial-temporal
13 drivers of bycatch risk were more important than fishing-related drivers of risk, suggesting that spatial-
14 temporal closures would be more effective than mesh size or soak time restrictions at ~~further~~ limiting
15 bycatch. For each species, we identified 2-5 hotspots of elevated bycatch risk as candidates for temporary
16 seasonal closures. Bycatch risk for harbor seal (*Phoca vitulina*) and California sea lion (*Zalophus*
17 *californianus*), the species with the greatest bycatch risk, is especially high from April 1st to June 15th,
18 suggesting that hotspot closures during this 2.5-month time period could be ~~especially~~particularly
19 efficient. Our study also highlights the value of competing multiple modeling approaches to identify
20 methods that best predict rare bycatch events.

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22 **Keywords:** gillnet, bycatch, marine mammals, seabirds, area-based management

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11 24 1. Introduction

12
13 25 Bycatch, the accidental capture of non-target species in fisheries, presents a significant
14 26 conservation and economic challenge ([Crowder and Murawski, 1998; Soykan et al., 2008](#)). Bycatch of
15 27 large-bodied, slow-growing, low-productivity species such as marine mammals, sea turtles, and seabirds
16 28 ([Crowder and Murawski, 1998; Read et al., 2006; Soykan et al., 2008](#)) is of particularly high concern, as
17 29 the mortality of just a few individuals in these vulnerable populations, often recovering from historical
18 30 exploitation, can threaten population collapse and even extinction ([Geijer and Read, 2013; Read et al.,](#)
19 31 [2006](#)), ([Geijer and Read, 2013; Read et al., 2006](#)). As a result, many countries have established strict
20 32 mandates to limit bycatch of vulnerable species, which can result in fisheries closures and other severe
21 33 restrictions ([Crowder and Murawski, 1998; Senko et al., 2014](#)), ([Crowder and Murawski, 1998; Senko et](#)
22 34 [al., 2014](#)). These management disruptions can have serious social, cultural, and economic impacts on
23 35 fishing communities ([Smith et al., 2020](#)), ([Senko et al., 2014](#)). Due to the negative ecological, economic,
24 36 and social impacts, consequences of fishery bycatch, bycatch avoidance is an important objective for
25 37 global fishery management. A sustainably managed fishery with low bycatch can not only provide
26 38 ecological benefits, but also social and economic benefits by providing a sustainable source of income,
27 39 food, and nutrition.

28 40
29 41 To guide effective bycatch reduction policies, it is important to understand the magnitude of
30 42 historical and recent bycatch as well as the drivers of bycatch in a fishery. Estimates of total bycatch are
31 43 needed to determine whether bycatch exceeds management targets or is on pace to exceed targets in the
32 44 near future ([Bjørge et al., 2013; Geijer and Read, 2013; Read et al., 2006](#)), ([Bjørge et al., 2013; Geijer and](#)
33 45 [Read, 2013; Read et al., 2006](#)). Historical bycatch estimates also offer insights into the effectiveness of
34 46 past management interventions, which provide useful benchmarks for adapting management in response
35 47 to contemporary, recent bycatch levels and trends. Understanding the drivers of bycatch risk is critical to
36 48 guiding effective and efficient management adaptations. For example, determining whether bycatch is
37 49 concentrated within specific areas or seasons can support the design of time-area closures that prevent
38 50 fishing when and where risk is high while maintaining fishing opportunities elsewhere ([Lewison et al.,](#)
39 51 [2014; O'Keefe et al., 2023; Soykan et al., 2008](#)), ([Lewison et al., 2014; O'Keefe et al., 2023; Soykan et al.,](#)
40 52 [2008](#)). Similarly, gear restrictions, soak time restrictions, or time of day restrictions can be used to curb
41 53 bycatch if there are strong relationships between bycatch risk and gear or other characteristics of fishing
42 54 ([O'Keefe et al., 2023](#)), ([O'Keefe et al., 2023](#)). Without this information, bycatch management must be
43 55 precautionary to guarantee compliance with protected species legislation, which could forego
44 56 considerable fisheries yields.

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Because observer programs, which place trained scientists on fishing vessels to collect bycatch data, are costly and rarely cover all fishing trips, various analytical approaches have been developed to estimate unobserved bycatch and to evaluate drivers of bycatch risk. Ratio estimation, which a design-based approach that assumes that the rate of bycatch in observed fishing trips is proportional to the rate for all fishing trips, is one of the most common strategies (Cochran, 1977; Stock et al., 2019)(Cochran, 1977; Stock et al., 2019). The reliability of this simple approach increases if there are sufficient data to support estimates within meaningful spatial or temporal strata (e.g., regions, depth zones, seasons). However, ratio estimation can produce biased estimates if other factors (e.g., gear type, soak time, time of day) influence bycatch rates (ICES, 2007)(ICES, 2007), if the observed trips are not representative of the unobserved trips, or if low sample sizes lead to spuriously low or high bycatch rates within a stratum (Martin et al., 2015; McCracken, 2004; Ortiz and Arocha, 2004; Rochet and Trenkel, 2005)-(Martin et al., 2005; McCracken, 2004; Ortiz and Arocha, 2004; Rochet and Trenkel, 2005). Model-based approaches, which use either statistical (e.g., generalized linear models, generalized additive models) or machine learning (e.g., random forests, boosted regression trees) models to estimate bycatch, can overcome many of these limitations by incorporating a wider suite of covariates and areby allowing for non-linear relationships and are generally thought to produce better bycatch estimates (Stock et al., 2019)(Stock et al., 2019). Additionally, thesemodel-based approaches can support management by identifying drivers of bycatch risk and by predicting detailed hotspots of risk (Long et al., 2024; Lopez et al., 2024; Stock et al., 2019)(Long et al., 2024; Lopez et al., 2024; Stock et al., 2019).

The California set gillnet fishery would benefit from updated bycatch estimation due to growing concerns about the fishery's impact on protected marine mammals, which have led some conservation organizations to call for the fishery's closure (Birch et al., 2023; Birch and Shester, 2023)-(Birch et al., 2023; Birch and Shester, 2023). The fishery occurs in southern California and targets California halibut (*Paralichthys californicus*), white seabass (*Atractoscion nobilis*), and Pacific angel shark (*Squatina californica*), among other species. It is currently listed as a Category II fishery under the U.S. Marine Mammal Protection Act (MMPA), indicating that it presents a medium bycatch threat to protected marine species (NOAA, 2024)(NOAA, 2024). Bycatch of marine mammal and seabird species, including harbor porpoise (*Phocoena phocoena*), southern sea otter (*Enhydra lutris nereis*), and common murre (*Uria aalge*) was high during the 1980s and 1990s, prompting significant large-scale management interventions (Forney et al., 2001; Julian and Beeson, 1998)(Forney et al., 2001; Julian and Beeson, 1998). The fishery has also impacted pinniped species such as California sea lion (*Zalophus californianus*), harbor seal (*Phoca vitulina*), and northern elephant seal (*Mirounga angustirostris*). Total bycatch in the fishery has not been estimated since 2012 (Carretta et al., 2014) and there are concerns that bycatch may remain an

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11 92 issue (Birch et al., 2023; Birch and Shester, 2023)(Carretta et al., 2014) and some conservation groups are
12 93 concerned that bycatch remains an issue (Birch et al., 2023; Birch and Shester, 2023).
13 94

15 95 A number of management actions have been taken to reduce bycatch in the California set gillnet
16 96 fishery. During the 1980s, high bycatch of southern sea otters and common murres in central California
17 97 (Barlow et al., 1994)(Barlow et al., 1994) led to a depth restriction that closed fishing inside of 40
18 98 fathoms (73 m) in 1987 (Forney et al., 2001)(Forney et al., 2001). This restriction shut down the fishery
19 99 in the San Francisco area, effectively pushing it south of Pigeon Point and into Monterey Bay and Morro
20 100 Bay (Fig. 1A-2). In 1990, the state adopted Proposition 132 (CA Secretary of State, 1990)(CA Secretary
21 101 of State, 1990), which went into effect in 1994 and banned the fishery in mainland state waters (0-3
22 102 nautical miles) and in waters within 1 nautical mile or 70 fathoms of depth, whichever is less restrictive,
23 103 around the Channel Islands to further reduce bycatch of protected ~~marine~~ species (FGC §8610.1-
24 104 8610.16). In 2002, the state expanded the existing depth restriction, closing fishing inside of 60 fathoms
25 105 (110 m) to avoid the harbor porpoise population in Central California (14 CCR §104.1). This effectively
26 106 closed the fishery in Monterey Bay and Morro Bay (Fig. 1A-4). Currently, the fishery only operates in
27 107 ~~Southern~~southern California (south of Point Arguello) outside 3 nautical miles from the mainland and
28 108 outside 1 nautical mile or shallower than 70 fathoms (whichever is less) from the Channel Islands.
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31 109 Although these regulations are believed to have reduced bycatch in the set gillnet fishery,
32 110 (Carretta et al., 2014), they have also greatly reduced fishery participation and revenues. The
33 111 implementation of the 40 fathom depth restriction in 1987 triggered a precipitous decline in participation
34 112 from ~400 vessels in 1987 to ~100 vessels in 1994. Since then, participation has continued to decline,
35 113 with only~40 vessels active in 2022, and the vast majority (>90%) of landings coming from just 13
36 114 vessels (CDFW, 2023) (Fig. (CDFW, 2023) (Fig. 1B). Fishing effort has similarly decreased from an
37 115 estimated ~15,000 days of fishing trips in 1987 to 1,000 daystrips in 2022 (Fig. 1C). This reduction in
38 116 effort has significantly reduced bycatch levels (Carretta et al., 2014) (Carretta et al., 2014) but at large
39 117 costs to fishery revenues. Fleetwide revenues decreased from US\$15 million in 1987 to onlyUS\$1
40 118 million in 2022 (Fig. 1C; both values in 2022 dollars). Despite declining fishing effort and bycatch,
41 119 thereconservation groups are callslobbying for additional restrictions, including permanent closure, to
42 120 further avoid bycatch (Birch et al., 2023; Birch and Shester, 2023)(Birch et al., 2023; Birch and Shester,
43 121 2023). There is thus great need for scientific guidance on management regulations that are likely to
44 122 provide conservation benefits while also avoiding unnecessary burdens on the fishing industry.
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In this study, we reconstruct the bycatch of select marine mammals and seabirds in the California set gillnet fishery from 1981-2022 and identify drivers of bycatch that could be used to ~~design refine~~
~~management measures that effectively and to more~~ efficiently reduce bycatch, ~~where efficient~~
~~management achieves conservation objectives while minimizing impacts on fishing opportunities~~. We
focus on six protected species encountered in the set gillnet fishery: California sea lion, harbor seal,
northern elephant seal, harbor porpoise, common murre, and Brandt's cormorant, (*Phalacrocorax*
penicillatus). Southern sea otter, despite being one of the original species of management concern, is not
included in this analysis due to data limitations (Fig. 2A). We use ratio estimation methods to reconstruct
historical bycatch levels ~~and compare recent bycatch levels to management targets~~. These methods, which
have been used to estimate bycatch in the fishery at various points in the past, (Table S1), provide a
complete time series of bycatch estimates using methods approved for stock assessment and management.
We then use random forest models to evaluate drivers of bycatch risk and to make predictions of spatial
bycatch risk for four species of concern; Brandt's cormorant and harbor porpoise were excluded ~~from this~~
~~portion of the analysis~~ due to poor model performance (Table 21). Based on these results, we make
recommendations for how management could ~~effectively more efficiently~~ manage bycatch risk through
measures such as seasonal or spatial closures, depth restrictions, or gear restrictions.

2. Methods

2.1 Overview

We used a design-based ratio estimation approach to estimate bycatch of select marine mammal
and seabird species in the California set gillnet fishery from 1981-2022. Briefly, the ratio estimation
approach estimates total bycatch by applying the bycatch rates (i.e., bycatch per trip) on the sample of
fishing trips with trained observers (section 2.4.1) to all fishing trips, which are documented in the
logbooks maintained by all fishing vessels (section 2.4.2). Although model-based approaches, including
the random forest approach used in the second portion of our analysis, are generally thought to produce
better estimates of bycatch than design-based approaches (Stock et al., 2019)(see section 2.6.4 and the
supplemental information for more details)(Stock et al., 2019)evaluate drivers of bycatch and map
hotspots of bycatch risk for four of the six evaluated species. We used a random forest approach, a
machine learning method increasingly used in ecology, because of its high predictive skill for rare events,
ability to model non-linear relationships, and insensitivity to collinear or unimportant predictor variables
relative to classical regression techniques such as generalized linear models or generalized additive

models (Cutler et al., 2007; Prasad et al., 2006). All analysis was done in R (R Core Team, 2024) and all code and non-confidential data are available on GitHub here: [add link post double-blind peer review].

2.2 Study area

Our study area spans southern and central California, U.S.A. from the U.S.-Mexico border (32°N) to Point Reyes (38°N), which is just north of San Francisco Bay (**Fig. 1A**). The study area falls within the California Current Ecosystem, a highly productive eastern boundary upwelling system spanning from the southern tip of the Baja Peninsula, Mexico (28°N) to the U.S.-Canada border (48°N). Seasonal upwelling of cold, nutrient-rich water fuels populations of krill, squid, sardines, and other low trophic species that are fed upon by larger fishes, sea turtles, seabirds, and marine mammals. The ecosystem also supports important commercial and recreational fisheries and other human uses that benefit tens of millions of people living along the U.S. West Coast. We assessed bycatch within seven regions of the study area, which are shown in **Fig. 1A** and explained in detail in section 2.5.

2.3 The fishery

We defined the fishery using the definition in the MMPA List of Fisheries (NOAA, 2023)(NOAA, 2023): the ≥3.5 inch mesh set gillnet fishery targeting California halibut, white seabass, Pacific angel shark, and other species. Although this definition deviates from historical studies, which frequently focused on the portion of the fishery using mesh sizes larger than 8.0 or 8.5 inches (Table 4S1), the MMPA definition provides the legal basis for bycatch management and is more consistent with historical regulations. Specifically, a minimum mesh size of 3.5 inches was set for white seabass in 1941, though it was increased to 6.0 inches in 1988 (FGC §8623(d)). Since 1989, California halibut and Pacific angel shark have been targeted using a minimum mesh size of 8.5 inches (FGC §8625(a)). The set gillnet fishery principally excluded by this definition is that for Pacific herring (*Clupea pallasi*), which occurs in California’s four largest herring spawning areas — San Francisco Bay, Tomales Bay, Humboldt Bay, and Crescent City Harbor — using mesh sizes of 2.0 to 2.5 inches (CDFW, 2019).

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2.24 Data

Our analysis relies on two fisheries-dependent datasets: logbook data and observer data. All gillnet vessels are required to submit logbooks documenting when, where, and how they fished and how much catch was retained. Thus, logbooks characterize all fishing trips. However, because logbooks are self-reported and reported discards are unverifiable, logbooks likely underreport discarded bycatch, especially the bycatch of protected species. As a result, information from observer programs, which place

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11 185 trained observers on a sample of fishing trips (0-16.7% in this fishery; **Fig. 2B; Table S2**), are required to
12 inform estimates of bycatch for the unobserved trips recorded in vessel logbooks, which constitute the
13 majority of fishing effort (83.3-100% in this fishery; **Fig. 2B; Table S2**). Thus, the estimation of total
14 bycatch through ratio estimation depends on both the observer and logbook data. In contrast, the
15 evaluation of bycatch drivers and hotspots with the random forest models uses only the observer data, as
16 these are the only data to accurately record protected species bycatch when it occurs. These datasets are
17 described in detail below.
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19 2192 2.24.1 Observer data
20
21 193 We received observer data from 1983 to 2017 from the California Department of Fish and
22 Wildlife (CDFW). There was observer coverage in the California set gillnet fishery from 1983-1995
23 (coastwide), 1999-2000 (Monterey Bay area only), 2010-2013 (south of Point Conception only), and in
24 199 (2017 (south of Point Conception only) (**Fig. 2**). The observer program was run by CDFW from 1983-
25 1996 1989 and by the National Oceanic and Atmospheric Administration (NOAA) from 1990 onwards. **The**
26 1997 **percentage of annual fishing trips with onboard observers has varied over time, ranging from 0.3% of**
27 1998 **trips in 2006 to 16.7% of trips in 1993 (Fig. 2B; Table S2)**. Observers collected information on the
28 1999 amount and fate of catch (kept, discarded, or damaged), the length composition of the catch, the location
30 200 and time of the catch, and characteristics of the gear used to target the catch (**Fig. S2**). **In the state**
31 201 **observer data, we defined individual gillnet sets based on the date of fishing, the vessel, and the set**
32 202 **number and built a unique identifier to link set level information across state datasets (i.e., YYYY-MM-**
33 203 **DD-VesselID-Set#)**. **In the federal observer data, we defined individual gillnet sets based on the observer**
34 204 **trip number and the set number and built a unique identifier to link set level information across federal**
35 205 **datasets (i.e., TripID-Set#)**. **We developed a series of simple assumptions to impute missing values for a**
36 206 **few key variables (GPS coordinates, fishing depth, soak hour, mesh size) used to describe gillnet sets**
37 207 **documented in the observer data (Fig. S3; see supplemental methods for details)**.

4210 Although historical reports document low levels of observer coverage in Morro Bay, Monterey
43 211 Bay, and San Francisco in the 1980s (**Table 4S1**), the data that we received from CDFW excluded most
44 212 of these observations. We **digitized/recovered** a small portion of the missing raw data – observations from
45 213 Monterey Bay from 1987-1989 (**Fig. 2**) – from original CDFW data sheets that were given to a colleague
46 214 at the Southwest Fisheries Science Center during the late 1990s for a reanalysis of **historical**
47 215 bycatch rates in that region (**Forney et al., 2001**). **We extracted summaries of set-**
48 216 **level** bycatch rates from historical reports (**Table 4S1**) for years and regions missing raw data to support
49 217 the ratio estimation analysis (**Table S3**). **We converted set-level bycatch rates to trip-level bycatch rates**

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11 218 assuming an average of 3 sets per trip (Table S3), as indicated by the observer data (Fig. S4). Fig.
12 219 Figure 2C illustrates the coverage of the available, recovered, and lost observer data.

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14 221 We developed a series of simple assumptions to impute missing values for a few key variables
15 222 (GPS coordinates, fishing depth, soak hour, mesh size) reported in the observer data (Fig. S3A). We
16 223 assigned missing GPS coordinates using the median coordinates for observed trips within the statistical
17 224 block most frequently visited by the vessel – in order of preference – that week, month, or year based on
18 225 the logbook data (described below). We derived missing fishing depths by extracting depths from 25
19 226 meter resolution bathymetry data (CDFW, 2002) (Fig. S3B). We reassigned missing soak hours the mode
20 227 value for a vessel and target species (Fig. S3C). We reassigned missing mesh sizes the mode for – in order
21 228 of preference – the vessel and target species, the target species, or all vessels (Fig. S3DE). We assigned
22 229 each GPS coordinate to the nearest statistical reporting block (see Fig. 1A), which allows points
23 230 erroneously falling on land to be assigned a likely statistical block. We derived the distance from shore, a
24 231 covariate used to explain bycatch rates in the random forest model, as the distance of each set to the
25 232 nearest point on shore.

3 233
31 234 For each species of marine mammal, seabird, and sea turtle documented in the observer data, we
32 235 calculated the total number of captures observed in the California set gillnet fishery (Fig. 2). Throughout
33 236 this analysis, we focus on the six species with more than 50 observations: Common murre
34 237 (2,381), California sea lion (1,372), harbor seal (519), Brandt's cormorant (118), Northern elephant seal (78),
35 238 and harbor porpoise (97). Unfortunately, this excludes southern sea otter, which was a species of significant conservation concern in the 1980s, but whose bycatch was only
36 239 documented in the lost observer data.

4 240
41 241 2.2 Logbook data

42 242 We received logbook data from the commercial gillnet fishery from 1981 to 2022 from
43 243 CDFW (Fig. 1). All California gillnet vessels are required to submit logbooks documenting all of their
44 244 fishing trips; as a result, these logbooks represent all fishing effort associated with the fishery. These data
45 245 describe vessel information (vessel name, unique identifier, permit number); when (date), where
46 246 (statistical reporting block id; Fig. 1A), and how long (hours) a vessel fished; what fish it targeted; what
47 247 type of gear it used (drift or set gillnet) and characteristics of this gear (length, mesh size, fishing depth;
48 248 Fig. S2); what species it caught; and the amount (number and/or weight) and fate of this catch (kept,
49 249 released, or lost, including the identity of predators preying on released fish). We attempted to identify
50 250 individual fishing sets within the logbook data as the unique combination of vessel administration

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1 251 information (vessel name, either vessel id or boat number, permit number), where, when, and how long a
1 252 vessel fished (block id, date, and fishing hours), and characteristics of the gear (net length, mesh size, and
1 253 fishing depth). This analysis revealed an average of 1 set per trip, which is inconsistent with the 3 sets per
1 254 trip documented in the more accurate observer data. We term these unique identifiers “pseudo-sets” and
1 255 view them as roughly equivalent to a fishing trip (Fig. S4). ~~We derived the distance from shore, a covariate used to explain bycatch rates in the random forest model, as the median distance from shore of observed trips within the reported block given that exact locations are not reported.~~

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2 259 We developed a series of simple assumptions to impute missing or unrealistic values for a few
2 260 key variables (fishing depth, soak hour, mesh size) ~~reported in the logbook data (Fig. S5A)~~. We
2 261 reassigned both missing (including 0 values) and unrealistic fishing depths, which we defined as depths
2 262 exceeding the maximum depth in the reported fishing block, the median depth of the fishing block (Fig.
2 263 S5B). We computed the median and maximum depths of each fishing block using 25-meter resolution
2 264 bathymetry data (CDFW, 2002). We reassigned missing soak hours (including 0 values) the mode value
2 265 for a vessel. We capped rare and unlikely soak times exceeding 96 hours (4 days) at 96 hours; however,
2 266 such soak times could theoretically occur during rough weather when it is unsafe to haul gear (Fig. S5C).
3 267 We reassigned missing (including 0 values) and unrealistic mesh sizes, which we defined as mesh sizes
3 268 exceeding 20 inches, using a hierarchical procedure (Fig. S5D). For logbooks with both vessel
3 269 identification and target species information, we assigned the mesh size most commonly used by the
3 270 vessel when targeting that target species. For logbooks with only target species information (no vessel
3 271 identification), we assigned the mesh size most commonly used when targeting that target species across
3 272 all vessels (Figs. S5 & S6).
3 273 ~~used to describe gillnet pseudo-sets documented in the logbook data (Fig. S5 & S6; see supplemental
4 274 methods for details).~~

4 275 2.24.3 Sea surface temperature

4 276 Because sea surface temperature (SST) is a common driver of the distributions of both target and
4 277 bycatch species (Hazen et al., 2018)(Hazen et al., 2018), we used SST as an environmental covariate in
4 278 the random forest models described below. We derived the SST associated with each set documented in
4 279 the observer and ~~trip documented in the~~ logbook data using the NOAA 1/4° Daily Optimum Interpolation
4 280 Sea Surface Temperature (OISST) dataset, which interpolates observations from different monitoring
5 281 platforms (e.g., satellites, ships, buoys, and Argo floats) to provide a globally complete grid of SST from
5 282 September 1, 1981 to present (Huang et al., 2021)(Huang et al., 2021). For sets reported in the observer

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1283 data, we extracted the SST at the reported GPS location on the reported day of fishing. For ~~setstrips~~
1284 reported in logbooks, we calculated the average SST in the reported block on the reported day of fishing.
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15285 2.35 Ratio estimation
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17286 We estimated annual bycatch for each study species using ratio estimators. Ratio estimators
18287 assume that the rate of bycatch in observed fishing trips is proportional to the rate of bycatch within all
19288 fishing trips within a given stratum (Cochran, 1977). This assumption requires that the
20289 characteristics of observed trips do not systematically differ from the characteristics of all trips, which
21290 was confirmed by a two-sided Kolmogorov-Smirnov test (Fig. for six key traits (i.e., day of year, depth,
22291 latitude, mesh size, distance from shore, soak time) (Fig. S7). We used trips rather than sets as the
23292 sampling unit given the inability to identify unique sets in the logbooks logbook data (Fig. S4B). This is
24293 valid because the number of gillnet sets per fishing trip (median: 3 sets/trip; interquartile range: 2-4
25294 sets/trip) has been consistent through time (Fig. S4A).
26 Under this approach, the bycatch rate for
27295 species s in stratum i ($r_{s,i}$) – where, in this case, strata are defined by years and regions (see next
28296 paragraph) – is thus calculated as:
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$$r_{s,i} = \frac{k_{s,i}}{d_{s,i}}$$

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35300 Where $k_{s,i}$ is the total number of individuals of species s captured in observed trips occurring in strata i
36301 and $d_{s,i}$ is the total number of observed trips occurring in strata i . The total estimate of bycatch of species s
37302 in strata i ($m_{s,i}$) is then calculated by multiplying the bycatch rate ($r_{s,i}$) by the total number of trips to have
38303 occurred in the strata (D_i):
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$$m_{s,i} = D_i * r_{s,i}$$

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44307 Where the total number of trips (D_i) is derived from the logbook data.
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46309 We calculated annual bycatch estimates using a seven-region stratification scheme (Figs. 1A, S8).
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48310 This stratification scheme combines the scheme by Diamond and Hanan (1986) and Julian (1993) for
49311 areas north and south of Point Conception, respectively. Although early efforts to estimate bycatch in the
50312 California set gillnet fishery often stratified estimates by region and season (Table 4S1), later efforts
51313 found that observer coverage was often too limited to employ complex temporal stratification and that
52314 estimates between temporally stratified and unstratified approaches were generally similar (Table 4S1).
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11 1315 Stratum-specific bycatch rates for years without observer coverage in the stratum are borrowed from the
12 1316 closest year (forwards or backwards) with observer coverage in the stratum (**Fig. 2C & S9**~~+~~**), as has been**
13 1317 **the practice in previous studies.** We collated annual bycatch estimates from past studies (**Table 4S1**) for
14 1318 comparison with our updated estimates (**Figs. S10-S12**).
15 1319

16 1320 Although there are methods for estimating the uncertainty of bycatch estimates generated through
17 1321 ratio estimation ([Julian and Beeson, 1998](#))[\(Julian and Beeson, 1998\)](#), we were unable to implement these
18 1322 methods because they rely on bootstrap procedures that sample from the bycatch rates of observed trips.
19 1323 Because these procedures require raw observer data, we cannot use them for (1) ~~species strata~~-years
20 1324 where summary values from historical reports are used because the raw data have been lost or (2) ~~species-~~
21 1325 ~~strata~~-years without observer data. [from within one of the fished strata. As a result, only 6 of the 42](#)
22 1326 [evaluated years had the data required to estimate uncertainty: 2006, 2007, 2010, 2011, 2012, and 2017](#)
23 1327 (**Fig. 1AC; Fig. 2BC**). An exploration of the uncertainty estimates [from generated in historical reports](#)
24 1328 [with access to the lost data](#) (**Table 4**) [suggestS1 suggests](#) that the median coefficient of ~~estimate variation~~
25 1329 for [estimates of](#) annual bycatch estimates ranges from [a low of](#) 0.14 for harbor seal to [a high](#) 0.47 for
26 1330 harbor porpoise ([Figure](#)**Fig. S10B**).
27 1331

28 1332 [We evaluated the sustainability of recent estimated marine mammal bycatch by comparing it to](#)
29 1333 [the potential biological removal \(PBR\) for each stock, which is defined under the MMPA as the](#)
30 1334 [maximum number of animals, not including natural mortalities, that may be removed from a marine](#)
31 1335 [mammal stock while allowing that stock to reach or maintain its maximum sustainable population. We](#)
32 1336 [extracted each PBR from its most recent stock assessment \(**Fig. 7**\) and compared it to the average](#)
33 1337 [estimated catch over the last 10 years \(2013-2022\). A fishery is managed based on its classification into](#)
34 1338 [one of three categories: Category I fisheries cause annual mortality and serious injury \(M/SI\) greater than](#)
35 1339 [50% of the PBR, Category II fisheries cause annual M/SI between 1 and 50% of the PBR, and Category](#)
36 1340 [III fisheries cause annual M/SI less than 1% of the PBR. A fishery is considered to be approaching the](#)
37 1341 [MMPA's "zero mortality rate goal" \(ZMRG\) when annual M/SI is below 10% of the PBR.](#)
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11 342 2.46 Random forest modeling
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14 343 2.6.1 Model training
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16 344 We used random forest classification models trained on the observer data to identify
17 345 drivers of bycatch risk for each of the six study species. 2.4.1 Model training
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19 346 We used random forest classification models trained on the observer data to identify drivers of
20 347 bycatch risk for each of the six study species. We used a random forest approach, a machine learning
21 348 method that ensembles predictions from hundreds of decision trees, rather than a classical regression
22 349 method (e.g., generalized linear or additive models) because of their comparatively high predictive skill
23 350 for rare events, ability to model non-linear relationships, and insensitivity to collinear or unimportant
24 351 predictor variables (Cutler et al., 2007; Prasad et al., 2006). We considered nine attributes of fishing as
25 352 potential drivers of bycatch risk: haul depth (fathoms), mesh size (inches), soak time (hours), latitude
26 353 ($^{\circ}$ N), longitude ($^{\circ}$ W), distance from shore (km), Julian day, sea surface temperature ($^{\circ}$ C), and whether the
27 354 fishing occurs near an island (i.e., within 10 km of island coast). These attributes were selected based on
28 355 their demonstrated relationship to bycatch risk in other papers (e.g., (Bettoli and Scholten, 2006; Bjørge et
29 356 al., 2013; Kroetz et al., 2020)) and their availability in the observer data or their ability to be derived
30 357 through remote sensing (i.e., distance from shore, temperature, island area). They represent a range of
31 358 spatial (latitude, longitude, distance from shore, depth, island area), temporal (Julian day), environmental
32 359 (temperature), and fishing-related (soak time, mesh size) attributes. For each species, we classified an
33 360 observed set as having (1) or not having (0) bycatch, and trained a classification model assuming a
34 361 Bernoulli distribution in the response variable.
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41 363 Because bycatch of protected species is rare, observed fishing sets show strong class imbalance
42 364 towards sets without bycatch compared to sets with bycatch. To illustrate, the percent of observed sets
43 365 with bycatch is as follows: California sea lion (1.02%), common murre (0.43%), harbor seal (0.43%),
44 366 Brandt's cormorant (0.09%), northern elephant seal (0.076%), and harbor porpoise (0.074%). Therefore,
45 367 without a proper sample balancing method, predictions are likely to be biased towards the majority class
46 368 (sets without bycatch), leading to an underestimation of bycatch risk. For this reason, we considered four
47 369 approaches for accounting for class imbalance resulting from bycatch rarity.
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51 371 The first three approaches employ different sample balancing methods: (1) downsampling, (2)
52 372 upsampling, and (3) synthetic minority over-sampling (SMOTE), which uses a mixture of down and
53 373 upsampling ([More and Rana, 2017](#)). The downsampling approach randomly
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1374 removes observations of the majority group (sets without bycatch) to obtain equal representation of the
1375 majority and minority (sets with bycatch) group. The upsampling approach randomly samples
1376 observations from the minority group with replacement to obtain equal representation with the majority
1377 group. The synthetic minority over-sampling (SMOTE) approach both up-samples the minority group and
1378 down-samples the majority group. It up-samples the minority class by synthesizing new cases from its
1379 nearest five neighbors and down-samples the majority class by randomly drawing samples from that
1380 group. We created each balanced dataset using the *themis* package in R ([Hvitfeldt, 2023](#))[\(Hvitfeldt, 2023\)](#)
1381 and fit random forest models to these datasets using the *randomForests* package in R ([Liaw and Wiener,](#)
1382 [2002](#))[\(Liaw and Wiener, 2002\)](#).
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384 The fourth approach employs weighted random forests, which use observation weighting rather
385 than sample balancing to elevate the importance of the minority class. In this “cost-sensitive” learning
386 approach ([More and Rana, 2017](#))[\(More and Rana, 2017\)](#), higher weights are assigned to minority
387 observations so that the model receives a higher penalty for misclassifying these observations, helping to
388 reduce bias towards the majority class. We evaluated multiple weighting schemes to optimize the
389 predictive skill of this approach. Specifically, we assigned majority observations (sets without bycatch) a
390 weight of 1 and assigned minority observations (sets with bycatch) weights of 25 to 200 in increments of
391 25. Thus, a total of eight candidate weighted random forest models were evaluated as described below.
392 We fit the weighted random forest model using the *ranger* package in R ([Wright and Ziegler,](#)
393 [2017](#))[\(Wright and Ziegler, 2017\)](#).

395 We trained each of the eleven candidate models (three balanced random forest models, eight
396 weighted random forest models) on 80% of the observer data, withholding the remaining 20% for model
397 testing. In training the models, we performed a grid search to identify the “mtry” hyperparameter – the
398 number of variables to randomly sample as candidates at each node split – that maximizes Cohen’s kappa
399 under 10-fold cross validation (**Fig. S13**). While accuracy measures the proportion of correctly classified
400 categorizations, Cohen’s kappa measures the proportion of correct classifications while accounting for the
401 probability of being correct by chance and is a better measure of predictive skill, especially for
402 imbalanced datasets ([Cohen, 1968](#))[\(Cohen, 1968\)](#). Although there are no definitive rules for interpreting
403 Cohen’s kappa, general guidelines suggest that values above 0.7 are ‘[excellent](#)’, ‘[excellent](#)’, 0.4-0.7 are
404 ‘[good](#)’, ‘[good](#)’, 0.2-0.4 are ‘[fair](#)’, ‘[fair](#)’, and below 0.2 are ‘[poor](#)’[“poor”](#) ([Fleiss et al., 2013; Landis and](#)
405 [Koch, 1977](#))[\(Fleiss et al., 2013; Landis and Koch, 1977\)](#). We identified the best fitting model as the
406 model generating the highest Cohen’s kappa on the training data. We applied this model to the test dataset
407 for an independent evaluation of its predictive power. We only evaluated four species (California sea lion,

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11 common murre, harbor seal and northern elephant seal) whose best models exhibited “fair” or better
12 performance in their training dataset and close to “fair” performance on the test dataset for the rest of the
13 analysis (**Table 21**).
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16 **2.46.2 Model evaluation and prediction**
17

18 We evaluated the drivers of bycatch risk for each species by inspecting the variable importance
19 and the marginal effects of each variable as estimated in the best fitting model. Variable importance was
20 evaluated as the total decrease in node impurities from splitting on the variable averaged over all trees.
21 The impurity measure is corrected when building the model to reduce its bias towards continuous
22 variables ([Nembrini et al., 2018](#)) ([Nembrini et al., 2018](#)). Marginal effects measure the impact of the
23 changes in one variable on the response variable while all other variables are held constant. The marginal
24 effects plots provide the scientific basis for our discussions of management regulations that could
25 effectively and efficiently reduce bycatch risk.
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30 ~~We used the best fitting model to generate annual estimates of protected species bycatch
31 from 1981 to 2022 by predicting whether “pseudo sets” recorded in logbooks were likely to have
32 captured each study species.~~
33 **2.6.3 Mapping spatial bycatch risk**

34 ~~We predict to pseudo sets rather than trips because the random forest model is trained on set level
35 covariates in the observer data. We used the best fitting model for each species to estimate the probability
36 that a logged pseudo set included bycatch of a species then categorized the pseudo set as with or without
37 bycatch using a species specific probability threshold. We derived the species specific probability
38 thresholds as the threshold that maximizes Cohen’s kappa when applied to the training datasets (Fig.
39 S14). We selected the probability threshold based on Cohen’s kappa rather than the area under the
40 receiver operator curve (AUC) because (1) the models were tuned and selected based on Cohen’s kappa
41 and (2) simulation work shows that deriving thresholds based on AUC tends to overestimates the
42 prevalence of rare events while it underestimates the prevalence of common events (Freeman and Moisen,
43 2008; Manel et al., 2001). We summed the number of pseudo sets predicted to have bycatch each year,
44 converted this sum to “true sets” assuming three sets per pseudo set (Fig. S4AB), and multiplied this sum
45 by the median number of captures when a capture occurs to generate estimates of the total number of
46 captured animals (Fig. S4C). We opted not to employ a more complex two stage or hurdle model
47 approach, where a second model estimates the number of captured individuals when bycatch occurs,
48 given the rarity of bycatch events larger than one for all species but common murre (Fig. S4C).~~
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To generate maps of spatial bycatch risk, we used the best fitting model to predict risk to a 0.02° grid spanning southern California (32-35°N and 117-121°W), the only area where the fishery can operate under current regulations. Conceptually, our metric of spatial bycatch risk represents the probability of bycatch at a given location under recent average conditions. We derived this metric by first predicting the probability of bycatch on every calendar day (Julian day 1 to 365) for each grid cell. The ~~eight~~^{nine} input variables for making these predictions were derived as follows: (1) *latitude*, (2) *longitude*, (3) *distance from shore*, ~~and~~(4) *depth*, ~~and~~(5) *island proximity* (i.e., whether the location was within 10 km of an *island*) were derived based on the centroid of the grid cell; (6) *soak time* was set to 24 hours, the logbook mode; (7) *mesh size* was set to 8.5 inches, the logbook mode; (8) *Julian day* was the input day; ~~and~~ (9) *sea surface temperature* (SST) was the average SST for that day and grid cell from 2010-2022. We then calculated the weighted average of the daily risk weighting by the amount of logbook entries (a metric of fishing effort) occurring on that Julian day from 2010-2022. We cropped the predictions to areas within 20 km of shore as this is the farthest offshore that the fishery has operated (Fig. S7). We delineated bycatch hotspots as areas falling within the 95th (California sea lion, harbor seal) or 99th (common murre, northern elephant seal) percentile contour of spatial risk for each species. A higher threshold was used for common murre and northern elephant seal because they exhibit a large number of very low risk cells.

2.6.4 Bycatch estimation

We explored using the random forests models to estimate annual bycatch (1981-2022) in the fishery but found them to be unsuitable for this specific case study. Although model-based approaches generally perform better than design-based approaches at estimating bycatch (reference), we found that our random forest models underpredicted bycatch risk in the 1980s and 1990s relative to the ratio estimator (Fig. S14). This is most likely because of the loss of observer data from the northern strata during this time period, fishery largely operated in the northern portion of the region. See the supplemental information for details on the estimation of bycatch using the random forest models.

3. Results

3.1 Ratio estimation

In general, estimated bycatch peaked in the mid-1980s, steadily declined following the 40-fathom depth restriction implemented in 1987, with a temporary increase in the late-1990s followed by continued decline (Fig. 3). Bycatch peaked, in order of decreasing magnitude, at 5,059 common murre in 1984, 3,437 California sea lion in 1987, 2,605 harbor seal in 1986, 560 harbor porpoise in 1985, 453 Brandt's

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11 470 cormorant in 1991, and 432 northern elephant seal in 1986 (Fig. 3). This pattern reflects trends in fishing
12 effort, which also declined after a peak in 1985, with a brief expansion in the late-1990s followed by
13 continued decline (Fig. 1C). Estimated bycatch of Brandt's cormorant follows a similar pattern but lags
14 behind the patterns for other species: estimated bycatch precipitously declines after a peak in 1989 with a
15 temporary increase occurring in the early 2000s (Fig. 3). This pattern is driven by a steep increase in
16 Brandt's cormorant bycatch rates in the Channel Islands region in 1990 that is assumed to have persisted
17 to today (Fig. S9). None of the other species experienced such a pronounced and impactful change in
18 bycatch rates (Fig. S9). Estimated bycatch of harbor porpoise~~harbor porpoise~~, northern elephant seal, and
19 common murre declined especially sharply following the 2002 exclusion of fishing from waters shallower
20 than 60 fathoms. ~~This regulation effectively closed the fishery in Monterey Bay and Morro Bay and~~
21 ~~restricted it to only Southern California~~. ~~Slight differences between our estimates of annual bycatch and~~
22 ~~those from historical studies (Fig. S11)~~ are driven by a mixture of differences in our methods and input
23 data (Fig. S12 Fig. 1A), where harbor porpoise do not occur and where common murre occur only in
24 winter in low densities (Fig. S15).

25 484
26 485 The sustainability of recent estimated marine mammal bycatch ~~can be weighed against their was~~
27 486 ~~evaluated as a percentage of the~~ potential biological removal (PBR) ~~for~~ each stock, which is defined
28 487 under the MMPA as the maximum number of animals, not including natural mortalities, that may be
29 488 removed from a marine mammal stock while allowing that stock to reach or maintain its maximum
30 489 sustainable population (Fig. 7). Based on this sustainability reference point, bycatch concerns, in order of
31 490 decreasing threat, are as follows: harbor seal (65 individuals per year = 4.0% of a PBR of 1,641
32 491 individuals), harbor porpoise (1 individual per year = 1.5% of a PBR of 65 individuals in the Morro Bay
33 492 stock, where the bycatch took place), California sea lion (194 individuals per year = 1.4% of a PBR of
34 493 14,011 individuals), and northern elephant seal (7 individuals per year = 0.1% of a PBR of 5,122
35 494 individuals) (Fig. 7). The assessment that bycatch during the last 10 years poses the greatest risk to harbor
36 495 seals is supported by the fact that the harbor seal stock size has been stable or declining in recent years
37 496 while all of the other marine mammal populations have been undergoing sustained population growth
38 497 (Fig. 7). The sustainability of estimated seabird bycatch is more difficult to evaluate given more limited
39 498 population monitoring data (Fig. 7) and the lack of legally binding reference points for defining allowed
40 499 incidental take. However, increasing Brandt's cormorant nests from 1980 to 2020 (Fig. 7) and steeply
41 500 reduced bycatch of common murre (Fig. 3) suggests low risks posed to these species.

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43 502 — Our estimates of annual bycatch are generally aligned with estimates from historical
44 503 studies (Fig. S11). Slight differences between our estimates and those from historical studies are driven
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1504 by a mixture of differences in our methods and input data. While we apply a consistent approach for
1505 defining the fishery, stratifying the data, and estimating bycatch, historical studies have employed
1506 variable fishery definitions, stratification schemes, and estimation methodologies (Table 1). First, we
1507 consistently defined the fishery as using ≥ 3.5 " set gillnets, while historical studies have considered set
1508 gillnets ≥ 8 ", ≥ 8.5 ", or of unspecified sizes (Table 1). Furthermore, it is unclear whether historical studies
1509 filtered out fishing sets based on the reported target species and whether this decision was consistent. We
1510 do not define the fishery based on reported target species given extreme heterogeneity in how this
1511 information is reported. Differences in fishery definitions, as well as differences in data cleaning methods,
1512 likely lead to the slight differences in effort and observed bycatch attributed to the fishery (Fig. S12),
1513 which inevitably causes differences in bycatch estimates. Second, we used a consistent six region
1514 stratification scheme, while historical studies used a mixture of stratification schemes ranging from no
1515 stratification to spatial stratification to spatial temporal stratification (Table 1). This impacts the ratio
1516 estimators and the magnitudes of bycatch. Finally, we used vessel days (trips) as the sample unit, whereas
1517 historical studies oscillated between trips and sets as the preferred sample unit (Table 1). This is
1518 problematic not only because of the lack of consistency but also because sets cannot be uniquely
1519 identified in the logbook data.

32 520 3.2 Random forest modeling

33 521 3.2.1 Model performance

36 522 The best-fitting model performed well for four of the six study all species: common murre, harbor
37 523 seal, California sea lion, and northern elephant seal. All four species utilized weighted random forests but
38 524 with different case weights (Table 2; Fig. , except S13). The models for two species, Brandt's Brandt's
39 525 cormorant and harbor porpoise, which exhibited poor performance (Cohen's kappa less than 0.2) and were
40 526 therefore excluded from further consideration because of their poor performance (Table 2). Weighted
41 527 random forest models performed best for California sea lion, common murre, northern elephant seal, and
42 528 harbor seal with case weights of 25, 25, 25, and 75, respectively (Table 1; Fig. S13). This highlights
43 529 the importance of evaluating multiple modeling approaches when predicting rare bycatch events. For
44 530 common murre, Cohen's kappa was 0.71, indicating "good" performance, while for harbor seal,
45 531 California sea lion, and northern elephant seal, Cohen's kappa was 0.25, 0.24, and 0.23, respectively,
46 532 indicating "fair" performance. Model performance was strongly positively correlated with the frequency
47 533 of bycatch observations (Table 2). The best fitting model for common murre, the most common bycatch
48 534 species, exhibited a , i.e., species with more observed bycatch events produced models with greater skill
49 535 (Table 1; $r^2 = 0.64$ for Cohen's kappa value greater than 0.4 indicating "good" performance while the best
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11 536 models for harbor seal and California sea lion, somewhat common bycatch species, exhibited values
12 greater than 0.2 indicating “fair” performance. The best fitting models for Brandt’s cormorant and harbor
13 porpoise, rare bycatch species, exhibited training data). Cohen’s kappa values less than 0.2 on the test
14 dataset, indicating “poor” performance, and were not evaluated any further. Although northern elephant
15 seal bycatch is also rare, its best fitting model exhibited “fair” performance, and it was evaluated further.
16 Cohen’s kappa values were positively correlated with the area under the receiver operator curve
17 (AUC) ($r^2 = 0.68$ for the training dataset), indicating minimal tradeoffs in using this metric for model
18 selection (Table 2).
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23 544 3.2.2 Drivers of bycatch risk

24 545 The importance of the evaluated explanatory variables in determining bycatch risk varied by
25 546 species but some general patterns emerged (Fig. 4). In general, spatial (latitude, longitude, depth, and
26 547 distance from shore) and temporal (Julian day) variables were more influential than variables associated
27 548 with the environment (sea surface temperature) or the fishing methodology (soak time, mesh size).
28 549 Whether fishing occurred close to an island (a spatial variable) was the exception, as it was consistently
29 550 the least important variable. Sea surface temperature, which is closely related to space and time, was
30 551 generally more important than soak time and always more important than mesh size. These results suggest
31 552 that spatial temporal management may have better ability to manage bycatch risk than gear modifications
32 553 or soak time regulations, though shorter soak times could reduce discard mortality for non-target fish
33 554 species.

34 555
35 556 The species exhibit a mixture of correlated similar and non-correlated dissimilar responses to the
36 557 explanatory variables (Fig. 5). California sea lion and harbor seal exhibit similar responses in bycatch
37 558 risk. Both species have higher bycatch risk in shallower depths in nearshore areas with a spike in risk
38 559 occurring around 34°N latitude, including some deeper offshore areas (Fig. 6). They also exhibited a
39 560 pronounced increase in risk during the spring, lasting approximately from Apr 1 (90th day of the year) to
40 561 June 15 (166th day of the year). They are infrequently caught in nets with mesh sizes smaller than 8.5
41 562 inches, though the use of such nets is rare (Fig. S5E). Variability in bycatch risk for common murre and
42 563 northern elephant seal is most strongly determined by latitude and longitude (Fig. 5), with the only area of
43 564 elevated risk in contemporary fishing grounds occurring just north of Point Conception (Fig. 6). For all
44 565 four species, bycatch risk exhibits an asymptotic relationship with soak time and a dome-shaped
45 566 relationship with temperature; however, though the shapes of these relationships differ by species. The
46 567 species exhibit complex and variable relationships to temperature.

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1568 3.2.3 Maps of bycatch risk

1569 The species exhibited different patterns of spatial bycatch risk. California sea lion bycatch risk is
1570 predicted to be highest in four areas: (1) on the northern coasts of the northern Channel Islands, especially
1571 on the northern coast of Santa Rosa; (2) a small nearshore area west of Santa Barbara; (3) the eastern
1572 coast of Santa Cruz Island; and (4) the northwestern shores of Santa Catalina and San Clemente Islands;
1573 (Fig. 6). The high risk around the northern Channel Islands is likely related to the large haulouts of sea
1574 lions in that area (Fig. S15).

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1576 6. Harbor seal bycatch risk is predicted to be highest in four areas: (1) the sliver of nearshore
1577 area stretching from Santa Barbara to Point Sal; (2) the eastern coasts of Santa Cruz Island; (3) a broad
1578 coastal area near Point Mugu; and (4) the sliver of nearshore area stretching from Point Mugu to Point
1579 Vicente (Fig. 6). Most of these areas correspond to the locations of large harbor seal haulouts on Santa
1580 Cruz Island and near Point Mugu (Fig. S15).

1581
1582 6. Common murre bycatch risk is predicted to be negligible throughout most of southern
1583 California (Fig. 6). It is only predicted to be high in a small patch near Point Sal and even there, the
1584 maximum risk index is much lower than for the other evaluated species. This is consistent with the
1585 distribution of the species, which has no breeding colonies or permanent foraging grounds in southern
1586 California (Fig. S15). The absence of common murre in southern California largely explains the
1587 significant drop in common murre bycatch (Fig. 3) since the fishery was pushed out of central California
1588 (Fig. 1).

1589
1590 Like common murre, northern elephant seal bycatch risk is also predicted to be negligible
1591 throughout most of southern California except in the region near Point Sal (Fig. 6).

1592 Although northern elephant seals breed on the Channel Islands and near San Simeon/Cambria
1593 (Fig. S15) from December to March, they disperse to their distant foraging grounds (males to Alaska and
1594 females to oceanic waters far West of California) before the fishing season peaks from April to June,
1595 significantly reducing their vulnerability to the gillnet fishery.

1696 3.2.4 Temporal trends in estimated bycatch

1697 The random forest models estimate trends in bycatch that are similar to the estimates from the
1698 ratio estimator from 2000–2022 (Fig. 3). The agreement between these two different approaches
1699 underscores the result that bycatch has decreased to low levels as a result of management and reduced
1700 fishing effort. However, the estimates produced by the two approaches diverge from 1981–2000 by

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1101 various extents. While they generally agree for California sea lion back to 2000, the random forest model
1102 underpredicts bycatch relative to the ratio estimator in the late 1990s and overpredicts in the 1980s and
1103 early 1990s (Fig. 3). While the approaches generally agree for harbor seal back to 1995, the random forest
1104 model underpredicts bycatch relative to the ratio estimator before 1995, especially in the Channel Islands
1105 and Ventura strata (Fig. S16). For common murre, the random forest model overpredicts bycatch relative
1106 to the ratio estimator in the mid- to late 1990s and underpredicts relative to the ratio estimator in earlier
1107 years, especially in Morro and Monterey Bays. These underpredictions likely occur because of the
1108 unequal impacts of lost data from the northern strata in the 1980s (Fig. 2). While the ratio estimation
1109 method leverages summarized observer data from old reports, meaning that it sees data from this time
1110 period, the lack of the raw observer data from this time period means that the random forest model does
1111 learn from this period. As a result, it is likely to underpredict risk in early years in northern strata because
1112 it has largely learned from late years in southern California, where risk has been lower. For this reason,
1113 we recommend the use of bycatch estimates from the ratio estimator over the random forest model until a
1114 time when the 1980s observer data is rediscovered.

30615 4. Discussion

32616 Our study provides the first update to total estimates of protected species bycatch in the
33617 California set gillnet fishery since 2012 (Carretta et al., 2014). We find that bycatch, once high and
34618 unsustainable for some species (Forney et al., 2021, 2001) (Forney et al., 2021, 2001), is now well below
35619 the “zero mortality rate goal” (ZMRG) of 10% of the potential biological removal (50 C.F.R. § 229.2).
36620 Recent marine mammal bycatch estimates range from 0.1–4.0% of their potential biological removals and
37621 common murre bycatch has been effectively eliminated. All of the evaluated populations, including the
38622 once declining and heavily depleted Morro Bay harbor porpoise population, are growing or stable. These
39623 advances, while directly attributable to management interventions, are more due to reductions in fishing
40624 effort (i.e., fewer fishing trips) than to reductions in bycatch rates (i.e., lower bycatch per unit
41625 effort fishing trip). This highlights a steep tradeoff between conservation and fisheries objectives under the
42626 current management regime: while populations of protected species have undergone sustained growth,
43627 fishing opportunities and revenues have undergone sustained prolonged declines. Despite this, there have
44628 been calls for more bycatch-motivated restrictions to the fishery (Birch et al., 2023; Birch and Shester,
45629 2023) (Birch et al., 2023; Birch and Shester, 2023). Our results indicate that current fishing operations do
46630 not pose a threat to the evaluated species, which suggests that current management is sufficient at limiting
47631 bycatch. However, targeting management toward spatial-temporal bycatch hotspots could improve
48632 economic outcomes while keeping bycatch low.

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11633 Our results suggest that spatial-temporal management could more efficiently and effectively
12634 manage bycatch risk than gear modifications or soak time regulations. Specifically, bycatch rates for
13635 California sea lions and harbor seals are greatest from April 1 to June 15, suggesting that a 2.5 month
14636 seasonal closure of bycatch ~~spots for these two species could reduce bycatch risk. Spatially, our findings~~
15637 indicate that the existing state water closure (0–3 nautical miles offshore) in Southern California could be
16638 adjusted to target specific bycatch hotspots around the Channel Islands, where both California sea lions
17639 and harbor seals face higher bycatch risk compared to nearshore areas (except for the nearshore strip from
18640 Santa Barbara to Point Sal). This adjustment could potentially open up more fishing grounds in the
19641 nearshore region.~~hotspots for these two species could prevent bycatch while allowing the opening of less~~
20642 ~~risky but currently closed areas to fishing. These hotspots are predicted to disproportionately contribute to~~
21643 ~~bycatch yet are of minor fishing importance (Fig. 6B) suggesting that the loss of fishing opportunities in~~
22644 ~~these areas during brief seasonal closures could be easily made up by opening areas of low predicted~~
23645 ~~bycatch risk. As a result, seasonal closures could broaden fishing opportunities while continuing to meet~~
24646 ~~bycatch avoidance objectives.~~ However, we caution that such hotspot closures could exacerbate bycatch
25647 problems if fishing effort is displaced and concentrated in areas of secondarily high risk ([Free et al., 2023](#))
26648 ([Free et al., 2023](#)). Therefore, monitoring of fishing effort and bycatch rates are important for
27649 verifying that seasonal closures achieve their conservation and fisheries objectives. Additionally, any
28650 changes in current management strategies must take into account spatial-temporal patterns of bycatch and
29651 relative sensitivity of each species.

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31653
32654 Hotspots of bycatch risk are aligned with the location of known haulouts, breeding colonies, and
33655 foraging grounds. High California seal lion bycatch risk around the northern Channel Islands is likely
34656 related to the large haulouts of sea lions in that area (Fig. S15). Similarly, hotspots of harbor seal bycatch
35657 risk correspond to the locations of large harbor seal haulouts on Santa Cruz Island and near Point Mugu
36658 (Fig. S15). The absence of common murre bycatch risk in southern California is consistent with the
37659 distribution of the species, which has no breeding colonies or permanent foraging grounds in southern
38660 California (Fig. S15). Similarly, the negligible risk for northern elephant seals is consistent with the
39661 phenology of their migrations. Although northern elephant seals breed on the Channel Islands and near
40662 San Simeon/Cambria (Fig. S15) from December to March, they disperse to their distant foraging grounds
41663 (males to Alaska and females to oceanic waters far West of California) before the fishing season peaks
42664 from April to June, significantly reducing their vulnerability to the gillnet fishery. Finally, the elimination
43665 of harbor porpoise bycatch is consistent with the confinement of the fishery to southern California where
44666 harbor porpoise do not occur (Fig. S15).

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The completion and safekeeping of accurate fisheries-dependent data is fundamental to producing accurate bycatch assessments and effective management strategies. While summaries of historical data facilitated reliable bycatch estimates through ratio estimation, the loss of raw observer data from the 1980s likely impeded our ability to accurately estimate bycatch in the northern strata using random forests, an approach often thought to be more accurate than ratio estimation (Stock et al., 2019)(Stock et al., 2019). Notably, the lost data document a period when fishing was allowed in shallower, more inshore, and more northern waters (Fig. 1A). Recovering this data would enhance our ability to assess the drivers of bycatch in the northern region and could provide insights for re-evaluating previous management strategies. Furthermore, missing meta-data on critical gear characteristics (e.g., mesh size, net length, net height, net material; Fig. S3) in the available observer data also limited our ability to identify the potential for these management levers to reduce bycatch risk. Ensuring the complete documentation of gear characteristics, perhaps by prioritizing characteristics known to impact bycatch risk in other gillnet fisheries (Northridge et al., 2017)(Northridge et al., 2017), is important to maximizing the utility of expensive, and sometimes controversial, observer programs (Suuronen and Gilman, 2020).(Suuronen and Gilman, 2020). Finally, the ability to delineate individual sets in the logbooks and improved documentation of the characteristics of logged sets would enhance future bycatch estimates by allowing sets to be the sampling unit and by avoiding assumptions about missing data, respectively. This could be achieved by redesigning logbooks, training fishers on completing logbooks, expanding electronic monitoring, and/or demonstrating that better data can actually lead to fewer restrictions.

Our results highlight the importance of considering multiple modeling approaches when estimating and evaluating rare bycatch events. Although model-based methods (e.g., random forests) for estimating bycatch are often preferred to sample-based methods (e.g., ratio estimators) (Stock et al., 2019)(Stock et al., 2019), we find complementary value in using both approaches. While ratio estimation generated more reliable bycatch estimates due to its ability to leverage both raw and summarized data, the random forest model provided the empirical basis for assessing drivers of bycatch risk. Additionally, agreement between the two approaches over the past two decades reinforces predictions that recent bycatch levels have been low for the evaluated species. Furthermore, our results highlight the value of considering multiple sample balancing approaches when evaluating bycatch using model-based methods, as the specification of the best performing model varied by species. Recent efforts to estimate bycatch in West Coast fisheries using random forests have used only a single sample balancing technique (e.g., Carretta, 2023); we encourage future efforts to compare multiple approaches to optimize estimates of rare bycatch events (More and Rana, 2017). Finally, we highlight our derivation of model-specific probability

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11 701 thresholds for classifying logged fishing efforts as having or not having bycatch using Cohen's kappa as
12 702 being aligned with recommended best practices for classifying rare events (Freeman and Moisen, 2008;
13 703 Manel et al., 2001). Specifically, research shows that classification based on: (1) the default 0.5
14 704 probability threshold underestimates rare events, (2) true skill statistics overestimates rare events; and (3)
15 705 Cohen's kappa provides the least biased predictions. This best practice should also be considered in future
16 706 bycatch estimation efforts.(More and Rana, 2017).
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20 708 The continued recovery of protected species will require management of stressors besides
21 709 fisheries bycatch, some of which may present even larger threats (Avila et al., 2018; Oldach et al.,
22 710 2022);(Avila et al., 2018; Oldach et al., 2022). For example, in Summer 2023, there were over 1,000
23 711 statewide strandings of California sea lions and other pinnipeds attributed to domoic acid toxicosis
24 712 resulting from an intense bloom of harmful diatoms in the *Pseudo-nitzschia* genus (SCCOOS, 2023;
25 713 Smith et al., 2023);(SCCOOS, 2023; Smith et al., 2023). Harmful algal blooms are increasing in
26 714 frequency, duration, and intensity on the West Coast (Hallegraeff et al., 2021)(Hallegraeff et al., 2021) as
27 715 a result of ocean warming and eutrophication (McKibben et al., 2017)(McKibben et al., 2017) suggesting
28 716 that, in the long-term, curbing climate change and nutrient runoff may be the most important actions for
29 717 stemming mortality for recovering pinniped populations. Furthermore, harassment and shooting are some
30 718 of the most commonly observed sources of mortality and serious injury for California sea lions, harbor
31 719 seals, and northern elephant seals (Carretta, 2023) (Fig. S17)(Carretta, 2023) (Fig. S16), suggesting the
32 720 need for greater outreach and enforcement to prevent these gratuitous forms of mortality. Finally, it is
33 721 important to understand the bycatch contributions of other fisheries, many of which report higher levels of
34 722 observed bycatch (Fig. S17S16) yet are not as heavily prosecuted as the California set gillnet fishery.
35 723 Modeling studies similar to this one are needed to determine whether higher apparent bycatch in these
36 724 fisheries is due to higher observer coverage or higher bycatch rates. However, the sustained recovery of
37 725 the evaluated populations suggests that total bycatch across all fisheries is low.
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44 727 There will always be tradeoffs between maximizing fishing opportunities and minimizing bycatch
45 728 of protected species (Samhouri et al., 2021);(Samhouri et al., 2021). As a result, managers often seek to
46 729 implement regulations that maximize fishing outcomes while keeping bycatch below legally defined
47 730 sustainable reference points (Kirby and Ward, 2014);(Kirby and Ward, 2014). The identification of such
48 731 strategies is seldom straightforward and depends on substantial investments in data and scientific
49 732 enterprises. Notably, they depend on monitoring populations of protected species to support the
50 733 assessment of their status and levels of allowable incidental take and monitoring bycatch in key fisheries
51 734 to support assessments of total bycatch, drivers of bycatch, and the effectiveness of past management
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11 35 interventions (Kirby and Ward, 2014; Punt et al., 2021). (Kirby and Ward, 2014; Punt et al., 2021). In the
12 36 absence of such data, management must often be precautionary to ensure compliance with protected
13 37 species legislation (Punt et al., 2021)(Punt et al., 2021). We illustrate the potential return on investment of
14 38 supporting such scientific enterprises as our results show that past management interventions have been
15 39 successful at reducing bycatch in the California set gillnet fishery well below target levels, opening the
16 40 door for more efficient restrictions and negating the need for unnecessary precaution. The continued
17 41 demonstration that monitoring programs can generate better outcomes for businesses could facilitate
18 42 increased public support and funding to identify win-win scenarios for fisheries and conservation in more
19 43 fisheries.
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8
9
10
11 744 References
12

- 13 745 Avila, I.C., Kaschner, K., Dormann, C.F., 2018. Current global risks to marine mammals: Taking stock of
14 the threats. *Biol. Conserv.* 221, 44–58. <https://doi.org/10.1016/j.biocon.2018.02.021>
- 15 746 Barlow, J., Baird, R.W., Heyning, J.E., Wynne, K., Manville, A.M., Lowry, L.F., Hanan, D., Sease, J.,
16 Burkanov, V., 1994. A Review of Cetacean and Pinniped Mortality in Coastal Fisheries Along
17 the West Coast of the USA and Canada and the East Coast of the Russian Federation. *Rep. Int.
18 Whal. Comm.* 15, 405–426.
- 19 747 Bireh, C., Blacow Draeger, A., Geoff Shester, PH.D., Webb, S., Purcell, E., 2023. The Net Consequence:
20 Impacts of Set Gillnets on California Ocean Biodiversity. Zenodo.
21 748 <https://doi.org/10.5281/ZENODO.7971874>
- 22 749 Bireh, C., Shester, G., 2023. Underreporting of Marine Mammal Takes in the California Set Gillnet
23 Fishery Underscores the Need for Observers. *Oceana*.
- 24 750 Bjørge, A., Skern Mauritzen, M., Rossman, M.C., 2013. Estimated bycatch of harbour porpoise
25 (*Phocoena phocoena*) in two coastal gillnet fisheries in Norway, 2006–2008. Mitigation and
26 implications for conservation. *Biol. Conserv.* 161, 164–173.
27 751 <https://doi.org/10.1016/j.biocon.2013.03.009>
- 28 752 CA Secretary of State, 1990. Proposition 132. Marine Resources. Initiative Constitutional Amendment.
- 29 753 Cameron, G.A., Forney, K.A., 2000. Preliminary Estimates of Cetacean Mortality in California/Oregon
30 Gillnet Fisheries for 1999 (International Whaling Commission Working Paper No. SC/52/024).
- 31 754 Cameron, G.A., Forney, K.A., 1999. Preliminary Estimates of Cetacean Mortality in the California
32 Gillnet Fisheries for 1997 and 1998 (No. SC/51/04). Southwest Fisheries Science Center.
- 33 755 Capitolo, P.J., McChesney, G.J., Beechaver, C.A., Rhoades, S.J., Shore, A., Carter, H.R., Parker, M.W.,
34 Eigner, L.E., 2012. Breeding population trends of Brandt's and double crested cormorants, Point
35 Sur to Point Mugu, California, 1979–2011.
- 36 756 Carretta, J.V., 2023. Sources of human-related injury and mortality for U.S. Pacific West Coast marine
37 mammal stock assessments, 2017–2021. <https://doi.org/10.25923/QWF2-9B97>
- 38 757 Carretta, J.V., 2002. Preliminary estimates of cetacean mortality in California gillnet fisheries for 2001
39 (No. SC/54/SM12). International Whaling Commission, Scientific Committee, Shimonoseki,
40 Japan.
- 41 758 Carretta, J.V., 2001. Preliminary estimates of cetacean mortality in California gillnet fisheries for 2000
42 (SC/53/SM9). International Whaling Commission, Scientific Committee, London, UK., London,
43 UK.
- 44 759 Carretta, J.V., Chivers, S.J., 2004. Preliminary estimates of marine mammal mortality and biological
45

1
2
3
4
5
6
7
8
9
10
11 777 sampling of cetaceans in California gillnet fisheries for 2003 (No. SC/56/SM1).
12 778 Carretta, J.V., Chivers, S.J., 2003. Preliminary estimates of marine mammal mortality and biological
13 sampling of cetaceans in California gillnet fisheries for 2002 (No. SC/55/SM3).
14
15 780 Carretta, J.V., Enriquez, L., 2012a. Marine mammal and seabird bycatch in California gillnet fisheries in
16 2010 (Administrative Report No. LJ 12-01). NOAA Southwest Fisheries Science Center, La
17 Jolla, CA.
18
19 783 Carretta, J.V., Enriquez, L., 2012b. Marine mammal and seabird bycatch in California gillnet fisheries in
20 2011 (NOAA Technical Memorandum NMFS No. NOAA-TM-NMFS-SWFSC-500). NOAA
21 Southwest Fisheries Science Center, La Jolla, CA.
22
23 786 Carretta, J.V., Enriquez, L., 2009. Marine mammal and seabird bycatch in observed California
24 commercial fisheries in 2007 (Administrative Report No. LJ 09-01). NOAA Southwest Fisheries
25 Science Center, La Jolla, CA.
26
27 789 Carretta, J.V., Enriquez, L., Villafana, C., 2014. Marine mammal, sea turtle, and seabird bycatch in
28 California gillnet fisheries in 2012 (NOAA Technical Memorandum NMFS No. NOAA-TM-
29 NMFS-SWFSC-526). NOAA Southwest Fisheries Science Center, La Jolla, CA.
30
31 792 Carretta, J.V., Oleson, E.M., Forney, K.A., Muto, M.M., Weller, D.W., Lang, A.R., Baker, J., Hanson, B.,
32 Orr, A.J., Barlow, J., Moore, J.E., Brownell Jr, R.L., 2022. U.S. Pacific marine mammal stock
33 assessments: 2021 (NOAA Technical Memorandum No. NOAA-TM-NMFS-SWFSC-663).
34 NOAA Southwest Fisheries Science Center, La Jolla, CA.
35
36 796 Carter, H.R., 2001. Population Trends of the Common Murre (*Uria aalge californica*), in: Biology and
37 Conservation of the Common Murre in California, Oregon, Washington, and British Columbia
38 Volume 1: Natural History and Population Trends.
39
40 799 CDFW, 2023. Evaluating Bycatch in the California Halibut Set Gill Net Fishery. California Department
41 of Fish and Wildlife, Sacramento, CA.
42
43 801 CDFW, 2021. California Wildlife Habitat Relationship System.
44
45 802 CDFW, 2019. California Pacific Herring Fishery Management Plan.
46
47 803 CDFW, 2014. Harbor Seals [ds106] GIS Dataset.
48
49 804 CDFW, 2010. Seabird Colonies: California, 2010.
50
51 805 CDFW, 2002. Bathymetry Project: 25m bathymetry dataset.
52
53 806 Cochran, W.G., 1977. Sampling Techniques. John Wiley and Sons.
54
55 807 Cohen, J., 1968. Weighted kappa: Nominal scale agreement provision for scaled disagreement or partial
56 credit. *Psychol. Bull.* 70, 213–220. <https://doi.org/10.1037/h0026256>
57
58 809 Condylies, S., 2023. priceR: Economics and Pricing Tools.
59
60 810 Crowder, L.B., Murawski, S.A., 1998. Fisheries Bycatch: Implications for Management. *Fisheries* 23, 8–
61
62
63
64
65

- 1
2
3
4
5
6
7
8
9
10
11 17. [https://doi.org/10.1577/1548-8446\(1998\)023<0008:FBIFM>2.0.CO;2](https://doi.org/10.1577/1548-8446(1998)023<0008:FBIFM>2.0.CO;2)
- 12 Diamond, S.L., Hanan, D.A., 1986. An Estimate of Harbor Porpoise Mortality in California Set Net
13 Fisheries, April 1, 1983 through March 31, 1984 (Administrative Report No. SWR-86-16).
14 NOAA Southwest Fisheries Science Center, La Jolla, CA.
- 15 Fleiss, J.L., Levin, B., Paik, M.C., 2013. Statistical Methods for Rates and Proportions. John Wiley &
16 Sons.
- 17 Forney, K.A., Benson, S.R., Cameron, G.A., 2001. Central California gillnet effort and bycatch of
18 sensitive species, 1990–1998, in: Melvin, E., Parrish, J.K. (Eds.), Seabird Bycatch: Trends,
19 Roadblocks, and Solutions. Alaska Sea Grant, University of Alaska Fairbanks, pp. 141–160.
20 <https://doi.org/10.4027/sbtrs.2001.08>
- 21 Forney, K.A., Carretta, J.V., Benson, S.R., 2014. Preliminary estimates of harbor porpoise abundance in
22 Pacific Coast waters of California, Oregon and Washington, 2007–2012.
- 23 Forney, K.A., Moore, J.E., Barlow, J., Carretta, J.V., Benson, S.R., 2021. A multidecadal Bayesian trend
24 analysis of harbor porpoise (*Phocoena phocoena*) populations off California relative to past
25 fishery bycatch. Mar. Mammal Sci. 37, 546–560. <https://doi.org/10.1111/mms.12764>
- 26 Free, C.M., Bellquist, L.F., Forney, K.A., Humberstone, J., Kauer, K., Lee, Q., Liu, O.R., Samhour, J.F.,
27 Wilson, J.R., Bradley, D., 2023. Static management presents a simple solution to a dynamic
28 fishery and conservation challenge. Biol. Conserv. 285, 110249.
29 <https://doi.org/10.1016/j.biocon.2023.110249>
- 30 Freeman, E.A., Moisen, G.G., 2008. A comparison of the performance of threshold criteria for binary
31 classification in terms of predicted prevalence and kappa. Ecol. Model. 217, 48–58.
32 <https://doi.org/10.1016/j.ecolmodel.2008.05.015>
- 33 Geijer, C.K.A., Read, A.J., 2013. Mitigation of marine mammal bycatch in U.S. fisheries since 1994.
34 Biol. Conserv. 159, 54–60. <https://doi.org/10.1016/j.biocon.2012.11.009>
- 35 Hallegraeff, G.M., Anderson, D.M., Belin, C., Bottein, M. Y.D., Bresnan, E., Chinain, M., Enevoldsen,
36 H., Iwataki, M., Karlson, B., McKenzie, C.H., Sunesen, I., Pitcher, G.C., Provoost, P.,
37 Richardson, A., Schweibold, L., Tester, P.A., Trainer, V.L., Yñiguez, A.T., Zingone, A., 2021.
38 Perceived global increase in algal blooms is attributable to intensified monitoring and emerging
39 bloom impacts. Commun. Earth Environ. 2, 117. <https://doi.org/10.1038/s43247-021-00178-8>
- 40 Hanan, D.A., Diamond, S.L., 1989. Estimates of Sea Lion, Harbor Seal, and Harbor Porpoise Mortalities
41 in California Set Net Fisheries for the 1986–87 Fishing Year.
- 42 Hanan, D.A., Diamond, S.L., Scholl, J.P., 1988. Estimates of Sea Lion and Harbor Seal Mortalities in
43 California Set Net Fisheries for 1983, 1984, and 1985.
- 44 Hanan, D.A., Diamond, S.L., Scholl, J.P., 1987. An Estimate of Harbor Porpoise Mortalities in California
- 45
46
47
48
49
50
51
52
53
54
55
56
57
58
59
60
61
62
63
64
65

1
2
3
4
5
6
7
8
9
10
11 345 Set Net Fisheries, April 1, 1985 through March 31, 1986 (Administrative Report No. SWR 87-5).
12 346 NOAA Southwest Fisheries Science Center, La Jolla, CA.
13 347 Hanan, D.A., Diamond, S.L., Scholl, J.P., 1986. An Estimate of Harbor Porpoise Mortality in California
14 Set Net Fisheries April 1, 1984 through March 31, 1985 (Administrative Report No. SWR 86-
15 46). NOAA Southwest Fisheries Science Center, La Jolla, CA.
16 349 Hazen, E.L., Seales, K.L., Maxwell, S.M., Briscoe, D.K., Welch, H., Bograd, S.J., Bailey, H., Benson,
17 350 S.R., Eguchi, T., Dewar, H., Kohin, S., Costa, D.P., Crowder, L.B., Lewison, R.L., 2018. A
18 dynamic ocean management tool to reduce bycatch and support sustainable fisheries. *Sci. Adv.* 4,
19 352 eaar3001. <https://doi.org/10.1126/sciadv.aar3001>
20 353 Huang, B., Liu, C., Banzon, V., Freeman, E., Graham, G., Hankins, B., Smith, T., Zhang, H. M., 2021.
21 354 Improvements of the Daily Optimum Interpolation Sea Surface Temperature (DOISST) Version
22 355 2.1. *J. Clim.* 34, 2923–2939. <https://doi.org/10.1175/JCLI-D-20-0166.1>
23 356 Hvitfeldt, E., 2023. *themis: Extra Recipes Steps for Dealing with Unbalance Data. R package.*
24 357 ICES, 2007. Report of the Workshop on Discard Raising Procedures (ICES CM No. 2007ACFM:06).
25 358 ICES, San Sebastian, Spain.
26 359 Julian, F., 1994. Pinniped and cetacean mortality in California gillnet fisheries: preliminary estimates for
27 360 1993 (International Whaling Commission No. SC/46/0). NOAA Southwest Fisheries Science
28 361 Center, La Jolla, CA.
29 362 Julian, F., 1993. Pinniped and Cetacean Mortality in California Gillnet Fisheries: Preliminary Estimates
30 363 for April 1 To June 30, 1992 (Administrative Report No. LJ 93-19). NOAA Southwest Fisheries
31 364 Science Center, La Jolla, CA.
32 365 Julian, F., Beeson, M., 1998. Estimates of marine mammal, turtle, and seabird mortality for two
33 366 California gillnet fisheries: 1990–1995. *Fish. Bull.* 96, 271–284.
34 367 Kirby, D.S., Ward, P., 2014. Standards for the effective management of fisheries bycatch. *Mar. Policy* 44,
35 368 419–426. <https://doi.org/10.1016/j.marpol.2013.10.008>
36 370 Konno, E.S., 1990. Effort Estimates of Gill Net Fisheries in California that Incidentally Catch Marine
37 371 Mammals, for the 1987-88 Fishing Year. NOAA Southwest Fisheries Science Center, Terminal
38 372 Island, CA.
39 373 Laake, J.L., Lowry, M.S., DeLong, R.L., Melin, S.R., Carretta, J.V., 2018. Population growth and status
40 374 of California sea lions. *J. Wildl. Manag.* 82, 583–595. <https://doi.org/10.1002/jwmg.21405>
41 375 Landis, J.R., Koch, G.G., 1977. The Measurement of Observer Agreement for Categorical Data.
42 376 *Biometrics* 33, 159–174. <https://doi.org/10.2307/2529310>
43 377 Lennert, C., Kruse, S., Beeson, M., Barlow, J., 1994. Estimates of incidental marine mammal bycatch in
44 378 California gillnet fisheries for July through December, 1990 (Report of the International Whaling
45
46
47
48
49
50
51
52
53
54
55
56
57
58
59
60
61
62
63
64
65

- 1
2
3
4
5
6
7
8
9
10
11 379 Commission No. SC/43/O.3).
12 380 Lewison, R.L., Crowder, L.B., Wallace, B.P., Moore, J.E., Cox, T., Zydellis, R., McDonald, S., DiMatteo,
13 A., Dunn, D.C., Kot, C.Y., Bjorkland, R., Kelez, S., Soykan, C., Stewart, K.R., Sims, M.,
14 Boustan, A., Read, A.J., Halpin, P., Nichols, W.J., Safina, C., 2014. Global patterns of marine
15 mammal, seabird, and sea turtle bycatch reveal taxa-specific and cumulative megafauna hotspots.
16 Proc. Natl. Acad. Sci. 111, 5271–5276. <https://doi.org/10.1073/pnas.1318960111>
17 384 Liaw, A., Wiener, M., 2002. Classification and Regression by randomForest. R News 2, 18–22.
18 385 Long, C.A., Ahrens, R.N.M., Jones, T.T., Siders, Z.A., 2024. A machine learning approach for protected
19 species bycatch estimation. Front. Mar. Sci. 11. <https://doi.org/10.3389/fmars.2024.1331292>
20 386 Lopez, J., Griffiths, S., Wallace, B.P., Cáceres, V., Rodríguez, L.H., Abrego, M., Alfaro Shigueto, J.,
21 Andraka, S., Brito, M.J., Bustos, L.C., Cari, I., Carvajal, J.M., Clavijo, L., Coeas, L., Paz, N. de,
22 Herrera, M., Mangel, J.C., Pérez Huaripata, M., Piedra, R., Dávila, J.A.Q., Rendón, L., Ríquez,
23 Baron, J.M., Santana, H., Suárez, J., Veelenturf, C., Vega, R., Zárate, P., 2024. Vulnerability of
24 the Critically Endangered leatherback turtle to fisheries bycatch in the eastern Pacific Ocean. I. A
25 machine learning species distribution model. Endanger. Species Res. 53, 271–293.
26 393 <https://doi.org/10.3354/esr01288>
27 394 Lowry, M., Condit, R., Hatfield, B., Allen, S.G., Berger, R., Morris, P.A., Le Boeuf, B.J., Reiter, J., 2014.
28 Abundance, Distribution, and Population Growth of the Northern Elephant Seal (*Mirounga*
29 *angustirostris*) in the United States from 1991 to 2010. Aquat. Mamm. 40, 20–31.
30 398 <https://doi.org/10.1578/AM.40.1.2014.20>
31 399 Lowry, M.S., 2021. Abundance and distribution of pinnipeds at the Channel Islands in southern
32 California, central and northern California, and southern Oregon during summer 2016–2019
33 (NOAA Technical Memorandum NMFS No. NOAA TM NMFS SWFSC 656). NOAA
34 Southwest Fisheries Science Center, La Jolla, CA.
35 400 Manel, S., Williams, H.C., Ormerod, S. J., 2001. Evaluating presence-absence models in ecology: the
36 need to account for prevalence. J. Appl. Ecol. 38, 921–931. <https://doi.org/10.1046/j.1365-2664.2001.00647.x>
37 405 Martin, S.L., Stohs, S.M., Moore, J.E., 2015. Bayesian inference and assessment for rare event bycatch in
38 marine fisheries: a drift gillnet fishery case study. Ecol. Appl. 25, 416–429.
39 408 <https://doi.org/10.1890/14-0059.1>
40 409 McCracken, M.L., 2004. Modeling a Very Rare Event to Estimate Sea Turtle Bycatch: Lessons Learned
41 (NOAA Technical Memorandum No. NMFS-PIFSC-3). Pacific Islands Fisheries Science Center.
42 411 McKibben, S.M., Peterson, W., Wood, A.M., Trainer, V.L., Hunter, M., White, A.E., 2017. Climatic
43 regulation of the neurotoxin domoic acid. Proc. Natl. Acad. Sci. 114, 239–244.

- 1
2
3
4
5
6
7
8
9
10
11 913 <https://doi.org/10.1073/pnas.1606798114>
12 914 More, A.S., Rana, D.P., 2017. Review of random forest classification techniques to resolve data
13 915 imbalance, in: 2017 1st International Conference on Intelligent Systems and Information
14 916 Management (ICISIM). Presented at the 2017 1st International Conference on Intelligent Systems
15 917 and Information Management (ICISIM), IEEE, Aurangabad, pp. 72–78.
16 918 <https://doi.org/10.1109/ICISIM.2017.8122151>
17 919 Nembrini, S., König, I.R., Wright, M.N., 2018. The revival of the Gini importance? *Bioinformatics* 34,
18 920 3711–3718. <https://doi.org/10.1093/bioinformatics/bty373>
19 921 NOAA, 2024. List of Fisheries for 2024, *Federal Register*.
20 922 NOAA, 2023. List of Fisheries for 2024, *Federal Register*.
21 923 Northridge, S., Coram, A., Kingston, A., Crawford, R., 2017. Disentangling the causes of protected
22 924 species bycatch in gillnet fisheries. *Conserv. Biol. J. Soc. Conserv. Biol.* 31, 686–695.
23 925 <https://doi.org/10.1111/cobi.12741>
24 926 O'Keefe, C.E., Cadrin, S.X., Glemariee, G., Rouxel, Y., 2023. Efficacy of Time Area Fishing Restrictions
25 927 and Gear Switching as Solutions for Reducing Seabird Bycatch in Gillnet Fisheries. *Rev. Fish.
26 928 Sci. Aquac.* 31, 29–46. <https://doi.org/10.1080/23308249.2021.1988051>
27 929 Oldach, E., Killeen, H., Shukla, P., Brauer, E., Carter, N., Fields, J., Thomsen, A., Cooper, C., Mellinger,
28 930 E., Wang, K., Hendrickson, C., Neumann, A., Bøving, P.S., Fangue, N., 2022. Managed and
29 931 unmanaged whale mortality in the California Current Ecosystem. *Mar. Policy* 140, 105039.
30 932 <https://doi.org/10.1016/j.marpol.2022.105039>
31 933 Ortiz, M., Arocha, F., 2004. Alternative error distribution models for standardization of catch rates of
32 934 non-target species from a pelagic longline fishery: billfish species in the Venezuelan tuna
33 935 longline fishery. *Fish. Res., Models in Fisheries Research: GLMs, GAMS and GLMMs* 70, 275–
34 936 297. <https://doi.org/10.1016/j.fishres.2004.08.028>
35 937 Perkins, P., Barlow, J., Beeson, M., 1994. Report on Pinniped and Cetacean Mortality in California
36 938 Gillnet Fisheries: 1988–1990 (Administrative Report No. LJ 94-11). NOAA Southwest Fisheries
37 939 Science Center, La Jolla, CA.
38 940 Perkins, P., Barlow, J., Beeson, M., 1992a. Report on Pinniped and Cetacean Mortality in California
39 941 Gillnet Fisheries: 1990–1991 (Administrative Report No. LJ 92-14). NOAA Southwest Fisheries
40 942 Science Center, La Jolla, CA.
41 943 Perkins, P., Barlow, J., Beeson, M., 1992b. Pinniped and cetacean mortality in California gillnet fisheries:
42 944 +1991 (No. SC/44/SM14).
43 945 Punt, A.E., Siple, M.C., Francis, T.B., Hammond, P.S., Heinemann, D., Long, K.J., Moore, J., Sepúlveda,
44 946 M., Reeves, R.R., Sigurðsson, G.M., Vikingsson, G., Wade, P.R., Williams, R., Zerbini, A.N.,
45
46
47
48
49
50
51
52
53
54
55
56
57
58
59
60
61
62
63
64
65

- 1
2
3
4
5
6
7
8
9
10
11 947 2021. Can we manage marine mammal bycatch effectively in low data environments? *J. Appl.*
12 *Ecol.* 58, 596–607. <https://doi.org/10.1111/1365-2664.13816>
13
14 948 Read, A.J., Drinker, P., Northridge, S., 2006. Bycatch of Marine Mammals in U.S. and Global Fisheries.
15 *Conserv. Biol.* 20, 163–169. <https://doi.org/10.1111/j.1523-1739.2006.00338.x>
16
17 951 Rochet, M. J., Trenkel, V.M., 2005. Factors for the variability of discards: assumptions and field
18 evidence. *Can. J. Fish. Aquat. Sci.* 62, 224–235. <https://doi.org/10.1139/f04-185>
19
20 952 Samhouri, J.F., Feist, B.E., Fisher, M.C., Liu, O., Woodman, S.M., Abrahms, B., Forney, K.A., Hazen,
21 E.L., Lawson, D., Redfern, J., Saez, L.E., 2021. Marine heatwave challenges solutions to human–
22 wildlife conflict. *Proc. R. Soc. B Biol. Sci.* 288, 20211607.
23 <https://doi.org/10.1098/rspb.2021.1607>
24
25 957 SCCOOS, 2023. California HAB Bulletin: May July 2023 [WWW Document]. URL
26 https://secoos.org/california_hab_bulletin/may_2023/ (accessed 5.31.24).
27
28 959 Senko, J., White, E.R., Heppell, S.S., Gerber, L.R., 2014. Comparing bycatch mitigation strategies for
29 vulnerable marine megafauna. *Anim. Conserv.* 17, 5–18. <https://doi.org/10.1111/aev.12051>
30
31 961 Smith, J., Cram, J.A., Berndt, M.P., Hoard, V., Shultz, D., Deming, A.C., 2023. Quantifying the linkages
32 between California sea lion (*Zalophus californianus*) strandings and particulate domoic acid
33 concentrations at piers across Southern California. *Front. Mar. Sci.* 10, 1278293.
34 <https://doi.org/10.3389/fmars.2023.1278293>
35
36 965 Smith, J.A., Tommasi, D., Sweeney, J., Brodie, S., Welch, H., Hazen, E.L., Muhling, B., Stohs, S.M.,
37 Jacox, M.G., 2020. Lost opportunity: Quantifying the dynamic economic impact of time-area
38 fishery closures. *J. Appl. Ecol.* 57, 502–513. <https://doi.org/10.1111/1365-2664.13565>
39
40 968 Soykan, C., Moore, J., Zydelis, R., Crowder, L., Safina, C., Lewison, R., 2008. Why study bycatch? An
41 introduction to the Theme Section on fisheries bycatch. *Endanger. Species Res.* 5, 91–102.
42 <https://doi.org/10.3354/esr00175>
43
44 971 Stoeck, B.C., Ward, E.J., Thorson, J.T., Jannot, J.E., Semmens, B.X., 2019. The utility of spatial model-
45 based estimators of unobserved bycatch. *ICES J. Mar. Sci.* 76, 255–267.
46 <https://doi.org/10.1093/icesjms/fsy153>
47
48 974 Suuronen, P., Gilman, E., 2020. Monitoring and managing fisheries discards: New technologies and
49 approaches. *Mar. Policy* 116, 103554. <https://doi.org/10.1016/j.marpol.2019.103554>
50
51 976 Wright, M.N., Ziegler, A., 2017. ranger: A Fast Implementation of Random Forests for High Dimensional
52 Data in C++ and R. *J. Stat. Softw.* 77, 1–17. <https://doi.org/10.18637/jss.v077.i01>

- 1
2
3
4
5
6
7
8
9
10
1178 [Avila, I.C., Kaschner, K., Dommann, C.F., 2018. Current global risks to marine mammals: Taking stock of the threats. Biol. Conserv. 221, 44–58.](#)
1179 <https://doi.org/10.1016/j.biocon.2018.02.021>
- 1180 [Barlow, J., Baird, R.W., Heyning, J.E., Wynne, K., Manville, A.M., Lowry, L.F., Hanan, D., Sease, J., Burkanov, V., 1994. A Review of Cetacean and Pinniped Mortality in Coastal Fisheries Along the West Coast of the USA and Canada and the East Coast of the Russian Federation. Rep. Int. Whal. Comm. 15, 405–426.](#)
- 1181 [Bettoli, P.W., Scholten, G.D., 2006. Bycatch rates and initial mortality of paddlefish in a commercial gillnet fishery. Fish. Res. 77, 343–347.](#)
1182 <https://doi.org/10.1016/j.fishres.2005.11.008>
- 1183 [Birch, C., Blacow-Draeger, A., Geoff Shester, PH.D., Webb, S., Purcell, E., 2023. The Net Consequence: Impacts of Set Gillnets on California Ocean Biodiversity. Zenodo.](#)
1184 <https://doi.org/10.5281/ZENODO.7971874>
- 1185 [Birch, C., Shester, G., 2023. Underreporting of Marine Mammal Takes in the California Set Gillnet Fishery Underscores the Need for Observers. Oceana.](#)
- 1186 [Bjørge, A., Skern-Mauritzen, M., Rossman, M.C., 2013. Estimated bycatch of harbour porpoise \(*Phocoena phocoena*\) in two coastal gillnet fisheries in Norway, 2006–2008. Mitigation and implications for conservation. Biol. Conserv. 161, 164–173.](#)
1187 <https://doi.org/10.1016/j.biocon.2013.03.009>
- 1188 CA Secretary of State, 1990. Proposition 132. Marine Resources. Initiative Constitutional Amendment.
- 1189 Cameron, G.A., Forney, K.A., 2000. Preliminary Estimates of Cetacean Mortality in California/Oregon Gillnet Fisheries for 1999 (International Whaling Commission Working Paper No. SC/52/O24).
- 1190 Cameron, G.A., Forney, K.A., 1999. Preliminary Estimates of Cetacean Mortality in the California Gillnet Fisheries for 1997 and 1998 (No. SC/51/04). Southwest Fisheries Science Center.
- 1191 Capitolo, P.J., McChesney, G.J., Bechaver, C.A., Rhoades, S.J., Shore, A., Carter, H.R., Parker, M.W., Eigner, L.E., 2012. Breeding population trends of Brandt's and double-crested cormorants, Point Sur to Point Mugu, California, 1979-2011.
- 1192 Carretta, J.V., 2023. Sources of human-related injury and mortality for U.S. Pacific West Coast marine mammal stock assessments, 2017-2021. <https://doi.org/10.25923/QWF2-9B97>
- 1193 Carretta, J.V., 2002. Preliminary estimates of cetacean mortality in California gillnet fisheries for 2001 (No. SC/54/SM12). International Whaling Commission, Scientific Committee, Shimonoseki, Japan.
- 1194 Carretta, J.V., 2001. Preliminary estimates of cetacean mortality in California gillnet fisheries for 2000 (SC/53/SM9). International Whaling Commission, Scientific Committee, London, UK., London, UK.
- 1195 Carretta, J.V., Chivers, S.J., 2004. Preliminary estimates of marine mammal mortality and biological sampling of cetaceans in California gillnet fisheries for 2003 (No. SC/56/SM1).
- 1196 Carretta, J.V., Chivers, S.J., 2003. Preliminary estimates of marine mammal mortality and biological sampling of cetaceans in California gillnet fisheries for 2002 (No. SC/55/SM3).
- 1197 Carretta, J.V., Enriquez, L., 2012a. Marine mammal and seabird bycatch in California gillnet fisheries in 2010 (Administrative Report No. LJ-12-01). NOAA Southwest Fisheries Science Center, La Jolla, CA.
- 1198 Carretta, J.V., Enriquez, L., 2012b. Marine mammal and seabird bycatch in California gillnet fisheries in 2011 (NOAA Technical Memorandum NMFS No. NOAA-TM-NMFS-SWFSC-500). NOAA Southwest Fisheries Science Center, La Jolla, CA.
- 1199 Carretta, J.V., Enriquez, L., 2009. Marine mammal and seabird bycatch in observed California commercial fisheries in 2007 (Administrative Report No. LJ-09-01). NOAA Southwest Fisheries Science Center, La Jolla, CA.

1
2
3
4
5
6
7
8
9
10

- 1029 [Carretta, J.V., Enriquez, L., Villafana, C., 2014. Marine mammal, sea turtle, and seabird bycatch](#)
1030 [in California gillnet fisheries in 2012 \(NOAA Technical Memorandum NMFS No. NOAA-](#)
1031 [TM-NMFS-SWFSC-526\). NOAA Southwest Fisheries Science Center, La Jolla, CA.](#)
- 1032 [Carretta, J.V., Oleson, E.M., Forney, K.A., Muto, M.M., Weller, D.W., Lang, A.R., Baker, J.,](#)
1033 [Hanson, B., Orr, A.J., Barlow, J., Moore, J.E., Brownell Jr, R.L., 2022. U.S. Pacific](#)
1034 [marine mammal stock assessments: 2021 \(NOAA Technical Memorandum No. NOAA-](#)
1035 [TM-NMFS-SWFSC-663\). NOAA Southwest Fisheries Science Center, La Jolla, CA.](#)
- 1036 [Carter, H.R., 2001. Population Trends of the Common Murre \(*Uria aalge californica*\), in: *Biology*](#)
1037 [and Conservation of the Common Murre in California, Oregon, Washington, and British](#)
1038 [Columbia Volume 1: Natural History and Population Trends.](#)
- 1039 [CDFW, 2023. Evaluating Bycatch in the California Halibut Set Gill Net Fishery, California](#)
1040 [Department of Fish and Wildlife, Sacramento, CA.](#)
- 1041 [CDFW, 2021. California Wildlife Habitat Relationship System.](#)
- 1042 [CDFW, 2019. California Pacific Herring Fishery Management Plan.](#)
- 1043 [CDFW, 2014. Harbor Seals \[ds106\] GIS Dataset.](#)
- 1044 [CDFW, 2010. Seabird Colonies: California, 2010.](#)
- 1045 [CDFW, 2002. Bathymetry Project: 25m bathymetry dataset.](#)
- 1046 [Cochran, W.G., 1977. Sampling Techniques. John Wiley and Sons.](#)
- 1047 [Cohen, J., 1968. Weighted kappa: Nominal scale agreement provision for scaled disagreement](#)
1048 [or partial credit. *Psychol. Bull.* 70, 213–220. <https://doi.org/10.1037/h0026256>](#)
- 1049 [Condylios, S., 2023. priceR: Economics and Pricing Tools.](#)
- 1050 [Crowder, L.B., Murawski, S.A., 1998. Fisheries Bycatch: Implications for Management,](#)
1051 [*Fisheries* 23, 8–17. \[https://doi.org/10.1577/1548-8446\\(1998\\)023<0008:FBIFM>2.0.CO;2\]\(https://doi.org/10.1577/1548-8446\(1998\)023<0008:FBIFM>2.0.CO;2\)](#)
- 1052 [Cutler, D.R., Edwards, T.C., Beard, K.H., Cutler, A., Hess, K.T., Gibson, J., Lawler, J.J., 2007.](#)
- 1053 [Random Forests for Classification in Ecology. *Ecology* 88, 2783–2792.](#)
- 1054 [Diamond, S.L., Hanan, D.A., 1986. An Estimate of Harbor Porpoise Mortality in California Set](#)
1055 [Net Fisheries, April 1, 1983 through March 31, 1984 \(Administrative Report No. SWR-](#)
1056 [86-16\). NOAA Southwest Fisheries Science Center, La Jolla, CA.](#)
- 1057 [Fleiss, J.L., Levin, B., Paik, M.C., 2013. Statistical Methods for Rates and Proportions. John](#)
1058 [Wiley & Sons.](#)
- 1059 [Forney, K.A., Benson, S.R., Cameron, G.A., 2001. Central California gillnet effort and bycatch of](#)
1060 [sensitive species, 1990–1998, in: Melvin, E., Parrish, J.K. \(Eds.\), *Seabird Bycatch:*](#)
1061 [Trends, Roadblocks, and Solutions. Alaska Sea Grant, University of Alaska Fairbanks,](#)
1062 [pp. 141–160. <https://doi.org/10.4027/sbtrs.2001.08>](#)
- 1063 [Forney, K.A., Carretta, J.V., Benson, S.R., 2014. Preliminary estimates of harbor porpoise](#)
1064 [abundance in Pacific Coast waters of California, Oregon and Washington, 2007–2012.](#)
- 1065 [Forney, K.A., Moore, J.E., Barlow, J., Carretta, J.V., Benson, S.R., 2021. A multidecadal](#)
1066 [Bayesian trend analysis of harbor porpoise \(*Phocoena phocoena*\) populations off](#)
1067 [California relative to past fishery bycatch. *Mar. Mammal Sci.* 37, 546–560.](#)
1068 [https://doi.org/10.1111/mms.12764](#)
- 1069 [Free, C.M., Bellquist, L.F., Forney, K.A., Humberstone, J., Kauer, K., Lee, Q., Liu, O.R.,](#)
1070 [Samhouri, J.F., Wilson, J.R., Bradley, D., 2023. Static management presents a simple](#)
1071 [solution to a dynamic fishery and conservation challenge. *Biol. Conserv.* 285, 110249.](#)
1072 [https://doi.org/10.1016/j.biocon.2023.110249](#)
- 1073 [Freeman, E.A., Moisen, G.G., 2008. A comparison of the performance of threshold criteria for](#)
1074 [binary classification in terms of predicted prevalence and kappa. *Ecol. Model.* 217, 48–](#)
1075 [58. <https://doi.org/10.1016/j.ecolmodel.2008.05.015>](#)
- 1076 [Geijer, C.K.A., Read, A.J., 2013. Mitigation of marine mammal bycatch in U.S. fisheries since](#)
1077 [1994. *Biol. Conserv.* 159, 54–60. <https://doi.org/10.1016/j.biocon.2012.11.009>](#)
- 1078 [Hallegraeff, G.M., Anderson, D.M., Belin, C., Bottein, M.-Y.D., Bresnan, E., Chinain, M.,](#)
1079 [Enevoldsen, H., Iwataki, M., Karlson, B., McKenzie, C.H., Sunesen, I., Pitcher, G.C.,](#)

54

55

56

57

58

59

60

61

62

63

64

65

- 1
2
3
4
5
6
7
8
9
10
11 80 Provoost, P., Richardson, A., Schweibold, L., Tester, P.A., Trainer, V.L., Yñiguez, A.T.,
11 81 Zingone, A., 2021. Perceived global increase in algal blooms is attributable to intensified
11 82 monitoring and emerging bloom impacts. *Commun. Earth Environ.* 2, 117.
11 83 <https://doi.org/10.1038/s43247-021-00178-8>
11 84 Hanan, D.A., Diamond, S.L., 1989. Estimates of Sea Lion, Harbor Seal, and Harbor Porpoise
11 85 Mortalities in California Set Net Fisheries for the 1986-87 Fishing Year.
11 86 Hanan, D.A., Diamond, S.L., Scholl, J.P., 1988. Estimates of Sea Lion and Harbor Seal
11 87 Mortalities in California Set Net Fisheries for 1983, 1984, and 1985.
11 88 Hanan, D.A., Diamond, S.L., Scholl, J.P., 1987. An Estimate of Harbor Porpoise Mortalities in
11 89 California Set Net Fisheries, April 1, 1985 through March 31, 1986 (Administrative
11 90 Report No. SWR 87-5). NOAA Southwest Fisheries Science Center, La Jolla, CA.
11 91 Hanan, D.A., Diamond, S.L., Scholl, J.P., 1986. An Estimate of Harbor Porpoise Mortality in
11 92 California Set Net Fisheries April 1, 1984 through March 31, 1985 (Administrative Report
11 93 No. SWR 86-16). NOAA Southwest Fisheries Science Center, La Jolla, CA.
11 94 Hazen, E.L., Scales, K.L., Maxwell, S.M., Briscoe, D.K., Welch, H., Bograd, S.J., Bailey, H.,
11 95 Benson, S.R., Eguchi, T., Dewar, H., Kohin, S., Costa, D.P., Crowder, L.B., Lewison,
11 96 R.L., 2018. A dynamic ocean management tool to reduce bycatch and support
11 97 sustainable fisheries. *Sci. Adv.* 4, eaar3001. <https://doi.org/10.1126/sciadv.aar3001>
11 98 Huang, B., Liu, C., Banzon, V., Freeman, E., Graham, G., Hankins, B., Smith, T., Zhang, H.-M.,
11 99 2021. Improvements of the Daily Optimum Interpolation Sea Surface Temperature
11 100 (DOISST) Version 2.1. *J. Clim.* 34, 2923–2939. <https://doi.org/10.1175/JCLI-D-20-0166.1>
11 101 Hvitfeldt, E., 2023. themis: Extra Recipes Steps for Dealing with Unbalanced Data. R package.
11 102 ICES, 2007. Report of the Workshop on Discard Raising Procedures (ICES CM No.
11 103 2007ACFM:06). ICES, San Sebastian, Spain.
11 104 Julian, F., 1994. Pinniped and cetacean mortality in California gillnet fisheries: preliminary
11 105 estimates for 1993 (International Whaling Commission No. SC/46/0). NOAA Southwest
11 106 Fisheries Science Center, La Jolla, CA.
11 107 Julian, F., 1993. Pinniped and cetacean mortality in California gillnet fisheries: preliminary
11 108 estimates for 1992.
11 109 Julian, F., Beeson, M., 1998. Estimates of marine mammal, turtle, and seabird mortality for two
11 110 California gillnet fisheries: 1990-1995. *Fish. Bull.* 96, 271–284.
11 111 Kirby, D.S., Ward, P., 2014. Standards for the effective management of fisheries bycatch. *Mar.
11 112 Policy* 44, 419–426. <https://doi.org/10.1016/j.marpol.2013.10.008>
11 113 Konno, E.S., 1990. Effort Estimates of Gill Net Fisheries in California that Incidentally Catch
11 114 Marine Mammals, for the 1987-88 Fishing Year. NOAA Southwest Fisheries Science
11 115 Center, Terminal Island, CA.
11 116 Kroetz, A.M., Mathers, A.N., Carlson, J.K., 2020. Evaluating protected species bycatch in the
11 117 U.S. Southeast Gillnet Fishery. *Fish. Res.* 228, 105573.
11 118 <https://doi.org/10.1016/j.fishres.2020.105573>
11 119 Laake, J.L., Lowry, M.S., DeLong, R.L., Melin, S.R., Carretta, J.V., 2018. Population growth and
11 120 status of California sea lions. *J. Wildl. Manag.* 82, 583–595.
11 121 <https://doi.org/10.1002/jwmg.21405>
11 122 Landis, J.R., Koch, G.G., 1977. The Measurement of Observer Agreement for Categorical Data.
11 123 *Biometrics* 33, 159–174. <https://doi.org/10.2307/2529310>
11 124 Lennert, C., Kruse, S., Beeson, M., Barlow, J., 1994. Estimates of incidental marine mammal
11 125 bycatch in California gillnet fisheries for July through December, 1990 (Report of the
11 126 International Whaling Commission No. SC/43/O 3).
11 127 Lewison, R.L., Crowder, L.B., Wallace, B.P., Moore, J.E., Cox, T., Zydelis, R., McDonald, S.,
11 128 DiMatteo, A., Dunn, D.C., Kot, C.Y., Bjorkland, R., Kelez, S., Soykan, C., Stewart, K.R.,
11 129 Sims, M., Boustany, A., Read, A.J., Halpin, P., Nichols, W.J., Safina, C., 2014. Global
11 130
54
55
56
57
58
59
60
61
62
63
64
65

1
2
3
4
5
6
7
8
9
10

- 1131 patterns of marine mammal, seabird, and sea turtle bycatch reveal taxa-specific and
1132 cumulative megafauna hotspots. *Proc. Natl. Acad. Sci.* 111, 5271–5276.
<https://doi.org/10.1073/pnas.1318960111>
- 1133 Liaw, A., Wiener, M., 2002. Classification and Regression by randomForest. *R News* 2, 18–22.
- 1134 Long, C.A., Ahrens, R.N.M., Jones, T.T., Siders, Z.A., 2024. A machine learning approach for
1135 protected species bycatch estimation. *Front. Mar. Sci.* 11.
<https://doi.org/10.3389/fmars.2024.1331292>
- 1136 Lopez, J., Griffiths, S., Wallace, B.P., Cáceres, V., Rodríguez, L.H., Abrego, M., Alfaro-
1137 Shiqueto, J., Andraka, S., Brito, M.J., Bustos, L.C., Cari, I., Carvajal, J.M., Clavijo, L.,
1138 Cocas, L., Paz, N. de, Herrera, M., Mangel, J.C., Pérez-Huaripata, M., Piedra, R.,
1139 Dávila, J.A.Q., Rendón, L., Ríquez-Baron, J.M., Santana, H., Suárez, J., Veelenturf, C.,
1140 Vega, R., Zárate, P., 2024. Vulnerability of the Critically Endangered leatherback turtle to
1141 fisheries bycatch in the eastern Pacific Ocean. I. A machine-learning species distribution
1142 model. *Endanger. Species Res.* 53, 271–293. <https://doi.org/10.3354/esr01288>
- 1143 Lowry, M., Condit, R., Hatfield, B., Allen, S.G., Berger, R., Morris, P.A., Le Boeuf, B.J., Reiter,
1144 .., 2014. Abundance, Distribution, and Population Growth of the Northern Elephant Seal
1145 (*Mirounga angustirostris*) in the United States from 1991 to 2010. *Aquat. Mamm.* 40, 20–
1146 31. <https://doi.org/10.1578/AM.40.1.2014.20>
- 1147 Lowry, M.S., 2021. Abundance and distribution of pinnipeds at the Channel Islands in southern
1148 California, central and northern California, and southern Oregon during summer 2016–
1149 2019 (NOAA Technical Memorandum NMFS No. NOAA-TM-NMFS-SWFSC-656).
1150 NOAA Southwest Fisheries Science Center, La Jolla, CA.
- 1151 Manel, S., Williams, H.C., Ormerod, S. j., 2001. Evaluating presence-absence models in
1152 ecology: the need to account for prevalence. *J. Appl. Ecol.* 38, 921–931.
<https://doi.org/10.1046/j.1365-2664.2001.00647.x>
- 1153 Martin, T.G., Wintle, B.A., Rhodes, J.R., Kuhnert, P.M., Field, S.A., Low-Choy, S.J., Tyre, A.J.,
1154 Possingham, H.P., 2005. Zero tolerance ecology: improving ecological inference by
1155 modelling the source of zero observations. *Ecol. Lett.* 8, 1235–1246.
<https://doi.org/10.1111/j.1461-0248.2005.00826.x>
- 1156 McCracken, M.L., 2004. Modeling a Very Rare Event to Estimate Sea Turtle Bycatch: Lessons
1157 Learned (NOAA Technical Memorandum No. NMFS-PIFSC-3). Pacific Islands Fisheries
1158 Science Center.
- 1159 McKibben, S.M., Peterson, W., Wood, A.M., Trainer, V.L., Hunter, M., White, A.E., 2017.
1160 Climatic regulation of the neurotoxin domoic acid. *Proc. Natl. Acad. Sci.* 114, 239–244.
<https://doi.org/10.1073/pnas.1606798114>
- 1161 More, A.S., Rana, D.P., 2017. Review of random forest classification techniques to resolve data
1162 imbalance, in: 2017 1st International Conference on Intelligent Systems and Information
1163 Management (ICISIM). Presented at the 2017 1st International Conference on Intelligent
1164 Systems and Information Management (ICISIM), IEEE, Aurangabad, pp. 72–78.
<https://doi.org/10.1109/ICISIM.2017.8122151>
- 1165 Nembrini, S., König, I.R., Wright, M.N., 2018. The revival of the Gini importance? *Bioinformatics*
1166 34, 3711–3718. <https://doi.org/10.1093/bioinformatics/bty373>
- 1167 NOAA, 2024. CA Halibut, White Seabass and Other Species Set Gillnet (>3.5 in mesh) - MMPA
1168 List of Fisheries [WWW Document]. NOAA. URL
1169 <https://www.fisheries.noaa.gov/national/marine-mammal-protection/ca-halibut-white->
1170 [seabass-and-other-species-set-gillnet-35-mesh](https://www.fisheries.noaa.gov/national/marine-mammal-protection/ca-halibut-white-seabass-and-other-species-set-gillnet-35-mesh) (accessed 6.4.24).
- 1171 NOAA, 2023. List of Fisheries for 2024, Federal Register.
- 1172 Northridge, S., Coram, A., Kingston, A., Crawford, R., 2017. Disentangling the causes of
1173 protected-species bycatch in gillnet fisheries. *Conserv. Biol. J. Soc. Conserv. Biol.* 31,
1174 686–695. <https://doi.org/10.1111/cobi.12741>
- 1175 O'Keefe, C.E., Cadrin, S.X., Glemarec, G., Rouxel, Y., 2023. Efficacy of Time-Area Fishing
- 1176
- 1177
- 1178
- 1179
- 1180
- 1181

54

55

56

57

58

59

60

61

62

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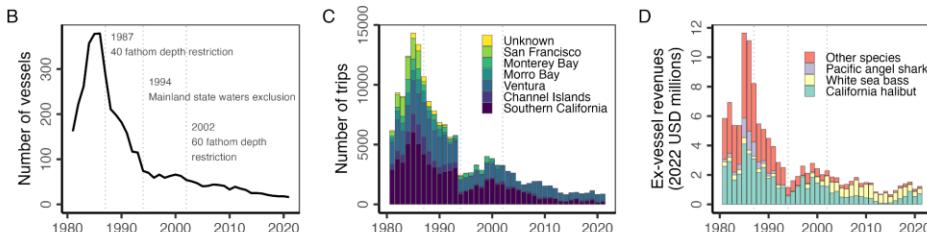
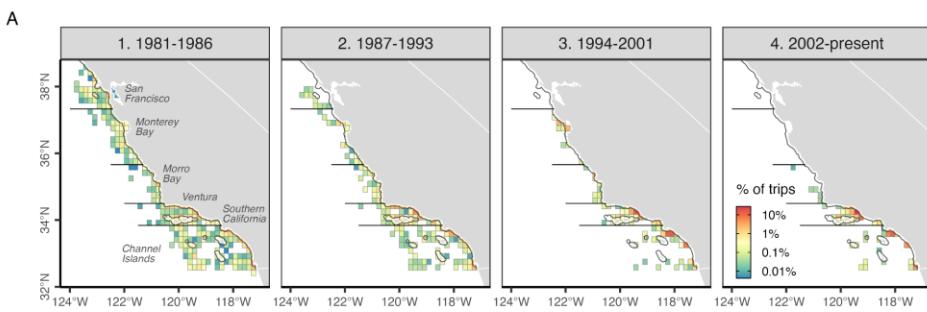
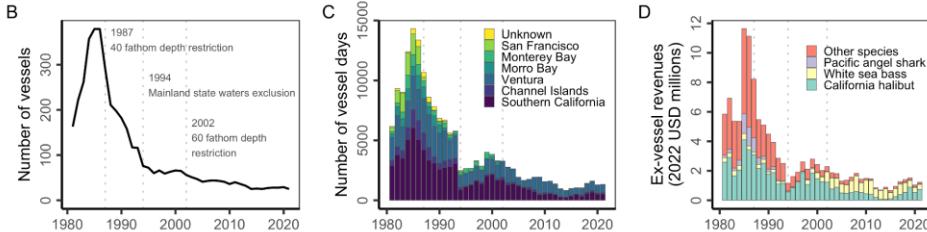
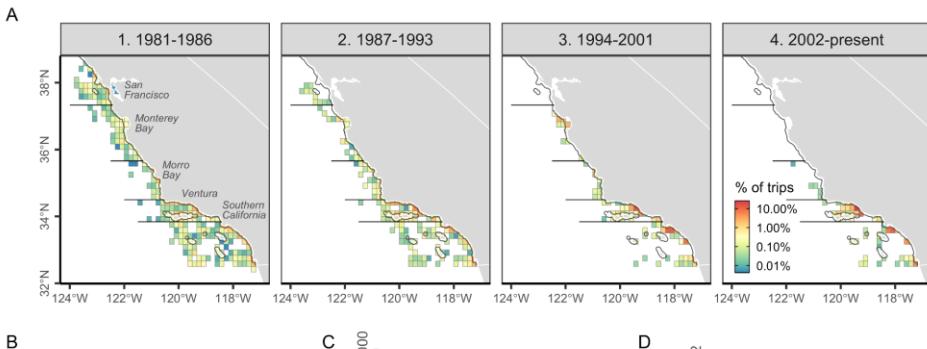
64

65

1
2
3
4
5
6
7
8
9
10
11 82 [Restrictions and Gear-Switching as Solutions for Reducing Seabird Bycatch in Gillnet Fisheries](#). *Rev. Fish. Sci. Aquac.* 31, 29–46.
11 83 <https://doi.org/10.1080/23308249.2021.1988051>
11 84
11 85 Oldach, E., Killeen, H., Shukla, P., Brauer, E., Carter, N., Fields, J., Thomsen, A., Cooper, C.,
11 86 Mellinger, L., Wang, K., Hendrickson, C., Neumann, A., Bøving, P.S., Fanque, N., 2022.
11 87 Managed and unmanaged whale mortality in the California Current Ecosystem. *Mar. Policy* 140, 105039. <https://doi.org/10.1016/j.marpol.2022.105039>
11 88
11 89 Ortiz, M., Arocha, F., 2004. Alternative error distribution models for standardization of catch
11 90 rates of non-target species from a pelagic longline fishery: billfish species in the
11 91 Venezuelan tuna longline fishery. *Fish. Res., Models in Fisheries Research: GLMs,*
11 92 *GAMS and GLMMs* 70, 275–297. <https://doi.org/10.1016/j.fishres.2004.08.028>
20 93 Perkins, P., Barlow, J., Beeson, M., 1994. Report on Pinniped and Cetacean Mortality in
21 94 California Gillnet Fisheries: 1988-1990 (Administrative Report No. LJ-94-11). NOAA
22 95 Southwest Fisheries Science Center, La Jolla, CA.
21 96 Perkins, P., Barlow, J., Beeson, M., 1992a. Report on Pinniped and Cetacean Mortality in
21 97 California Gillnet Fisheries: 1990-1991 (Administrative Report No. LJ-92-14). NOAA
22 98 Southwest Fisheries Science Center, La Jolla, CA.
21 99 Perkins, P., Barlow, J., Beeson, M., 1992b. Pinniped and cetacean mortality in California gillnet
22 00 fisheries: 1991 (No. SC/44/SM14).
21 01 Prasad, A.M., Iverson, L.R., Liaw, A., 2006. Newer Classification and Regression Tree
21 02 Techniques: Bagging and Random Forests for Ecological Prediction. *Ecosystems* 9,
21 03 181–199. <https://doi.org/10.1007/s10021-005-0054-1>
31 04 Punt, A.E., Siple, M.C., Francis, T.B., Hammond, P.S., Heinemann, D., Long, K.J., Moore, J.,
31 05 Sepúlveda, M., Reeves, R.R., Sigurðsson, G.M., Víkingsson, G., Wade, P.R., Williams,
31 06 R., Zerbini, A.N., 2021. Can we manage marine mammal bycatch effectively in low-data
31 07 environments? *J. Appl. Ecol.* 58, 596–607. <https://doi.org/10.1111/1365-2664.13816>
31 08 R Core Team (2024). R: A language and environment for statistical computing. R Foundation for
31 09 Statistical Computing, Vienna, Austria. URL <https://www.R-project.org/>
31 10 Read, A.J., Drinker, P., Northridge, S., 2006. Bycatch of Marine Mammals in U.S. and Global
31 11 Fisheries. *Conserv. Biol.* 20, 163–169. <https://doi.org/10.1111/j.1523-1739.2006.00338.x>
31 12 Rochet, M.-J., Trenkel, V.M., 2005. Factors for the variability of discards: assumptions and field
31 13 evidence. *Can. J. Fish. Aquat. Sci.* 62, 224–235. <https://doi.org/10.1139/f04-185>
31 14 Samhouri, J.F., Feist, B.E., Fisher, M.C., Liu, O., Woodman, S.M., Abrahms, B., Forney, K.A.,
31 15 Hazen, E.L., Lawson, D., Redfern, J., Saez, L.E., 2021. Marine heatwave challenges
41 16 solutions to human–wildlife conflict. *Proc. R. Soc. B Biol. Sci.* 288, 20211607.
41 17 <https://doi.org/10.1098/rspb.2021.1607>
41 18 SCCOOS, 2023. California HAB Bulletin: May-July 2023 [WWW Document]. URL
41 19 <https://sccoos.org/california-hab-bulletin/may-2023/> (accessed 5.31.24).
41 20 Senko, J., White, E.R., Heppell, S.S., Gerber, L.R., 2014. Comparing bycatch mitigation
41 21 strategies for vulnerable marine megafauna. *Anim. Conserv.* 17, 5–18.
41 22 <https://doi.org/10.1111/acv.12051>
41 23 Smith, J., Cram, J.A., Berndt, M.P., Hoard, V., Shultz, D., Deming, A.C., 2023. Quantifying the
41 24 linkages between California sea lion (*Zalophus californianus*) strandings and particulate
41 25 domoic acid concentrations at piers across Southern California. *Front. Mar. Sci.* 10,
41 26 1278293. <https://doi.org/10.3389/fmars.2023.1278293>
41 27 Soykan, C., Moore, J., Zydelis, R., Crowder, L., Safina, C., Lewison, R., 2008. Why study
51 28 bycatch? An introduction to the Theme Section on fisheries bycatch. *Endanger. Species*
51 29 Res. 5, 91–102. <https://doi.org/10.3354/esr00175>
51 30 Stock, B.C., Ward, E.J., Thorson, J.T., Jannot, J.E., Semmens, B.X., 2019. The utility of spatial
51 31 model-based estimators of unobserved bycatch. *ICES J. Mar. Sci.* 76, 255–267.
51 32 <https://doi.org/doi:10.1093/icesjms/fsy153>

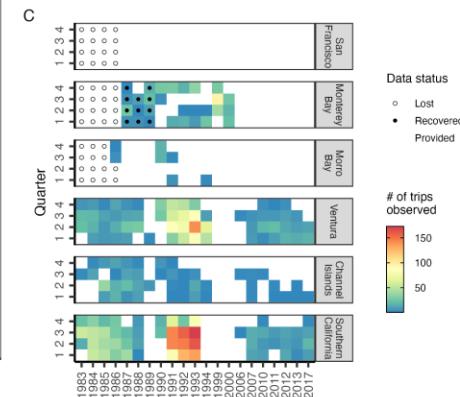
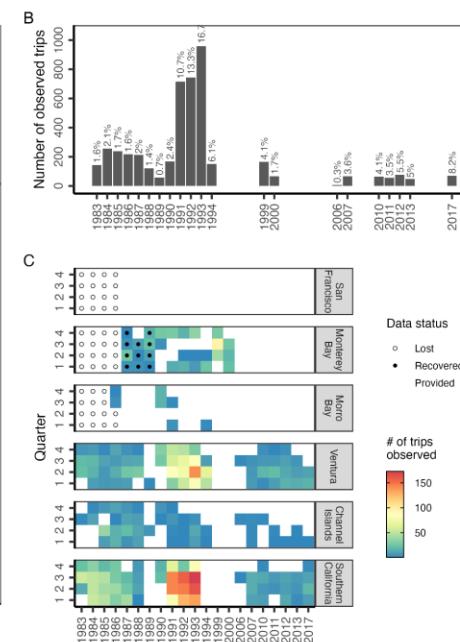
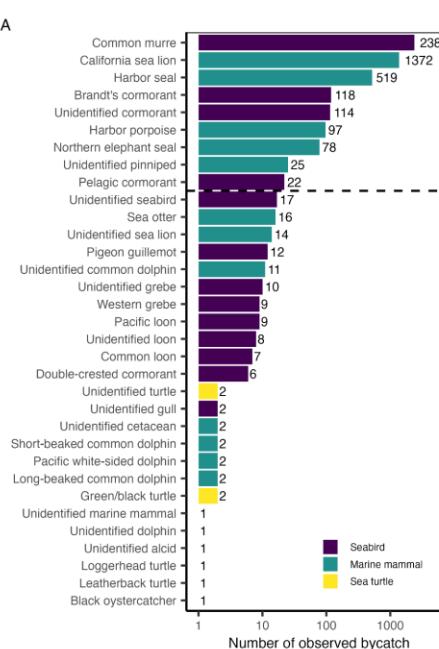
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11²³³ Suuronen, P., Gilman, E., 2020. Monitoring and managing fisheries discards: New technologies
12²³⁴ and approaches. Mar. Policy 116, 103554. <https://doi.org/10.1016/j.marpol.2019.103554>
13²³⁵ Wright, M.N., Ziegler, A., 2017. ranger: A Fast Implementation of Random Forests for High
14²³⁶ Dimensional Data in C++ and R. J. Stat. Softw. 77, 1–17.
15²³⁷ <https://doi.org/10.18637/jss.v077.i01>
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11241 **Fig. 1.** History of the California ≥3.5" set gillnet fishery. Panel A shows the spatial history of fishing
12242 effort during four regulatory periods. [Trips are reported by the 10 x 10 minute \(~18 x 18 km\) statistical](#)
13243 [blocks used for fisheries catch and effort reporting](#). The horizontal lines delineate geographical strata used
14244 in the ratio estimation analysis; strata are labeled in the first plot. The thin coastal line marks state waters
15245 (less than 3 nautical miles from the coast). Blocks visited by fewer than three vessels during each
16246 regulatory period are hidden to maintain confidentiality and comply with the “rule-of-three.” The other
17247 panels show time series of fisheries (B) participation, (C) effort, and (D) revenues. Vertical lines mark
20248 years in which major regulations, labeled in Panel B, were implemented; these define the regulatory
21249 periods used in Panel A. These regulations became operative on April 15, 1987; January 1, 1994; and
22250 April 26, 2002. See **Fig. S1** and the supplemental methods for details on estimating ex-vessel revenues
23251 from the fishery.
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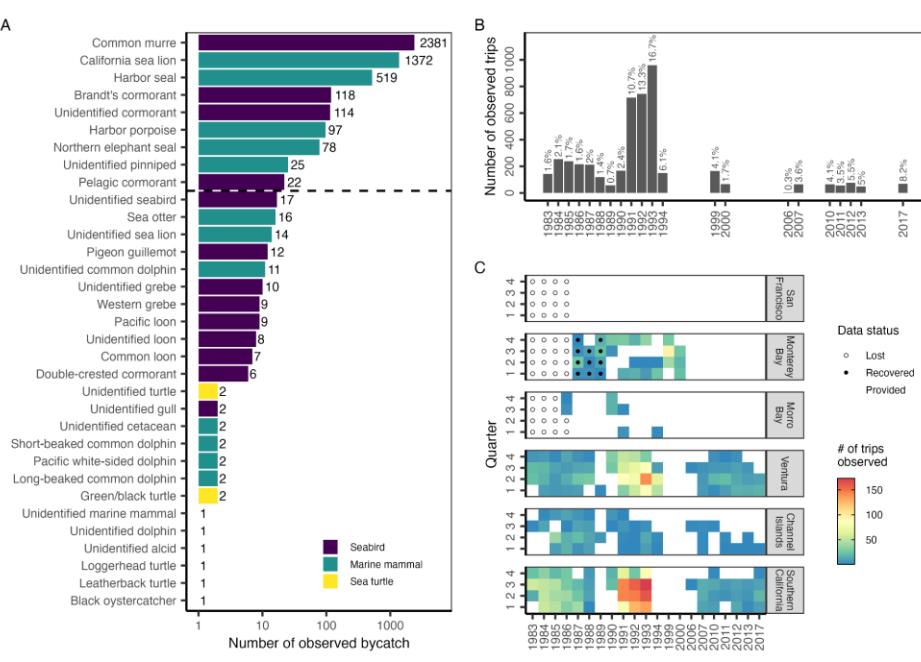


Fig. 2. History of observer coverage in the California $\geq 3.5"$ set gillnet fishery. Panel A shows the bycatch of marine mammals, seabirds, and sea turtles recorded by observers from 1983-2017. We focus on species with ≥ 50 observations, which are delineated by the horizontal dashed line. Note log-scale on x-axis. Panel B shows the number of observed trips (vessel-days) over time. The dark labels show the estimated percent of trips that were observed. Panel C shows the number of observed trips across the spatial (region) and temporal (quarters) strata considered in the ratio estimation analysis. See Fig. S8 for a map of the spatial strata. Quarters are defined as: 1 = JFM (winter), 2 = AMJ (spring), 3 = JAS (summer), and 4 = OND (fall).

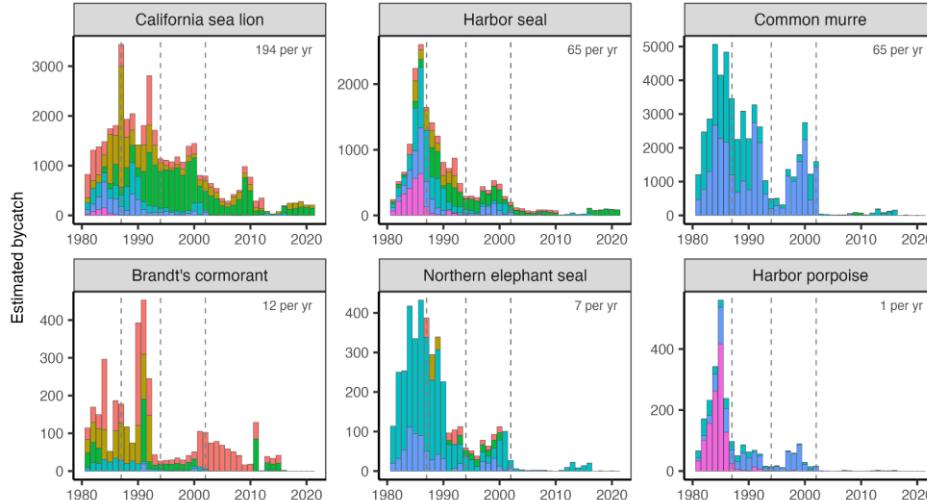
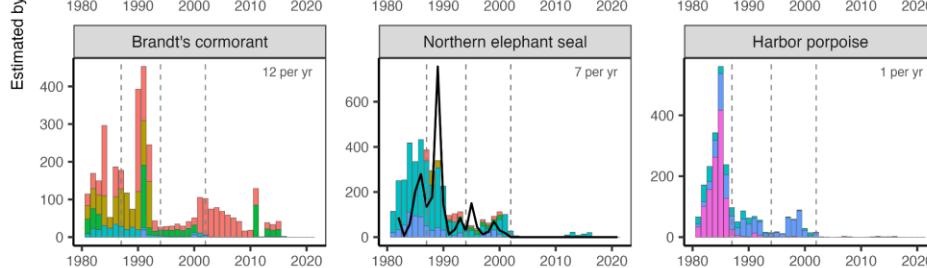
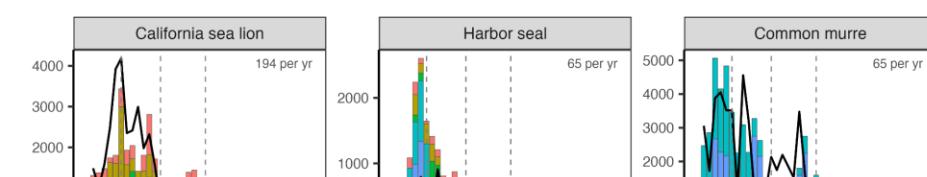
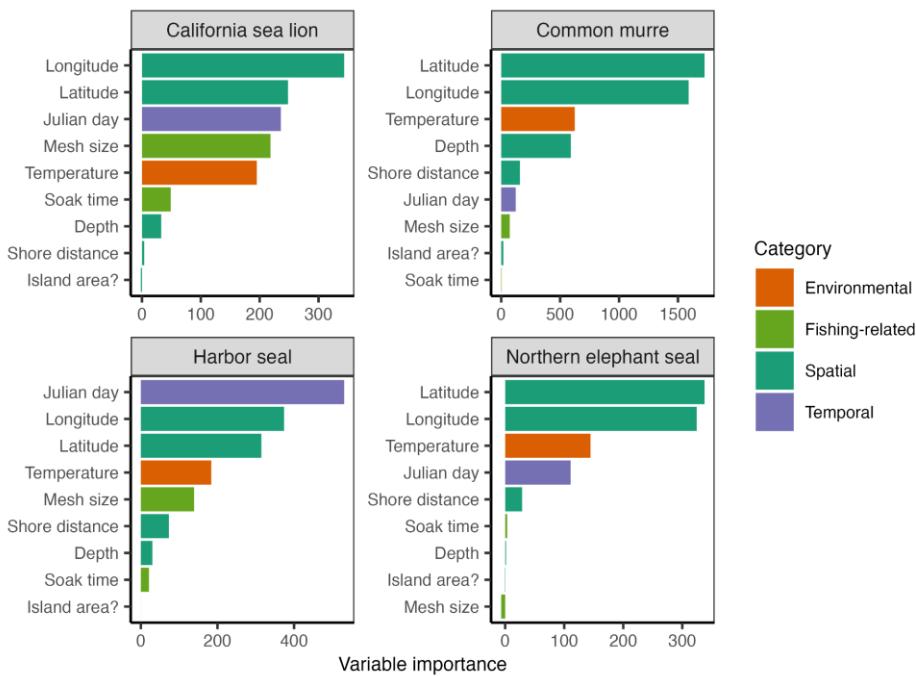


Fig. 3. Estimated bycatch in the California $\geq 3.5"$ set gillnet fishery from 1981-2021 predicted by using the ratio estimation (bars) and random forest (line) modeling approaches. Average estimated annual

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11 266 bycatch rates for the last 10 years (2012-2021) from the ratio estimator are marked in the top-right corner.
12 267 Species are listed in order of decreasing recent bycatch rates.



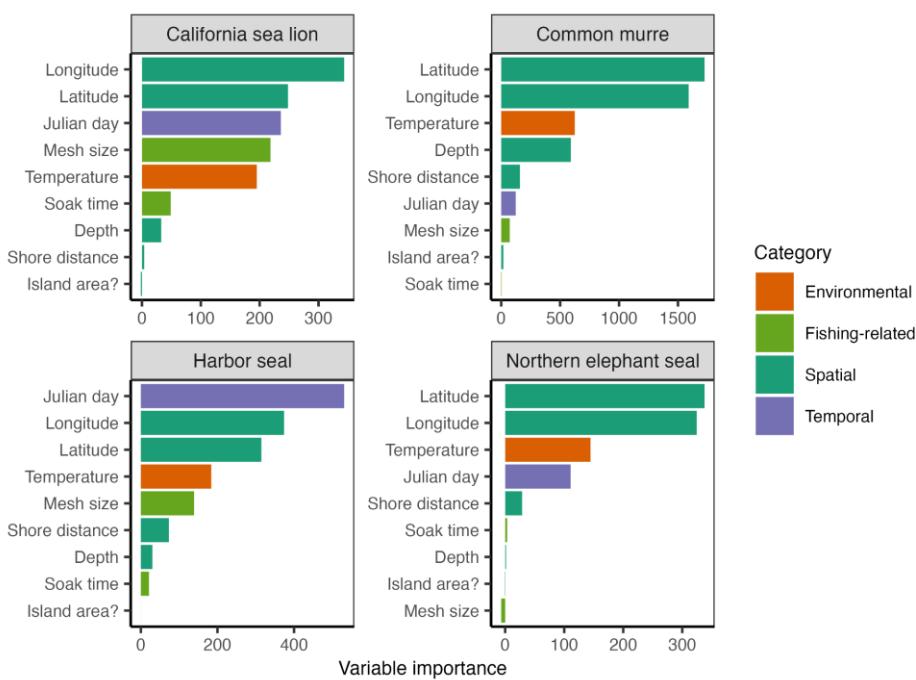
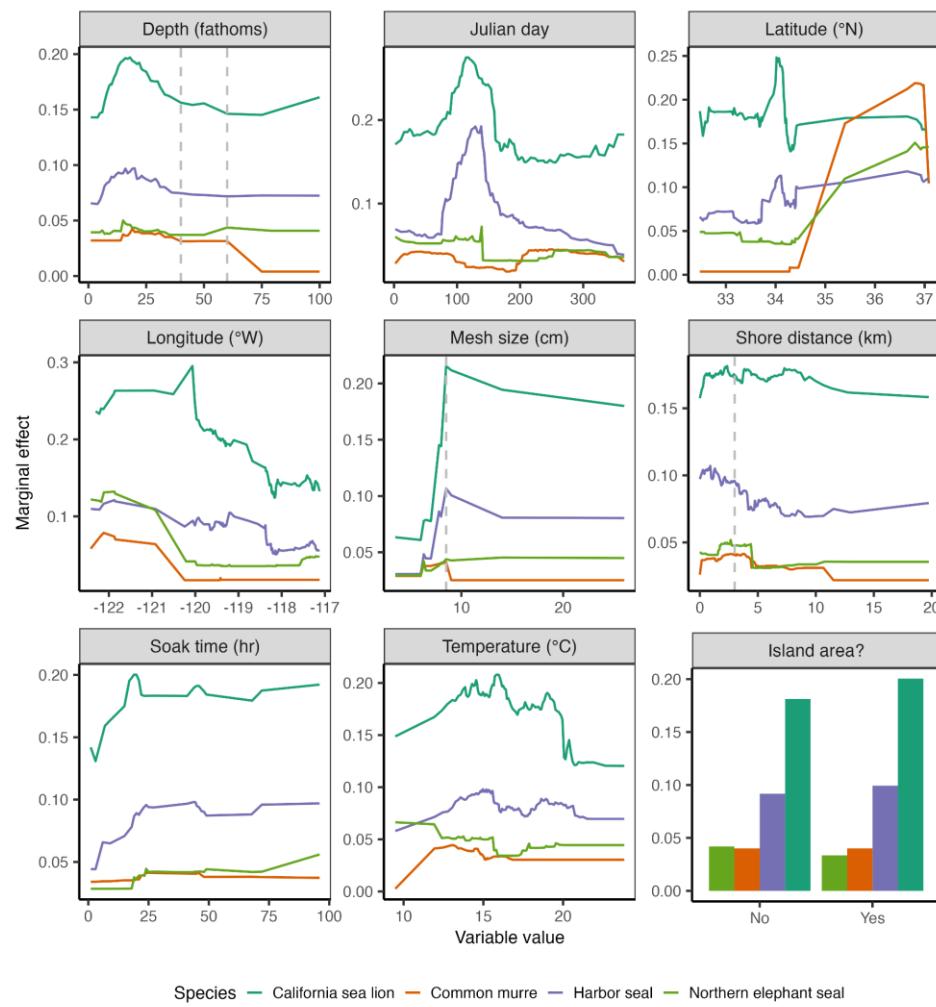


Fig. 4. Importance of the evaluated explanatory variables in the final random forest model for the study species with acceptable model performance. With the bias towards continuous variable corrected, variable importance is measured as the total decrease in node impurities from splitting on the variable averaged over all trees. Explanatory variables are colored based on the category of the variable.



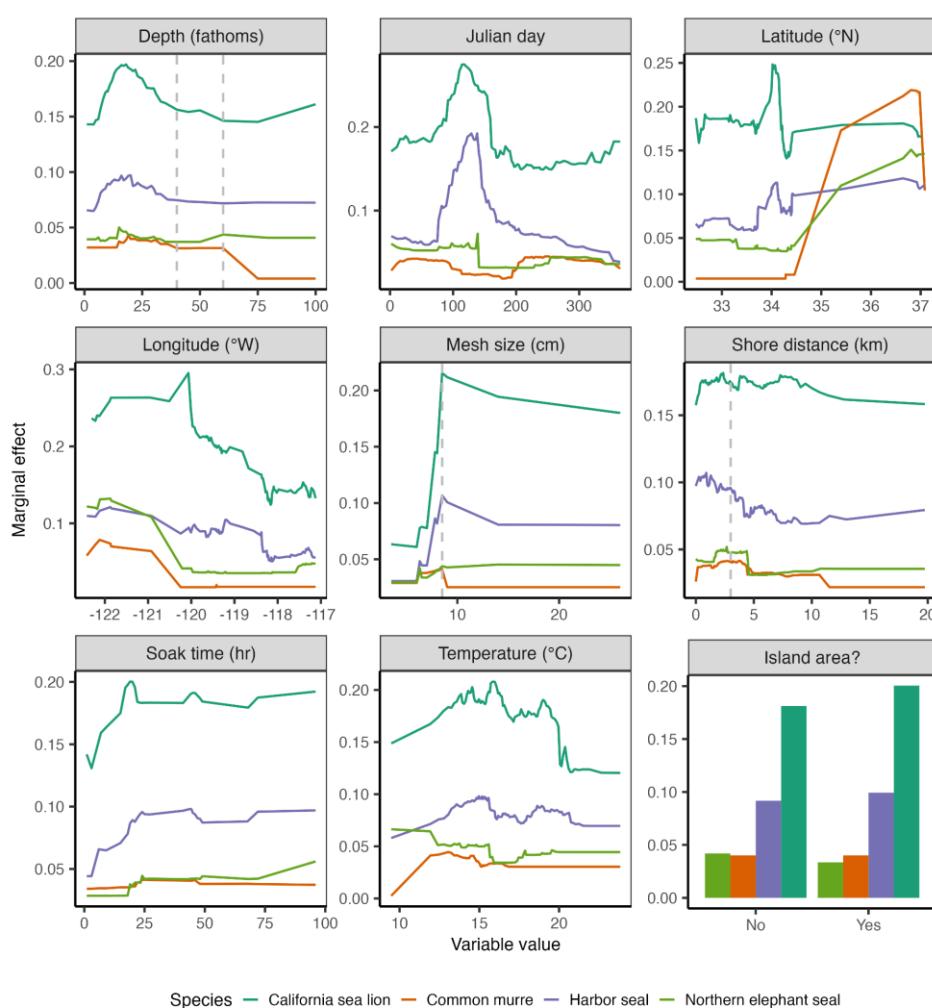
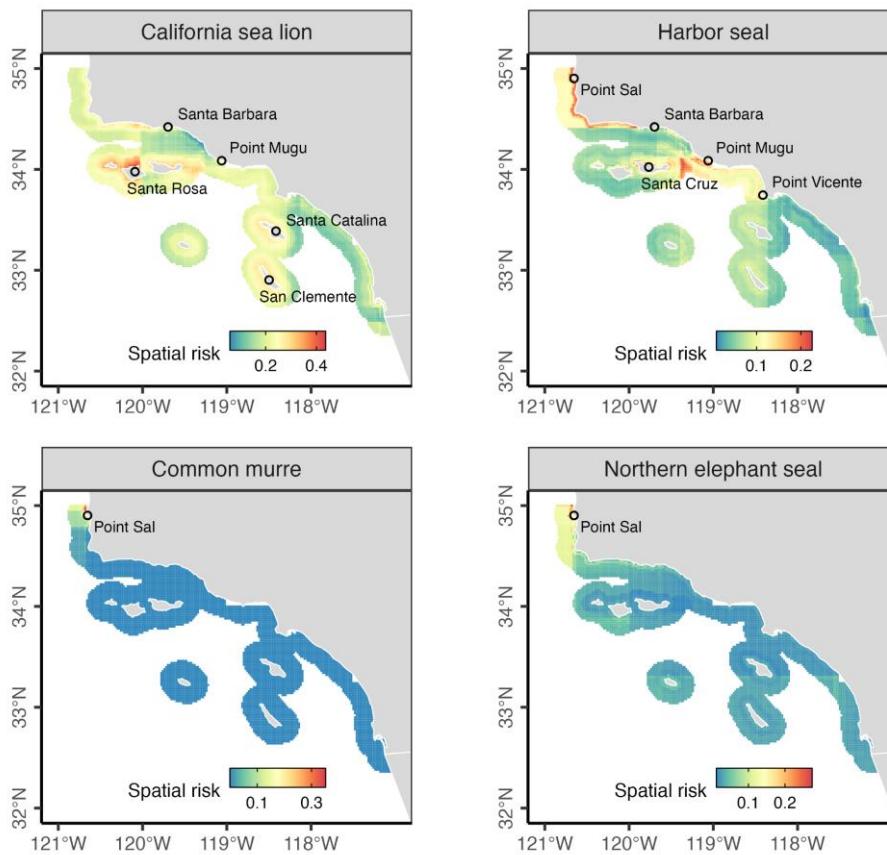


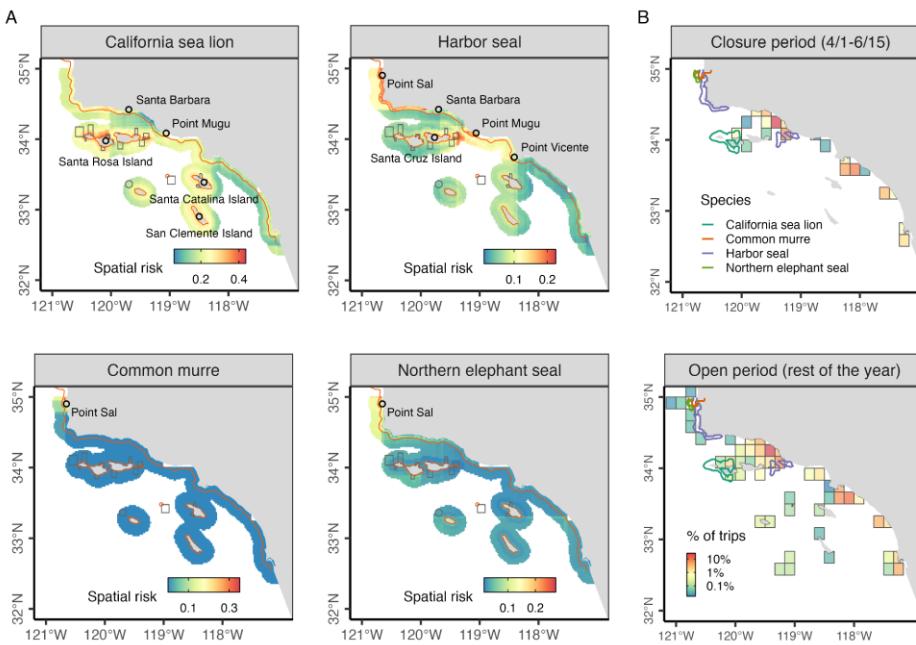
Fig. 5. Marginal effect of the evaluated explanatory variables on bycatch risk as estimated by the best fitting random forest model for the four study species with acceptable model performance. The marginal effect of each variable represents the effect of the variable when the other variables are held at their mean values. The dashed lines indicate 40 and 60 in Depth (fathoms), 8.5 in mesh size (cm), and 3 in shore distance (km).

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11 282 Fig. 6. Average



31 283 Fig. 6. The (A) spatial bycatch risk relative to current management areas and (B) spatial bycatch hotspots

32 relative to recent fishing effort. Panel A shows the average spatial bycatch risk as estimated by the best
33 fitting random forest model for the four study species with acceptable model performance. The spatial
34 bycatch risk represents the probability of bycatch at a given location under recent (2010–2021) average
35 conditions. Key landmarks for delineating bycatch hotspots are labeled in each panel. The thin orange
36 coastal line marks the nearshore areas from which gillnet fishing is excluded: within 3 nautical miles of
37 the mainland and within 1 nautical mile or shallower than 70 fathoms (whichever is closer to shore) from
38 the Channel Islands. The gray polygons indicate the locations of California Marine Protected Areas,
39 where all set gillnet fisheries are excluded. Spatial bycatch risk is shown only for southern California, as
40 this is the only area where the fishery can operate under current regulations. Panel B shows hotspots of
41 bycatch risk relative to recent fishing effort (2002–2022; see Fig 1A) during the proposed closed and open
42 periods.

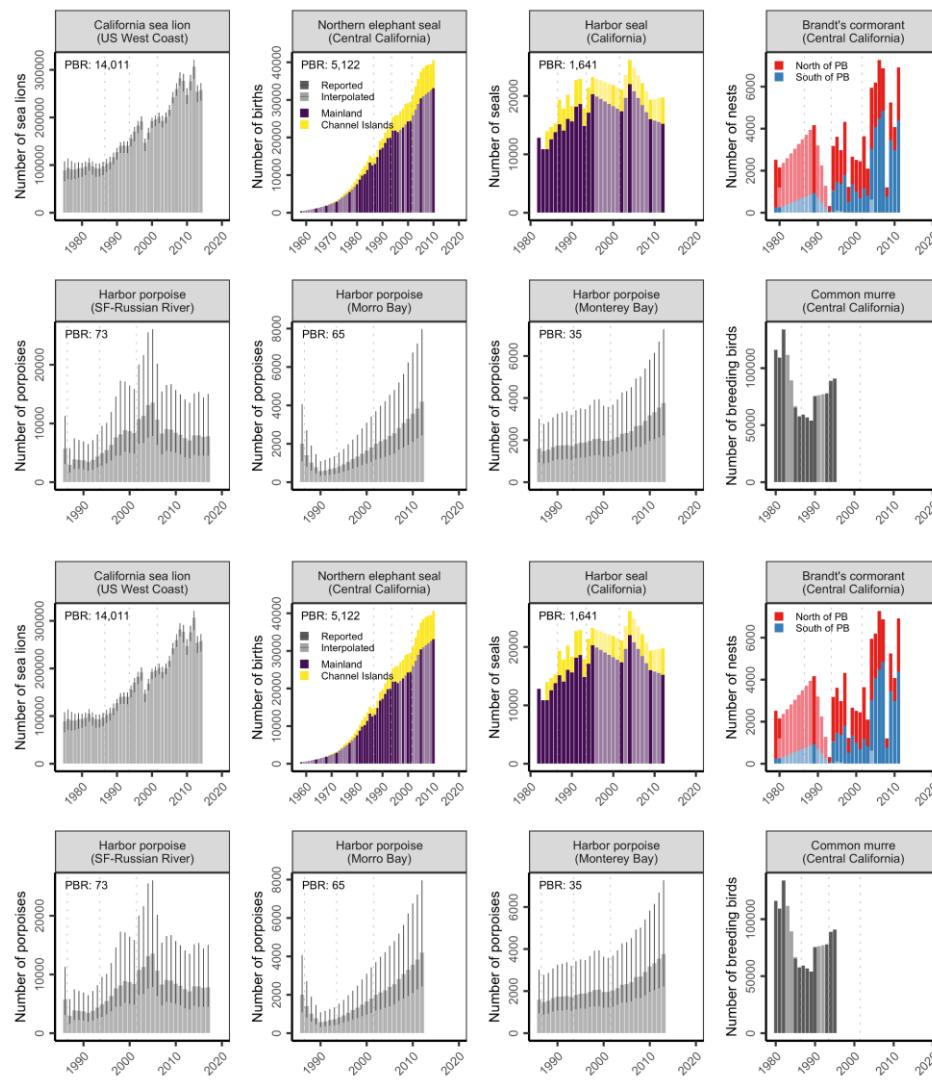


Fig. 7. Estimated abundance of populations of the six study species. The potential biological removal (PBR) indicates the maximum number of animals, not including natural mortalities, that may be removed from a marine mammal stock while allowing the stock to reach or maintain its optimum sustainable population. PBRs are only calculated for marine mammals. Error bars indicate 95% confidence or credible intervals. Years without reported values were filled using linear interpolation. Vertical lines mark

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11 303 years in which major bycatch regulations were implemented. Population estimates are from the following
12 304 sources: California sea lion ([Laake et al., 2018](#))[\(Laake et al., 2018\)](#), northern elephant seal ([Carretta et al.,](#)
13 305 [2022](#))[\(Carretta et al., 2022\)](#), harbor seal ([Carretta et al., 2022](#))[\(Carretta et al., 2022\)](#), harbor porpoise
14 306 ([Forney et al., 2021](#))[\(Forney et al., 2021\)](#), Brandt's cormorant ([Capitolo et al., 2012](#))[\(Capitolo et al.,](#)
15 307 [2012](#)), and common murre ([Carter, 2001](#))[\(Carter, 2001\)](#). Data from Carretta et al. (2022) were
16 308 graphically digitized.
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1109 **Table 1.** Historical bycatch estimation studies and their characteristics (the period column
1110 indicates month and year as MM/YY).

Study	Seasons	Period	Species	Fishery definition	Strata	Formatted:
(Barlow et al., 1994)	1983-87	(see papers)	Pinnipeds/cetaceans	(see included papers)	5-reg	Border: Top: (No border), Bottom: (No border), Left: (No border), Right: (No border), Between : (No border)
(Hanen et al., 1988)	1983-85	Apr 1-Mar 31	Sea-lion, harbor seals	Set nets for halibut/flounder/sharks		
(Diamond and Hanan, 1986)	1983	4/83–3/84	Harbor porpoise	≥8.0" set nets for halibut/flounder	3-regions	Sets
(Hanan et al., 1986)	1984	4/84–3/85	Harbor porpoise	Set nets for halibut/flounder/sharks	3-regions	Sets
(Hanan et al., 1987)	1985	4/85–3/86	Harbor porpoise	Set nets (but not for croaker)	3-regions	Sets
(Hanan and Diamond, 1989)	1986	4/86–3/87	Sea-lion, harbor seal, harbor porpoise	Set nets for halibut/flounder/sharks	5-regions, seasons	Sets
(Konno, 1990)	1987	4/87–3/88	Sea-lion, harbor seal, harbor porpoise	Set nets for halibut/flounder/sharks	5-regions, seasons	Sets
(Perkins et al., 1994)	1988-90	4/88–3/89 4/89–12/89 1/90–12/90	Pinnipeds/cetaceans	Set nets for halibut/angel shark	3-regions	Vessel day
(Lennert et al., 1994)	1990	7/90–12/90	Marine mammals	Set nets for halibut/angel shark	3-regions	Vessel-day
(Perkins et al., 1992a)	1990	7/90–6/91	Pinnipeds/cetaceans	≥8.0" nets for halibut/angel shark	3-regions	Sets
(Perkins et al., 1992b)	1991	1/91–12/91	Pinnipeds/cetaceans	Set nets for halibut/angel shark	SCal/Ventura: Quarters Other strata: Annual	Vessel day
(Julian, 1993)	1992	1/92–12/92	Pinnipeds/cetaceans	Set nets for halibut/angel shark	4-regions: quarterly	Vessel day
(Julian, 1994)	1993	1/93–12/93	Pinnipeds/cetaceans	Set nets for halibut	4-regions	Vessel-day
(Julian and Beeson, 1998)	1990-95	7/90-12/90 1/12, 91-95	Mammals/seabirds/turtles	≥8.5" nets for halibut/angel shark	SCal/Ventura: Quarters Other strata: Annual	Vessel-day
(Cameron and Forney, 1999)	1997-98	1/97–12/97 1/98–12/98	Cetaceans	≥8.5" nets for halibut/angel shark	1997: Geographical+seasonal 1998: Geographical only	Vessel day
(Cameron and Forney, 2000)	1999	1/99–12/99	Cetaceans	≥8.5" nets for halibut/angel shark	Geographical+seasonal	Vessel day
(Carretta, 2001)	2000	1/00–12/00	Cetaceans	≥8.5" nets for halibut/angel shark	SCal/Ventura: Quarters Other strata: Annual	Vessel day
(Carretta, 2002)	2001	1/01–12/01	Cetaceans	Set nets for halibut/angel shark	SCal/Ventura: Quarters Other strata: Annual	Vessel day
(Carretta and Chivers, 2003)	2002	1/02–12/02	Cetaceans	Set nets for halibut/angel shark	SCal/Ventura: Quarters Other regions: Annual	Vessel day
(Carretta and Chivers, 2004)	2003	1/03–12/03	Marine mammals	≥8.5" nets for halibut/angel shark	SCal/Ventura: Quarters Other regions: Annual	Vessel day
(Carretta and Enriquez, 2009)	2007	1/07–12/07	Mammals/seabirds	Set nets for halibut/white seabass	No stratification	Sets
(Carretta and Enriquez, 2012a)	2010	1/10–12/10	Mammals/seabirds	Set nets for halibut/white seabass	No stratification	Vessel day
(Carretta and Enriquez, 2012b)	2011	1/11–12/11	Mammals/seabirds	Set nets for halibut/white seabass	No stratification	Sets
(Carretta et al., 2014)	2012	1/12–12/12	Mammals/seabirds/turtles	Set nets for halibut/white seabass	No stratification	Vessel day

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11 12 **Table 21.** Performance of the best fitting random forest model by species.
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Species	# of bycatch observed	Best model	Mtry	Training data		Test data	
				Kappa	AUC	Kappa	AUC
California sea lion	1372	Weighted-25	3	0.24	0.78	0.23	0.78
Harbor seal	519	Weighted-75	2	0.25	0.83	0.15	0.81
Harbor porpoise	97	Weighted-50	2	0.34	0.98	-0.005	0.98
Common murre	2381	Weighted-25	6	0.71	0.99	0.61	0.97
Brandt's cormorant	118	Weighted-25	8	0.06	0.68	0.07	0.63
Northern elephant seal	78	Weighted-25	1	0.23	0.87	0.21	0.86

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1314 **Supplemental Methods Information**

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1315 **Imputing missing values in observed and logbooks data**

1316 We developed a series of simple assumptions to impute missing values for a few key variables
1317 (GPS coordinates, fishing depth, soak hour, mesh size) reported in the observer data (**Fig. S3A; Table**
1318 **S4**). We assigned missing GPS coordinates using the median coordinates for observed trips within the
1319 statistical block most frequently visited by the vessel – in order of preference – that week, month, or year
1320 based on the logbook data (described below). We derived missing fishing depths by extracting depths
1321 from 25-meter resolution bathymetry data (CDFW, 2002) (**Fig. S3B**). We reassigned missing soak hours
1322 the mode value for a vessel and target species (**Fig S3C**). We reassigned missing mesh sizes the mode for
1323 – in order of preference – the vessel and target species, the target species, or all vessels (**Fig. S3DE**). We
1324 assigned each GPS coordinate to the nearest statistical reporting block (see **Fig. 1A**), which allows points
1325 erroneously falling on land to be assigned a likely statistical block. We derived the distance from shore, a
1326 covariate used to explain bycatch rates in the random forest model, as the distance of each set to the
1327 nearest point on shore.

1328 We developed a series of simple assumptions to impute missing or unrealistic values for a few
1329 key variables (fishing depth, soak hour, mesh size) reported in the logbook data (**Fig. S5A; Table S4**).
1330 We reassigned both missing (including 0 values) and unrealistic fishing depths, which we defined as
1331 depths exceeding the maximum depth in the reported fishing block, the median depth of the fishing block
1332 (**Fig. S5B**). We computed the median and maximum depths of each fishing block using 25-meter
1333 resolution bathymetry data (CDFW, 2002). We reassigned missing soak hours (including 0 values) the
1334 mode value for a vessel. We capped rare and unlikely soak times exceeding 96 hours (4 days) at 96 hours;
1335 however, such soak times could theoretically occur during rough weather when it is unsafe to haul gear
1336 (**Fig. S5C**). We reassigned missing (including 0 values) and unrealistic mesh sizes, which we defined as
1337 mesh sizes exceeding 20 inches, using a hierarchical procedure (**Fig. S5DE**). For logbooks with both
1338 vessel identification and target species information, we assigned the mesh size most commonly used by
1339 the vessel when targeting that target species. For logbooks with only target species information (no vessel
1340 identification), we assigned the mesh size most commonly used when targeting that target species across
1341 all vessels (**Figs. S5 & S6**). We derived the distance from shore, a covariate used to explain bycatch rates
1342 in the random forest model, as the median distance from shore of observed trips within the reported block
1343 given that exact locations are not reported.

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11344 **Bycatch estimation using the random forest models**

11345 We used the best fitting model to generate annual estimates of protected species bycatch from
11346 1981 to 2022 by predicting whether “pseudo-sets” recorded in logbooks were likely to have captured each
11347 study species and assuming median numbers of sets and caught animals for “pseudo-sets” with bycatch.
11348 We predict to pseudo-sets rather than trips because the random forest model is trained on set-level
11349 covariates in the observer data. We used the best fitting model for each species to estimate the probability
11350 that a logged pseudo-set included bycatch of a species then categorized the pseudo-set as with or without
11351 bycatch using a species-specific probability threshold. We derived the species-specific probability
11352 thresholds as the threshold that maximizes Cohen’s kappa when applied to the training datasets (**Fig.**
11353 **S17**). We selected the probability threshold based on Cohen’s kappa rather than the area under the
11354 receiver operator curve (AUC) because (1) the models were tuned and selected based on Cohen’s kappa
11355 and (2) simulation work shows that deriving thresholds based on AUC tends to overestimates the
11356 prevalence of rare events while it underestimates the prevalence of common events (Freeman and Moisen,
11357 2008; Manel et al., 2001). We summed the number of pseudo-sets predicted to have bycatch each year,
11358 converted this sum to “true sets” assuming three sets per pseudo-set (**Fig. S4AB**), and multiplied this sum
11359 by the median number of captures when a capture occurs to generate estimates of the total number of
11360 captured animals (**Fig. S4C**). We opted not to employ a more complex two-stage or hurdle model
11361 approach, where a second model estimates the number of captured individuals when bycatch occurs,
11362 given the rarity of bycatch events larger than one for all species but common murre (**Fig. S4C**).
11363

11364 The random forest models estimate trends in bycatch that are similar to the estimates from the
11365 ratio estimator from 2000-2022 (**Fig. 3**). However, the estimates produced by the two approaches diverge
11366 from 1981-2000 by various extents. While they generally agree for California sea lion back to 2000, the
11367 random forest model underpredicts bycatch relative to the ratio estimator in the late 1990s and
11368 overpredicts in the 1980s and early 1990s (**Fig. 3**). While the approaches generally agree for harbor seal
11369 back to 1995, the random forest model underpredicts bycatch relative to the ratio estimator before 1995,
11370 especially in the Channel Islands and Ventura strata (**Fig. S14**). For common murre, the random forest
11371 model overpredicts bycatch relative to the ratio estimator in the mid- to late-1990s and underpredicts
11372 relative to the ratio estimator in earlier years, especially in Morro and Monterey Bays. These
11373 underpredictions likely occur because of the unequal impacts of lost data from the northern strata in the
11374 1980s (**Fig. 2**). Unlike the random forest models, the ratio estimators are able to use summarized observer
11375 data for this region and time period from old reports. As a result, the ratio estimators can learn from
11376 observations from this region and time period while the random forest models are blind to data from this
11377 region and period. Thus, the random forest models are likely to underpredict risk in early years in
11378 northern strata because they largely learned from late years in southern California, where bycatch risk was
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11378 lower. For this reason, we recommend the use of bycatch estimates from the ratio estimators over the
11379 random forest models until a time when the 1980s observer data is rediscovered.

11380 Comparison to bycatch estimates from historical reports

11381 Our estimates of annual bycatch are generally aligned with estimates from historical studies (Fig. 11382 S11). Slight differences between our estimates and those from historical studies are driven by a mixture of
11383 differences in our methods and input data. While we apply a consistent approach for defining the fishery,
11384 stratifying the data, and estimating bycatch, historical studies have employed variable fishery definitions,
11385 stratification schemes, and estimation methodologies (Table S1). First, we consistently defined the
11386 fishery as using $\geq 3.5"$ set gillnets, while historical studies have considered set gillnets $> 8"$, $> 8.5"$, or of
11387 unspecified sizes (Table S1). Furthermore, it is unclear whether historical studies filtered out fishing sets
11388 based on the reported target species and whether this decision was consistent. We do not define the
11389 fishery based on reported target species given extreme heterogeneity in how this information is reported.
11390 Differences in fishery definitions, as well as differences in data cleaning methods, likely lead to the slight
11391 differences in effort and observed bycatch attributed to the fishery (Fig. S12), which inevitably causes
11392 differences in bycatch estimates. Second, we used a consistent six-region stratification scheme, while
11393 historical studies used a mixture of stratification schemes ranging from no stratification to spatial
11394 stratification to spatial-temporal stratification (Table S1). This impacts the ratio estimators and the
11395 magnitudes of bycatch. Finally, we used vessel days (trips) as the sample unit, whereas historical studies
11396 oscillated between trips and sets as the preferred sample unit (Table S1). This is problematic not only
11397 because of the lack of consistency but also because sets cannot be uniquely identified in the logbook data.

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11398 Identifying set gillnet landing receipts and revenues

11399 We used landing receipts (a.k.a., fish tickets) to estimate ex-vessel revenues generated by the
11400 California $\geq 3.5"$ set gillnet fishery from 1981-2022. Among other information, landing receipts report the
11401 date, value, species, and gear of commercial landings. We identified landing receipts associated with the
11402 $\geq 3.5"$ set gillnet fishery through a multi-step filtering process. First, we filtered the landing receipts to the
11403 five gear types that could include $\geq 3.5"$ mesh set gillnets: trammel nets, set gillnets, small-mesh set
11404 gillnets, large-mesh set gillnets, or entangling nets (Fig. S1A). Entangling nets, which encompass both set
11405 and drift gillnets, were a widely used gear type from 1984-1993. As a result, this filter retained many
11406 swordfish landings and other landings associated with drift gillnets. It also retained many herring landings
11407 and other landings associated with set gillnets with mesh sizes smaller than 3.5 inches. To remove
11408 landing receipts associated with drift gillnets and set gillnets with mesh sizes smaller than 3.5 inches, we
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11409 used gillnet logbooks to identify landing receipts associated with known set gillnet vessels and logged set
11410 gillnet trips. We began by further filtering to only include vessels documented as using $\geq 3.5''$ set gillnets
11411 in the gillnet logbooks (**Fig. S1B**). After this filter, a large amount of swordfish and herring landings
11412 remained, indicating that many $\geq 3.5''$ set gillnet vessels use other gears. Thus, to further tie landing
11413 receipts with known $\geq 3.5''$ set gillnet trips, we explored four related approaches for linking landing
11414 receipts to logged $\geq 3.5''$ set gillnet trips. The first approach was the most strict and only considered
11415 landing receipts reported on the exact day of logged $\geq 3.5''$ set gillnet trips (**Fig. S1C**). This filter
11416 eliminated swordfish and herring landings but is likely to be overly restrictive. The date of landing may
11417 differ from the date of fishing because of misreporting, multi-day trips, or delayed sales. Thus, we
11418 explored three progressively less restrictive rules, which attributed landing receipts recorded within one
11419 (**Fig. S1D**), two (**Fig. S1E**), or three (**Fig. S1F**) days of logged $\geq 3.5''$ set gillnet trips to the fishery. We
11420 selected the landing receipts associated with the 3-day buffer as the final set of landing receipts associated
11421 with the fishery because it effectively eliminated landings of species not associated with the $\geq 3.5''$ set
11422 gillnet fishery (i.e., swordfish and herring) while being inclusive-within-reason of potential $\geq 3.5''$ set
11423 gillnet fishery landings. Finally, we adjusted daily ex-vessel landings values for inflation by converting
11424 all values to January 1, 2022 US dollars using the *priceR* package in R ([Condylios, 2023](#))[\(Condylios, 2023\)](#).
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34426 Mapping species ranges

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36 We mapped the range of the study species using range maps from the California Wildlife Habitat
37 Relationships (CWHR) System ([CDFW, 2021](#))[\(CDFW, 2021\)](#) (**Fig. S15**). The CWHR ranges were
38 developed by species-specific experts. Range maps were not developed for harbor porpoise as part of the
39 CWHR effort. We developed a range map for harbor porpoise assuming that harbor porpoise occur
40 primarily in waters shallower than 50 fathoms (92 meters) north of Point Conception ([Forney et al., 2014](#));[\(Forney et al., 2014\)](#). Harbor seal haulouts were mapped using the CDFW Harbor Seal Haulout
41 GIS dataset ([CDFW, 2014](#))[\(CDFW, 2014\)](#). CDFW conducted aerial surveys of all known haulout sites in
42 2001, 2002, and 2003 and counted the number of harbor seals observed in aerial photographs of each site.
43 We mapped northern elephant seal rookery size in 2010 using a database of counts developed by ([Lowry et al., 2014](#))[\(Lowry et al., 2014\)](#). Counts were generated through a review of ground and aerial
44 photographic surveys. We mapped California sea lion haulouts using data from ([Lowry, 2021](#))[\(Lowry, 2021\)](#). Haulouts were mapped in the Channel Islands between 2016–2019 using aerial photographic
45 surveys. Sea lion haulouts occur along the California coast but were not mapped to single sites in this
46 study and therefore not plotted in our range maps. We mapped seabird colonies using the 2010 CDFW
47 Seabird Colonies Database ([CDFW, 2010](#))[\(CDFW, 2010\)](#). These data were collected as part of the
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11442 Marine Life Protection Act (MLPA) planning process and report the maximum number of seabirds of 26
12443 species at all known colonies.
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Supplemental Tables

Table S1. Historical bycatch estimation studies and their characteristics (the period column indicates month and year as MM/YY).

Study	Seasons	Period	Species	Fishery definition	Stratification scheme	Sample unit
(Barlow et al., 1994)	1983-87	(see papers)	Pinnipeds/cetaceans	(see included papers)	(see included papers)	(see papers)
(Hanen et al., 1988)	1983-85	Apr 1-Mar 31	Sea lion, harbor seals	Set nets for halibut/flounder/sharks	5 regions	Sets
(Diamond and Hanan, 1986)	1983	4/83 - 3/84	Harbor porpoise	>8.0" set nets for halibut/flounder	3 regions	Sets
(Hanen et al., 1986)	1984	4/84 - 3/85	Harbor porpoise	Set nets for halibut/flounder/sharks	3 regions	Sets
(Hanen et al., 1987)	1985	4/85 - 3/86	Harbor porpoise	Set nets (but not for croaker)	3 regions	Sets
(Hanen and Diamond, 1989)	1986	4/86 - 3/87	Sea lion, harbor seal, harbor porpoise	Set nets for halibut/flounder/sharks	5 regions, seasons	Sets
(Konno, 1990)	1987	4/87 - 3/88	Sea lion, harbor seal, harbor porpoise	Set nets for halibut/flounder/sharks	5 regions, seasons	Sets
(Perkins et al., 1994)	1988-90	4/88 - 3/89 4/89 - 12/89 1/90 - 12/90	Pinnipeds/cetaceans	Set nets for halibut/angel shark	3 regions	Vessel-day
(Lennert et al., 1994)	1990	7/90 - 12/90	Marine mammals	Set nets for halibut/angel shark	3 regions	Vessel-day
(Perkins et al., 1992a)	1990	7/90 - 6/91	Pinnipeds/cetaceans	>8.0" nets for halibut/angel shark	3 regions	Sets
(Perkins et al., 1992b)	1991	1/91 - 12/91	Pinnipeds/cetaceans	Set nets for halibut/angel shark	SCal/Ventura: Quarters Other strata: Annual	Vessel-day
(Julian, 1993)	1992	1/92 - 12/92	Pinnipeds/cetaceans	Set nets for halibut/angel shark	4 regions: quarterly	Vessel-day
(Julian, 1994)	1993	1/93 - 12/93	Pinnipeds/cetaceans	Set nets for halibut	4 regions	Vessel-day
(Julian and Beeson, 1998)	1990-95	7/90-12/90 1-12, 91-95	Mammals/seabirds/turtles	>8.5" nets for halibut/angel shark	SCal/Ventura: Quarters Other strata: Annual	Vessel-day
(Cameron and Forney, 1999)	1997-98	1/97 - 12/97 1/98 - 12/98	Cetaceans	>8.5" nets for halibut/angel shark	1997: Geographical+seasonal 1998: Geographical only	Vessel-day
(Cameron and Forney, 2000)	1999	1/99 - 12/99	Cetaceans	>8.5" nets for halibut/angel shark	Geographical+seasonal	Vessel-day
(Carretta, 2001)	2000	1/00 - 12/00	Cetaceans	>8.5" nets for halibut/angel shark	SCal/Ventura: Quarters Other strata: Annual	Vessel-day
(Carretta, 2002)	2001	1/01 - 12/01	Cetaceans	Set nets for halibut/angel shark	SCal/Ventura: Quarters Other strata: Annual	Vessel-day
(Carretta and Chivers, 2003)	2002	1/02 - 12/02	Cetaceans	Set nets for halibut/angel shark	SCal/Ventura: Quarters Other regions: Annual	Vessel-day
(Carretta and Chivers, 2004)	2003	1/03 - 12/03	Marine mammals	>8.5" nets for halibut/angel shark	SCal/Ventura: Quarters Other regions: Annual	Vessel-day
(Carretta and Enriquez, 2009)	2007	1/07 - 12/07	Mammals/seabirds	Set nets for halibut/white seabass	No stratification	Sets
(Carretta and Enriquez, 2012a)	2010	1/10 - 12/10	Mammals/seabirds	Set nets for halibut/white seabass	No stratification	Vessel-day
(Carretta and Enriquez, 2012b)	2011	1/11 - 12/11	Mammals/seabirds	Set nets for halibut/white seabass	No stratification	Sets

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(Carretta et al., 2014)	2012	1/12 - 12/12	Mammals/seabirds/turtles	Set nets for halibut/white seabass	No stratification	Vessel-day
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Table S2. Number of fishing trips in the California 3.5 inch mesh set gillnet fishery by year.

Year	# of vessels	# of trips		Percent of trips		
		Total	Observed	Unobserved	Observed	Unobserved
1981	162	6139	0	6139	0.0%	100.0%
1982	222	9218	0	9218	0.0%	100.0%
1983	262	9012	143	8869	1.6%	98.4%
1984	357	12374	255	12119	2.1%	97.9%
1985	378	14314	238	14076	1.7%	98.3%
1986	379	13336	217	13119	1.6%	98.4%
1987	290	10667	213	10454	2.0%	98.0%
1988	211	8585	120	8465	1.4%	98.6%
1989	198	7811	58	7753	0.7%	99.3%
1990	182	6836	167	6669	2.4%	97.6%
1991	158	6668	716	5952	10.7%	89.3%
1992	117	5611	744	4867	13.3%	86.7%
1993	115	5754	959	4795	16.7%	83.3%
1994	74	2455	150	2305	6.1%	93.9%
1995	70	2616	0	2616	0.0%	100.0%
1996	59	2654	0	2654	0.0%	100.0%
1997	66	3310	0	3310	0.0%	100.0%
1998	59	2889	0	2889	0.0%	100.0%
1999	63	4026	165	3861	4.1%	95.9%
2000	66	3828	66	3762	1.7%	98.3%
2001	63	3289	0	3289	0.0%	100.0%
2002	54	3395	0	3395	0.0%	100.0%
2003	51	2779	0	2779	0.0%	100.0%
2004	47	2627	0	2627	0.0%	100.0%
2005	40	1930	0	1930	0.0%	100.0%
2006	41	1658	5	1653	0.3%	99.7%
2007	44	1797	65	1732	3.6%	96.4%
2008	43	1936	0	1936	0.0%	100.0%
2009	41	1934	0	1934	0.0%	100.0%
2010	33	1544	64	1480	4.1%	95.9%
2011	39	1575	55	1520	3.5%	96.5%
2012	35	1374	76	1298	5.5%	94.5%
2013	32	968	48	920	5.0%	95.0%
2014	25	819	0	819	0.0%	100.0%
2015	25	1014	0	1014	0.0%	100.0%
2016	24	1077	0	1077	0.0%	100.0%
2017	21	840	69	771	8.2%	91.8%
2018	19	948	0	948	0.0%	100.0%
2019	18	1151	0	1151	0.0%	100.0%
2020	18	841	0	841	0.0%	100.0%
2021	16	870	0	870	0.0%	100.0%

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Table S3. Bycatch rates extracted from historical reports (* mark years with raw observer data where the summary values from the historical reports are not needed).

Reference	Species	Region	Year	Estimated #			Raw data available?
				# sets observed	trips observed	Catch per trip	
Hanan et al. 1988	California sea lion	Channel Islands	1985	180	60.00	44	0.733 *
Hanan & Diamond 1989	California sea lion	Channel Islands	1986	66	22.00	54	2.455 *
Hanan et al. 1988	California sea lion	Monterey Bay	1983	22	7.33	12	1.636
Hanan et al. 1988	California sea lion	Monterey Bay	1984	126	42.00	19	0.452
Hanan et al. 1988	California sea lion	Monterey Bay	1985	49	16.33	5	0.306
Hanan & Diamond 1989	California sea lion	Monterey Bay	1986	36	12.00	4	0.333
Hanan et al. 1988	California sea lion	Morro Bay	1983	288	96.00	41	0.427
Hanan et al. 1988	California sea lion	Morro Bay	1984	374	124.67	22	0.176
Hanan et al. 1988	California sea lion	Morro Bay	1985	317	105.67	25	0.237
Hanan & Diamond 1989	California sea lion	Morro Bay	1986	137	45.67	13	0.285 *
Hanan et al. 1988	California sea lion	San Francisco	1983	158	52.67	4	0.076
Hanan et al. 1988	California sea lion	San Francisco	1984	300	100.00	8	0.080
Hanan et al. 1988	California sea lion	San Francisco	1985	348	116.00	3	0.026
Hanan & Diamond 1989	California sea lion	San Francisco	1986	419	139.67	2	0.014
Hanan et al. 1988	California sea lion	Southern California	1983	430	143.33	16	0.112 *
Hanan et al. 1988	California sea lion	Southern California	1984	571	190.33	13	0.068 *
Hanan et al. 1988	California sea lion	Southern California	1985	339	113.00	5	0.044 *
Hanan & Diamond 1989	California sea lion	Southern California	1986	425	141.67	15	0.106 *
Hanan et al. 1988	California sea lion	Ventura	1983	430	143.33	16	0.112 *
Hanan et al. 1988	California sea lion	Ventura	1984	571	190.33	13	0.068 *
Hanan et al. 1988	California sea lion	Ventura	1985	339	113.00	5	0.044 *
Hanan & Diamond 1989	California sea lion	Ventura	1986	425	141.67	15	0.106 *
Diamond & Hanan 1986	Harbor porpoise	Monterey Bay	1983	22	7.33	2	0.273
Hanan et al. 1986	Harbor porpoise	Monterey Bay	1984	126	42.00	2	0.048
Hanan et al. 1987	Harbor porpoise	Monterey Bay	1985	49	16.33	2	0.122
Hanan & Diamond 1989	Harbor porpoise	Monterey Bay	1986	36	12.00	1	0.083
Diamond & Hanan 1986	Harbor porpoise	Morro Bay	1983	288	96.00	7	0.073
Hanan et al. 1986	Harbor porpoise	Morro Bay	1984	374	124.67	3	0.024
Hanan et al. 1987	Harbor porpoise	Morro Bay	1985	317	105.67	3	0.028
Hanan & Diamond 1989	Harbor porpoise	Morro Bay	1986	137	45.67	3	0.066 *
Diamond & Hanan 1986	Harbor porpoise	San Francisco	1983	151	50.33	5	0.099
Hanan et al. 1986	Harbor porpoise	San Francisco	1984	299	99.67	14	0.140
Hanan et al. 1987	Harbor porpoise	San Francisco	1985	348	116.00	28	0.241
Hanan & Diamond 1989	Harbor porpoise	San Francisco	1986	419	139.67	12	0.086
Hanan et al. 1988	Harbor seal	Channel Islands	1985	180	60.00	13	0.217 *
Hanan et al. 1988	Harbor seal	Monterey Bay	1983	22	7.33	0	0.000
Hanan et al. 1988	Harbor seal	Monterey Bay	1984	126	42.00	10	0.238
Hanan et al. 1988	Harbor seal	Monterey Bay	1985	49	16.33	7	0.429
Hanan & Diamond 1989	Harbor seal	Monterey Bay	1986	36	12.00	9	0.750
Hanan et al. 1988	Harbor seal	Morro Bay	1983	288	96.00	17	0.177
Hanan et al. 1988	Harbor seal	Morro Bay	1984	374	124.67	29	0.233
Hanan et al. 1988	Harbor seal	Morro Bay	1985	317	105.67	84	0.795
Hanan & Diamond 1989	Harbor seal	Morro Bay	1986	137	45.67	25	0.547 *
Hanan et al. 1988	Harbor seal	San Francisco	1983	158	52.67	11	0.209
Hanan et al. 1988	Harbor seal	San Francisco	1984	300	100.00	22	0.220
Hanan et al. 1988	Harbor seal	San Francisco	1985	348	116.00	38	0.328
Hanan & Diamond 1989	Harbor seal	San Francisco	1986	419	139.67	60	0.430
Hanan et al. 1988	Harbor seal	Southern California	1983	430	143.33	0	0.000 *
Hanan et al. 1988	Harbor seal	Southern California	1984	571	190.33	4	0.021 *
Hanan et al. 1988	Harbor seal	Southern California	1985	339	113.00	2	0.018 *
Hanan & Diamond 1989	Harbor seal	Southern California	1986	425	141.67	8	0.056 *
Hanan et al. 1988	Harbor seal	Ventura	1983	430	143.33	0	0.000 *
Hanan et al. 1988	Harbor seal	Ventura	1984	571	190.33	4	0.021 *
Hanan et al. 1988	Harbor seal	Ventura	1985	339	113.00	2	0.018 *
Hanan & Diamond 1989	Harbor seal	Ventura	1986	425	141.67	8	0.056 *

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Table S4. Assumptions made throughout the analysis, their likelihood, and their potential impact on the results (*assumptions that are expected to be valid or true on average are not likely to impact the results).

Assumption	Likelihood and potential impact of assumption
Data analysis	
Ratio estimation	
1. Ratio estimation assumes that the rate of bycatch for observed fishing trips equals the rate for all fishing trips in a given stratum.	This assumption appears valid as the traits of observed trips are representative of the unobserved trips (Fig. S7) and we stratified our analysis using the 7-region scheme recommended in other studies (Fig. S8).
2. We assumed that the bycatch rate in years without observer data was equal to the bycatch rate in the closest year with data.	The validity of this assumption is difficult to assess given the inconsistency in the observer program (Fig. 2BC); however, it is this inconsistency that also makes the assumption necessary. Ultimately, our finding that recent bycatch is low is not likely sensitive to this assumption, as the effects of substantial decreases in fishing effort overwhelm the effects of changes in bycatch rates.
Bycatch estimation using the random forest models*	
1. We assumed that a “pseudo-set” (roughly equivalent to a trip) is equivalent to 3 sets (Fig. S4AB).	This assumption is likely to be true on average since it reflects the median value, which has been consistent through time (Fig. S4AB).
2. We assumed the number of captures per set is the median number of captures when a capture occurs (Fig. S4C).	This assumption is likely to be true on average since it reflects the median value and that for most of the evaluated species, deviations from the median have been rare (Fig. S4C).
Data imputation	
Observer data	
1. We assigned missing GPS coordinates using the median coordinates for observed trips within the statistical block – in order of preference – that week, month, or year based on the logbook data.	This assumption is likely to be true on average given that it uses vessel-specific median values.
2. We derived missing fishing depths by extracting depths from 25-meter resolution bathymetry data (Fig. S3B).	If the GPS coordinates were accurately reported, then the extracted depth estimates are accurate.
3. We reassigned missing soak hours the mode value for a vessel and target species.	This assumption is likely to be true on average given that it uses vessel-specific median values.
4. We reassigned missing mesh sizes the mode for – in order of preference – the vessel and target species, the target species, or all vessels (Fig. S3DE).	This assumption is likely to be true on average given that it uses mode values that are vessel-specific to the greatest extent possible.
5. We assigned each GPS coordinate to the nearest statistical reporting block, which allows points erroneously falling on land to be assigned a likely statistical block.	If the GPS coordinates were close to accurate, then this procedure is accurate on average.
Logbook data	
1. We reassigned both missing and unrealistic fishing depths, defined as depths exceeding the maximum depth the median value, in the reported fishing block, the median depth of the fishing block (Fig. S5B).	This assumption is likely to be true on average given it uses vessel-specific median values.
2. We reassigned missing soak hours the mode value for a vessel.	This assumption is likely to be true on average given that it uses vessel-specific mode values.
3. We capped unlikely soak times exceeding 96 hours at 96 hours.	The validity of this assumption is unclear because the values could be typos; however, values >96 hours were rare and exert little leverage on the results.

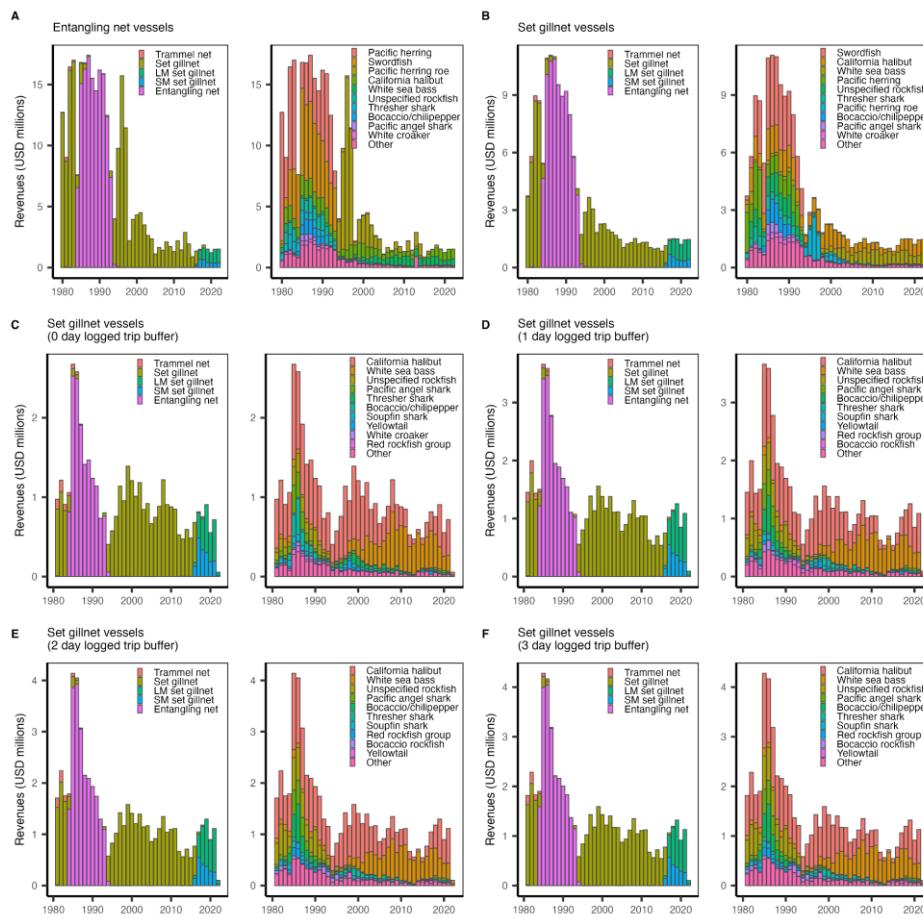
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11 4. We reassigned missing and unrealistic (>20 inch) mesh sizes using a hierarchical procedure (Fig. S5DE). For logbooks with both vessel id and target species info, we possible.
12 uses mode values that are vessel-specific to the greatest extent
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14 assigned the mesh size most commonly used by the vessel when targeting that target species. For logbooks
15 with only target species info (no vessel id), we assigned the mesh size most commonly used when targeting that
16 target species across all vessels (Figs. S5 & S6).
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18 1457 ** *This analysis is informational and only included in the supplemental information.*
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11 459 **Supplemental Figures**

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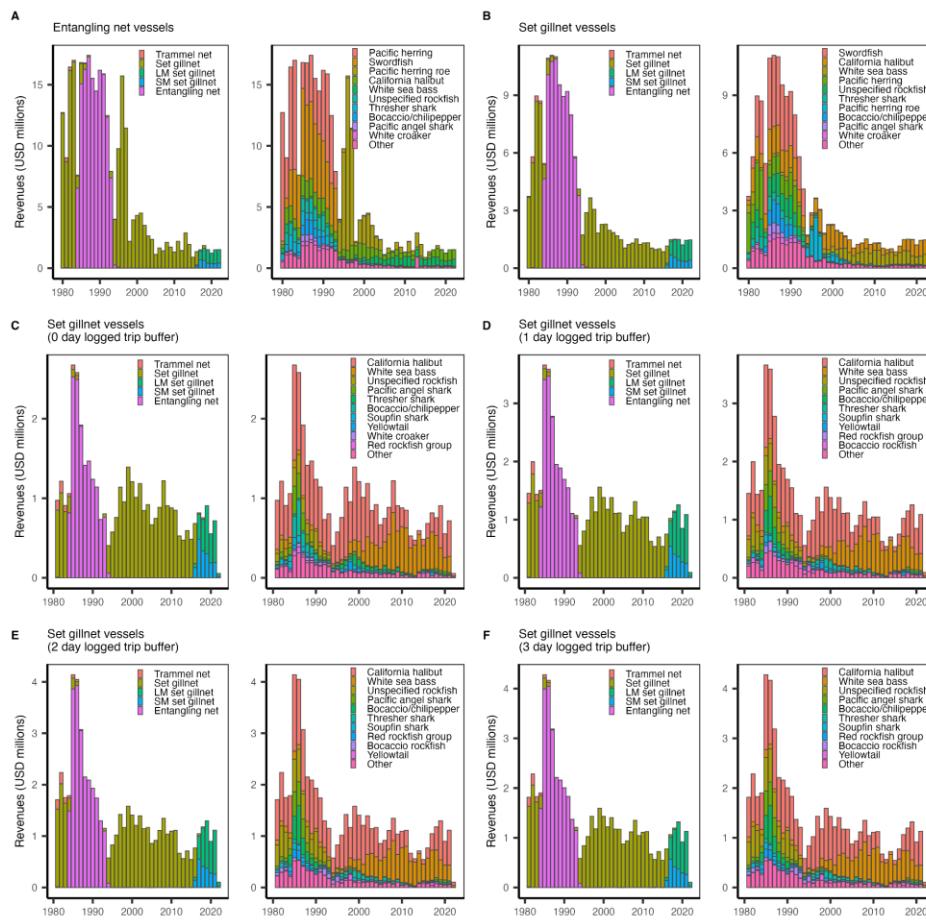


Fig. S1. Estimated ex-vessel revenues generated by the California $\geq 3.5"$ set gillnet fishery as estimated through six different filtration procedures. The filtration procedures examine the sum annual ex-vessel revenues reported on landing receipts from (A) vessels using various reported entangling net gears; (B) vessels using various entangling net gears that are known to use set gillnets based on logbooks; and (C-F) vessels known to use set gillnets based on logbooks that are dated within various buffers of a logged set gillnet trip. We adopted the final filter, which sums landing receipts date within 3 days of logged set gillnet trip, as the best estimate of ex-vessel revenues for the fishery. Revenues have not been adjusted for inflation (see Fig. 1D for the inflation adjusted ex-vessel revenues).

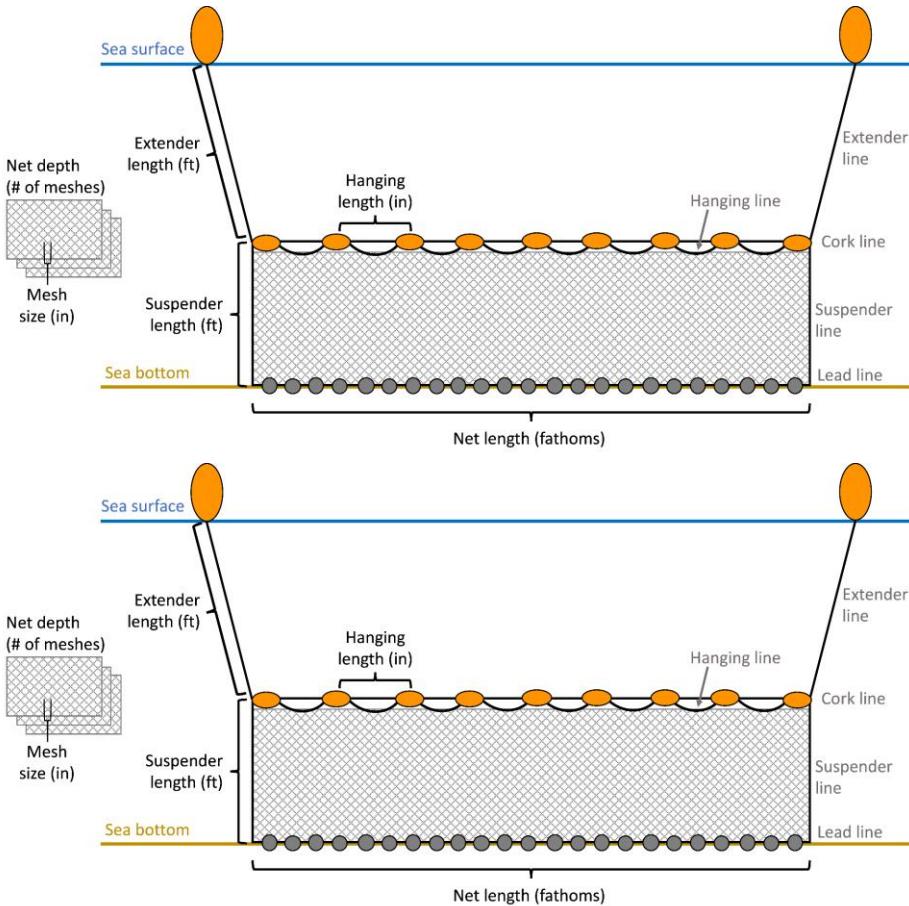
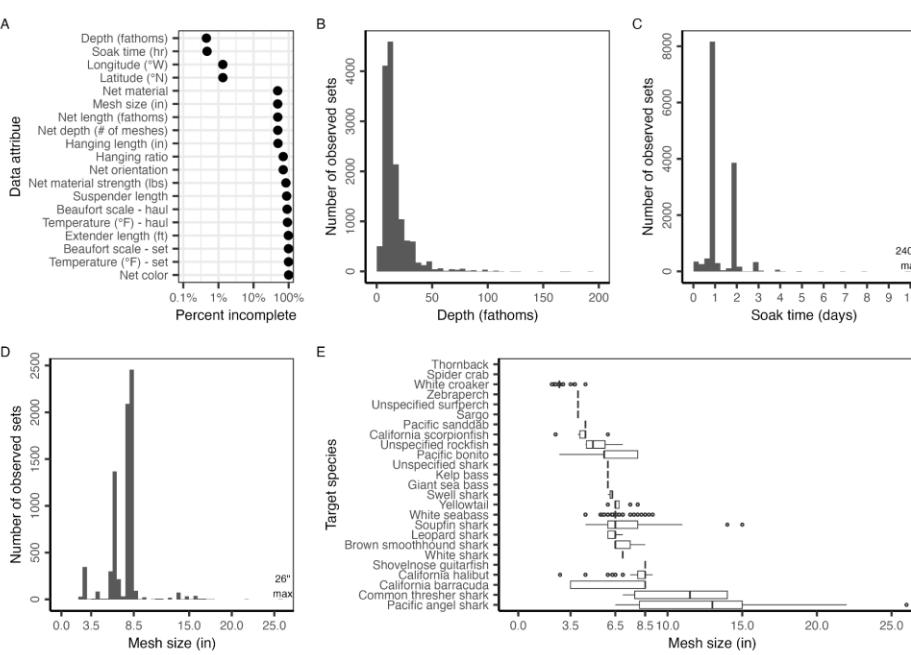


Fig. S2. Diagram of a typical California set gillnet illustrating the measurements reported in the logbook and observer data. Grey text indicates parts of the gillnet and black text indicates the reported measurements. The material, strength, and color of the net is also reported. Finally, the hanging ratio, a percentage that is calculated as the length of the mesh web divided by the length of the cork line, is reported.

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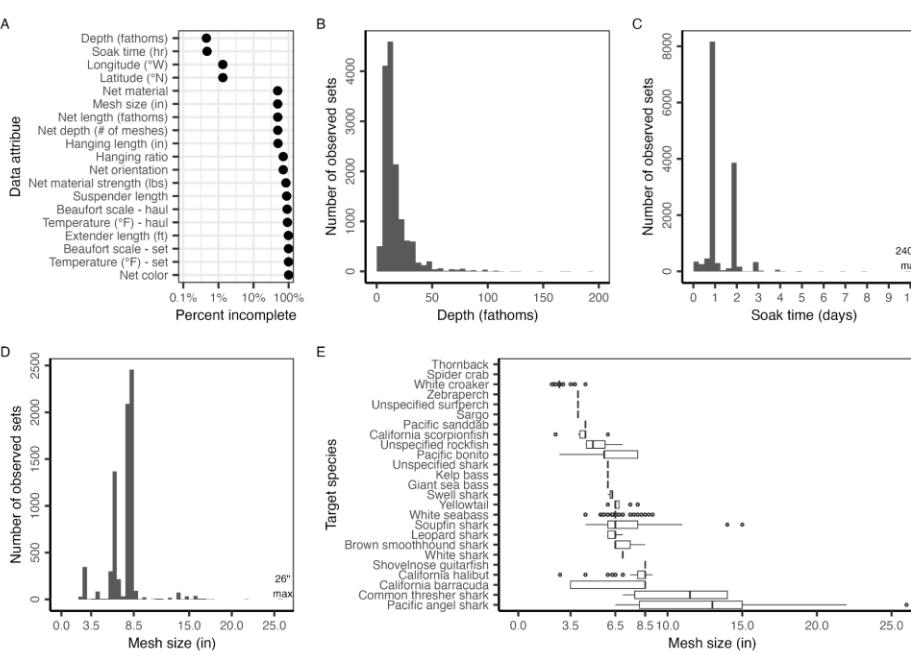
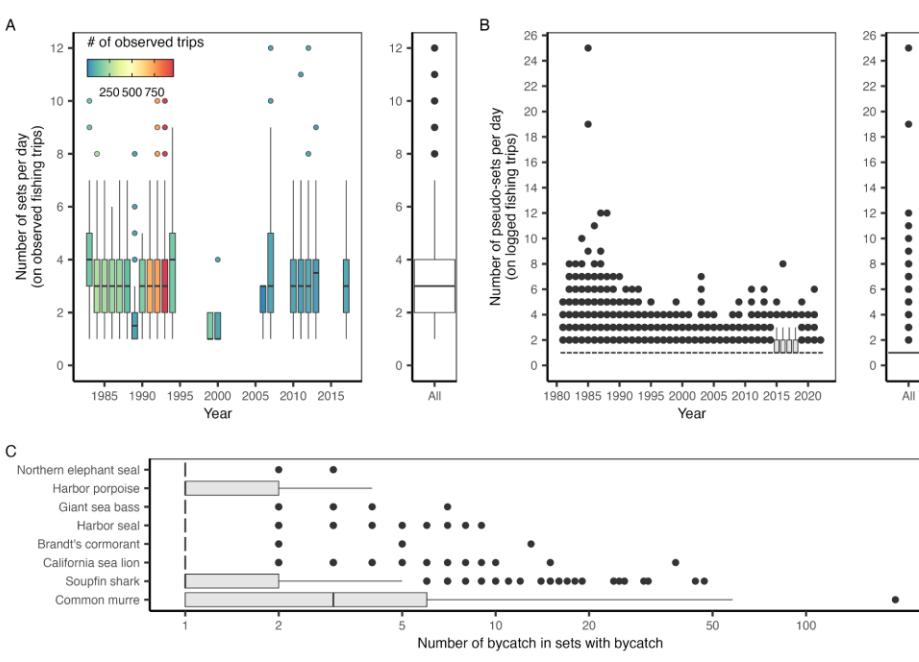


Fig. S3. Traits of the observed set gillnet metadata. Panel **A** shows the level of completeness of gillnet metadata. Panel **B** shows the distribution of reported depths. Panel **C** shows the distribution of reported soak times; the maximum reported soak time is 240 hours (10 days). Panel **D** shows the distribution of reported mesh sizes; the maximum reported mesh size is 26 inches. Panel **E** shows the distribution of reported mesh sizes by reported target species. In the boxplots, the solid line indicates the median, the box indicates the interquartile range (IQR; 25th to 75th percentiles), the whiskers indicate 1.5 times the IQR, and points indicate outliers.



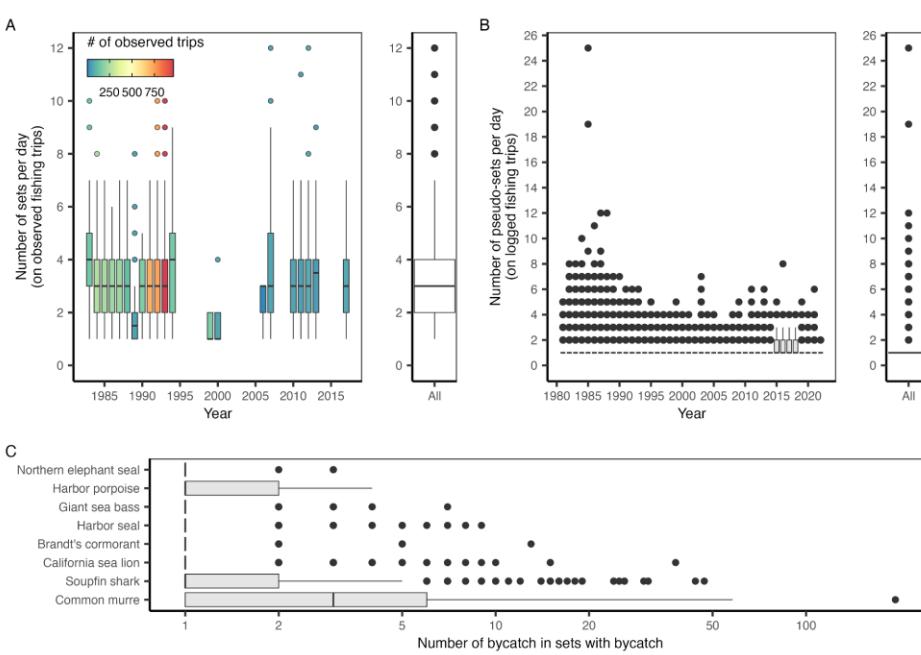
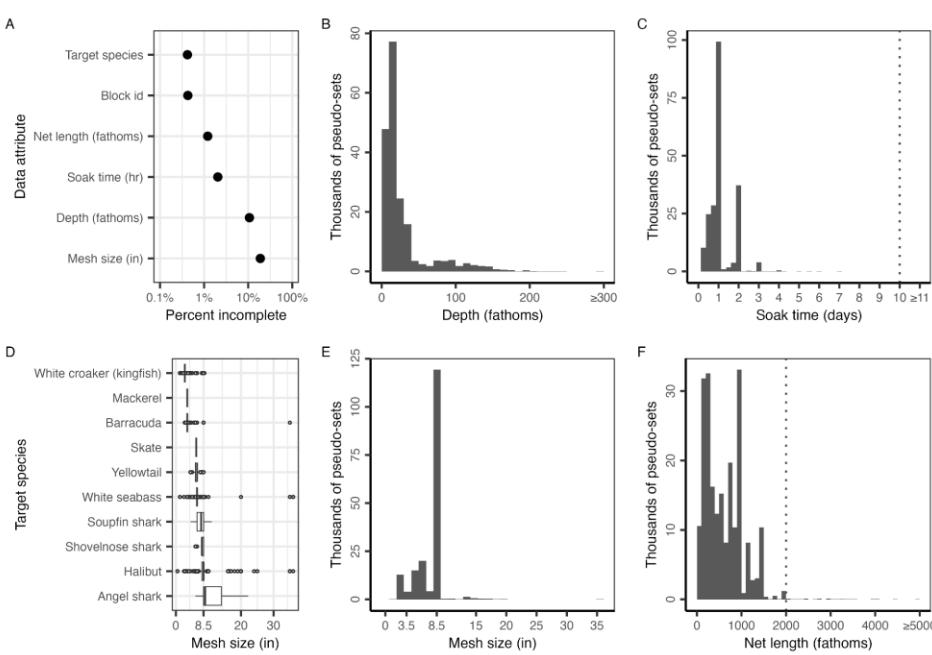


Fig. S4. The (A) number of sets per observed fishing trip by year and overall; (B) number of pseudo-sets per logged fishing trip by year and overall; and (C) number of bycatch in sets with bycatch of each of the species of interest. In the boxplots, the solid line indicates the median, the box indicates the interquartile range (IQR; 25th to 75th percentiles), the whiskers indicate 1.5 times the IQR, and points indicate outliers. In (A), the fill color indicates the number of observed fishing trips contributing to the annual distribution.



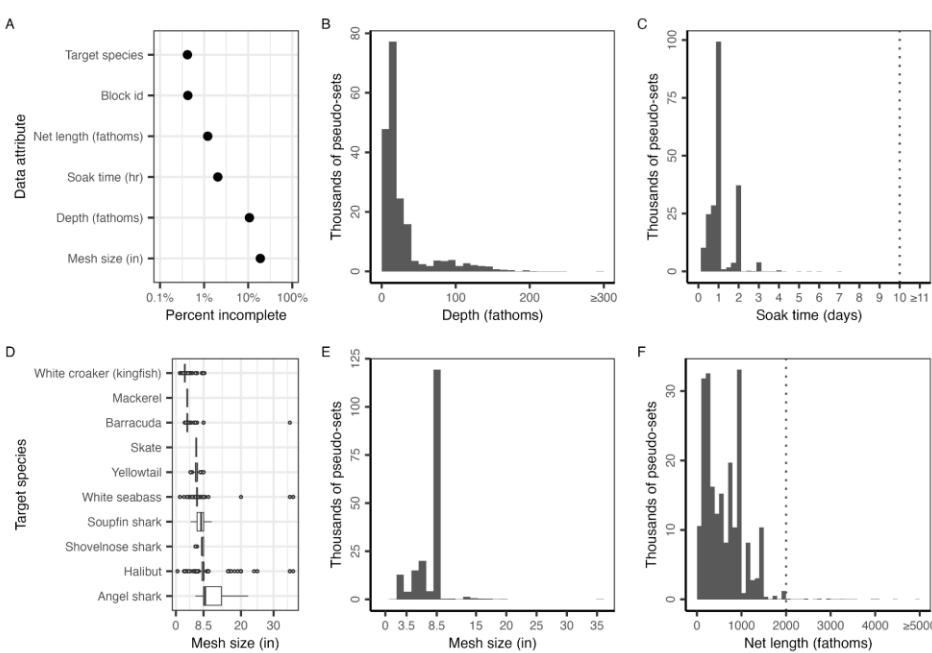
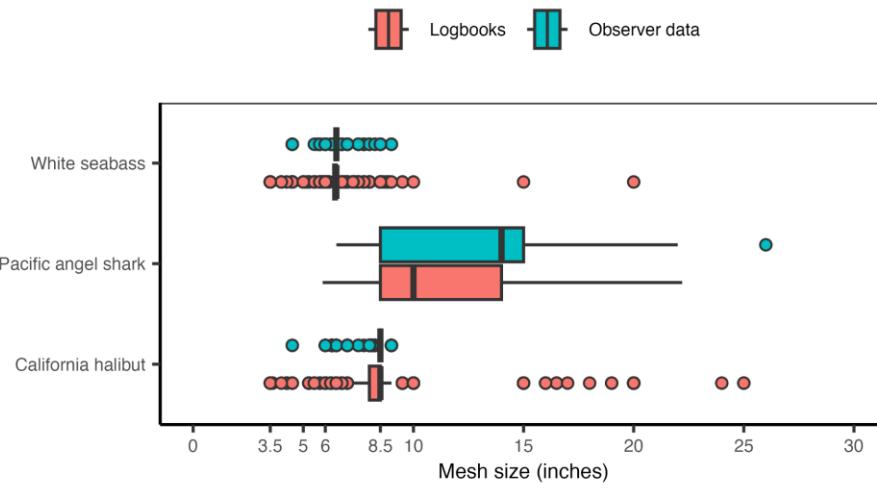
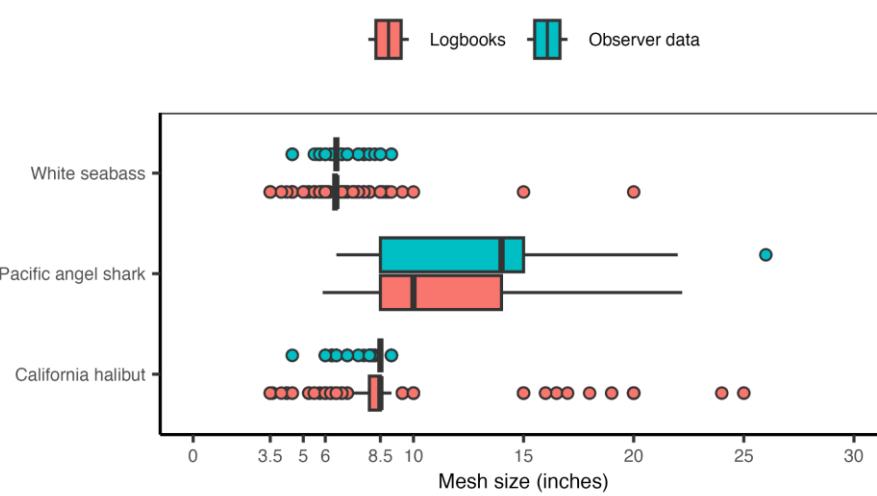
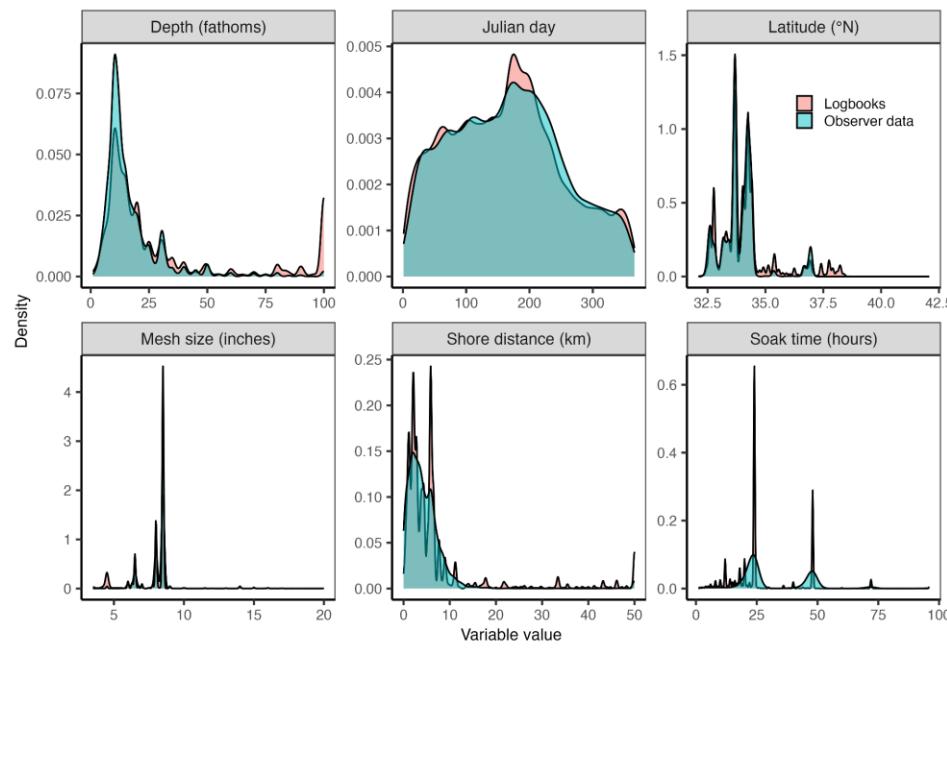


Fig. S5. Traits of the set gillnet sets documented in the logbook data. Panel **A** shows the level of completeness of gillnet logbook metadata. Panel **B** shows the distribution of reported depths. Panel **C** shows the distribution of reported soak times. The maximum reported soak time in the observer data is 10 days; rare values larger than this value were assumed to be unrealistic and were capped at the maximum. Panel **D** shows the distribution of reported mesh sizes by reported target species. In the boxplots, the solid line indicates the median, the box indicates the interquartile range (IQR; 25th to 75th percentiles), the whiskers indicate 1.5 times the IQR, and points indicate outliers. Panel **E** shows the distribution of reported mesh sizes. Panel **F** shows the distribution of reported net lengths.



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4506 **Fig. S6.** Mesh size (inches) by target species in the logbook (red) and observer (blue) data. Since 1989,
4507 California halibut and Pacific angel shark can only be targeted using mesh sizes larger than 8.5 inches.
4508 White seabass are typically targeted using a minimum mesh size of 6.0 inches; however, a small amount
4509 of incidental take (<20% of catch and ≤ 10 individuals) in mesh sizes between 3.5 to 6.0 inches is allowed
4510 from June 16 to March 14 (14, § 155.10).
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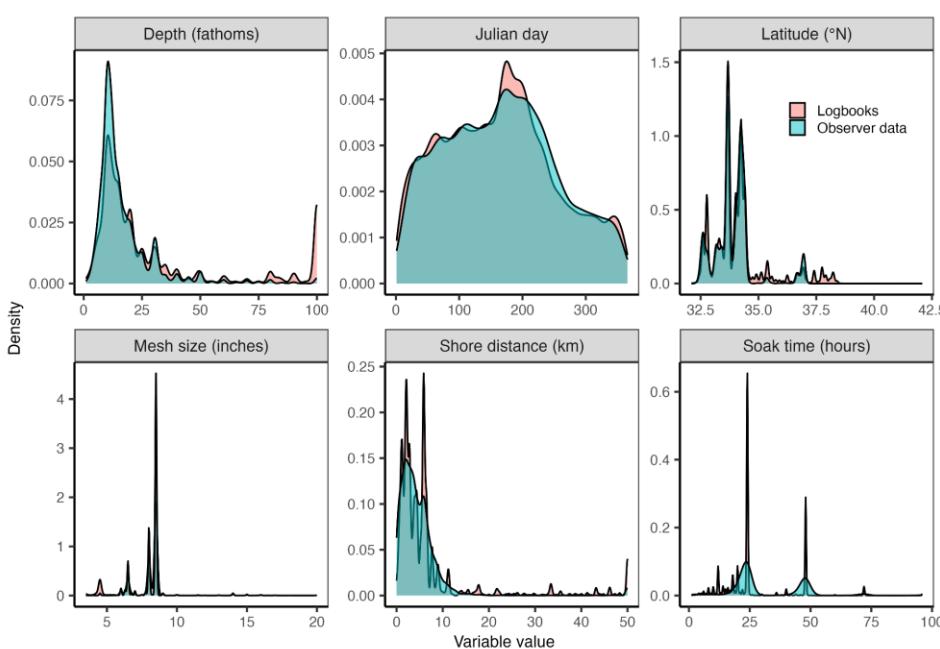
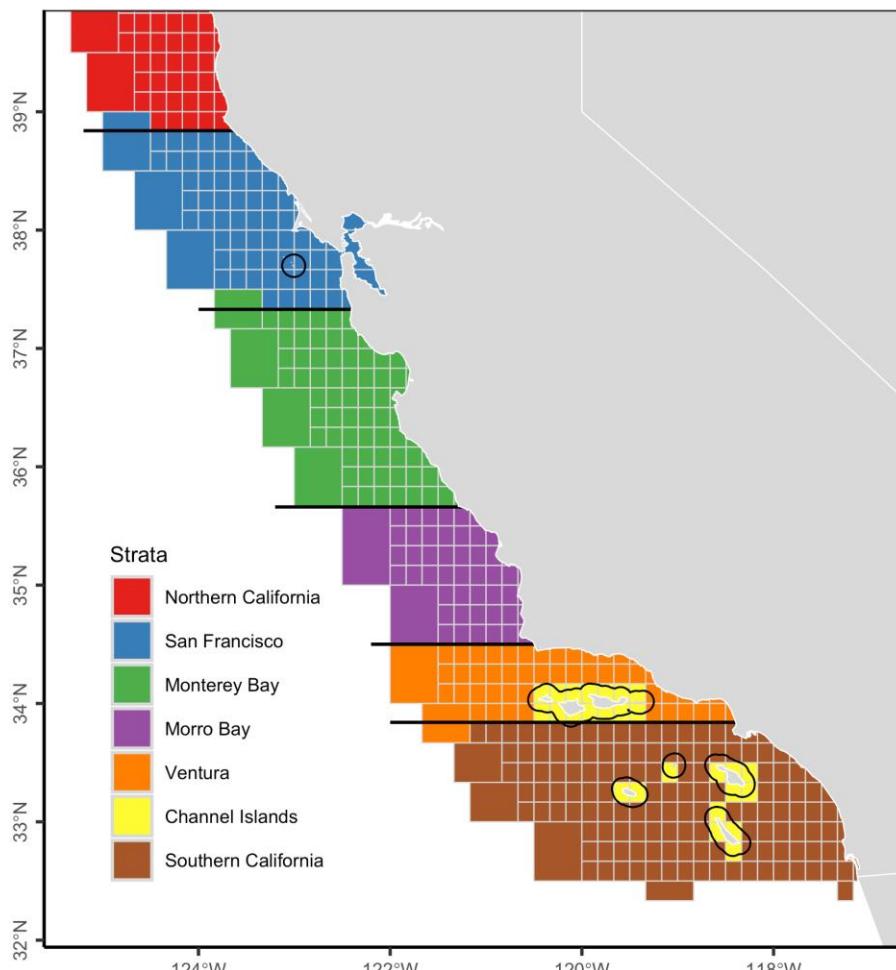


Fig. S7. Comparison of key set traits in the logbook (red) and observer (blue) data. We used a two-sided Kolmogorov-Smirnov test to confirm that the traits of the logbook and observer data could have come from the same probability distribution (all p-values < 0.001) could have come from the same probability distribution (all p-values < 0.001).



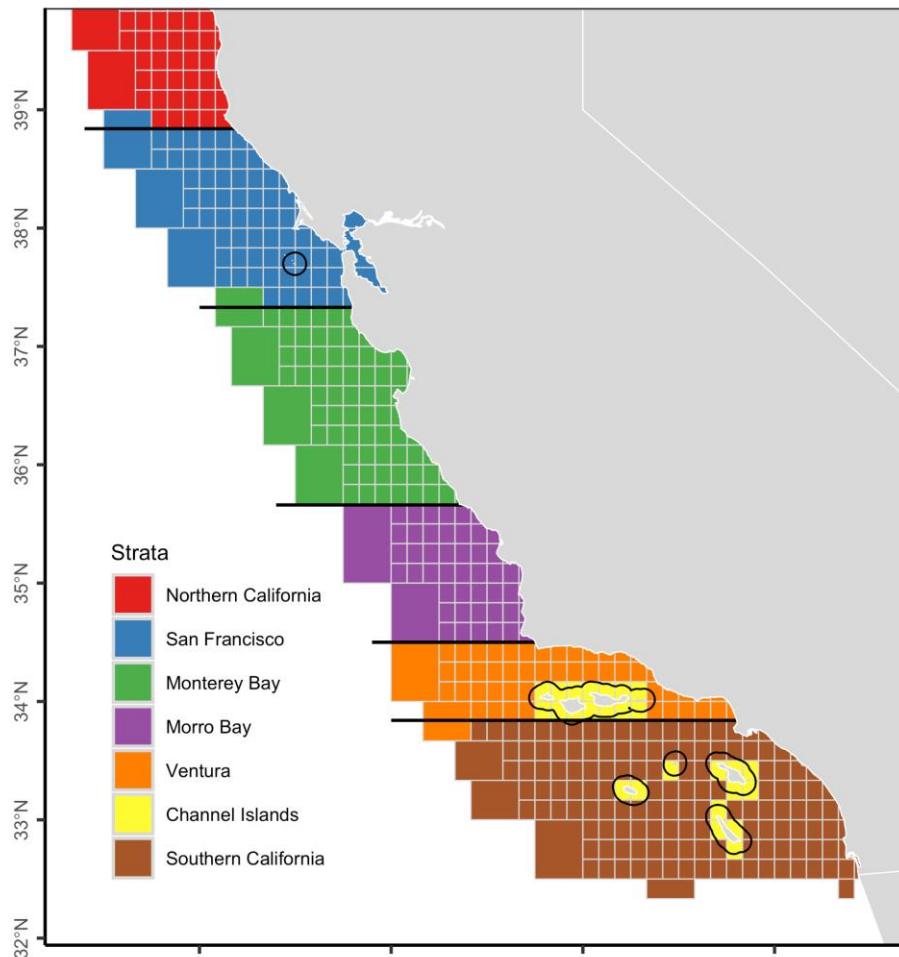


Fig. S8. The regional stratification scheme used throughout the analysis. The stratification scheme north of Point Conception was originally proposed by Diamond and Hanan (1986). The stratification scheme south of Point Conception was originally proposed by Julian (1993). The dark black lines around the Channel Islands show the 10 km buffer used to identify island-associated fishing trips.

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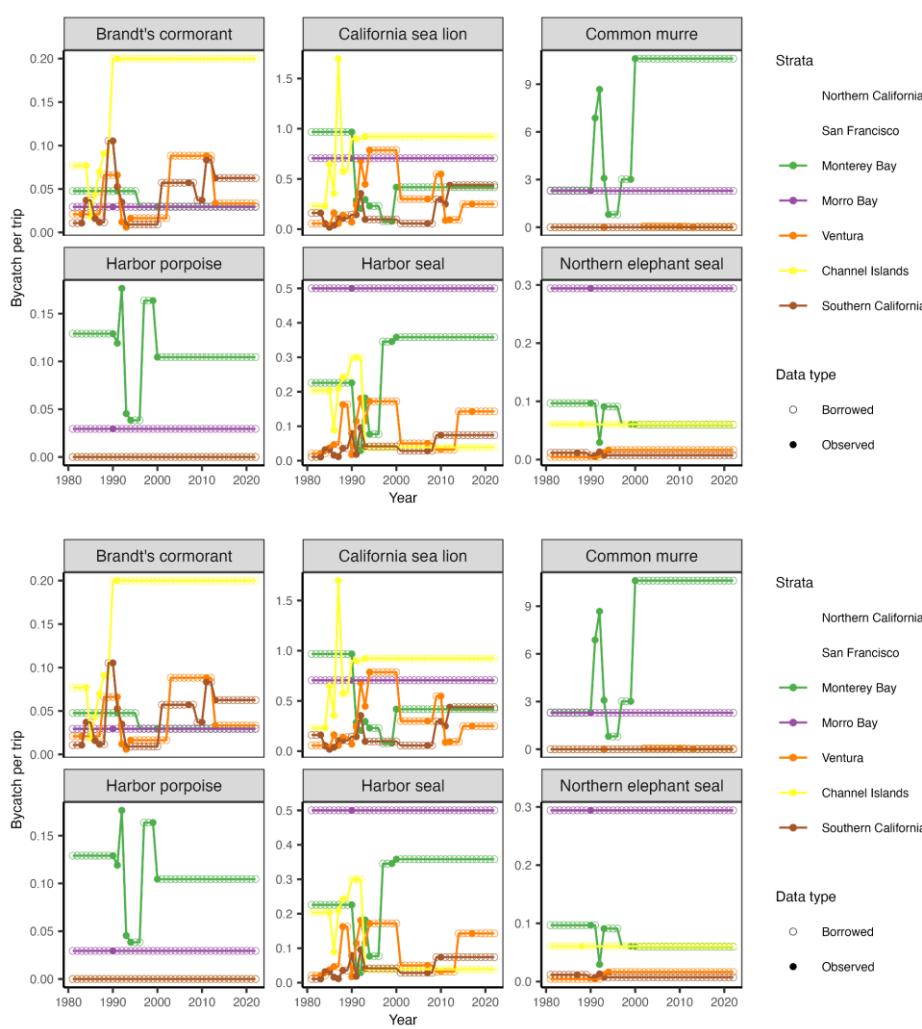
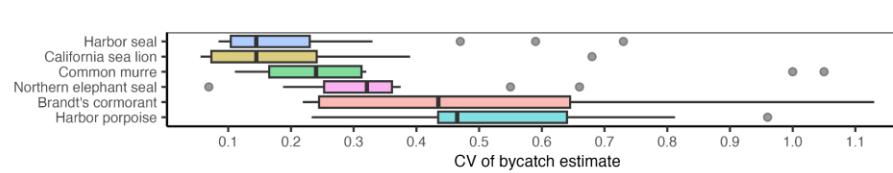
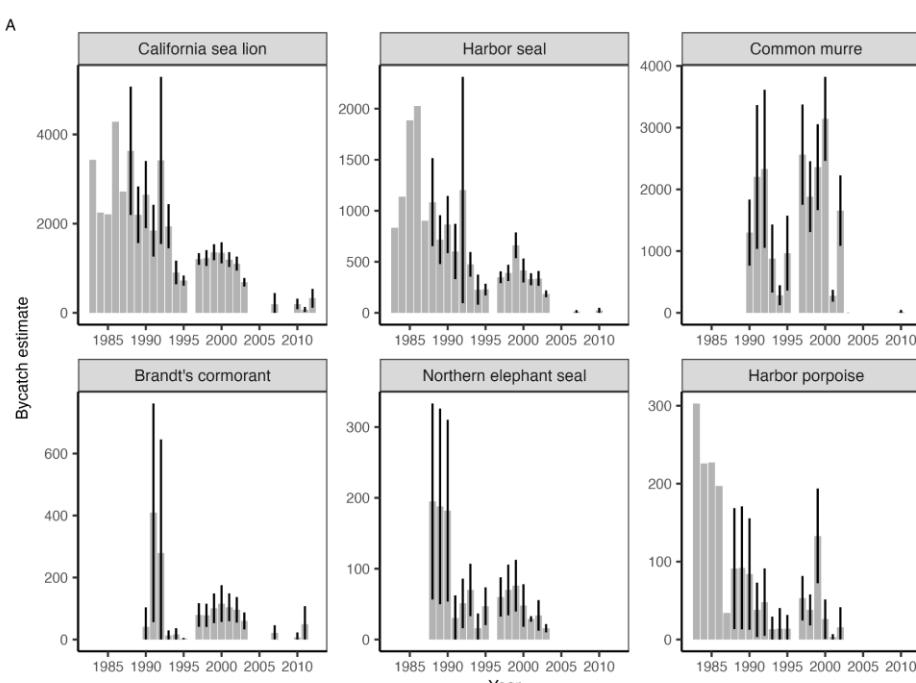


Fig. S9. Annual bycatch rates by species and regional strata as assumed in the ratio estimation analysis. Solid circles indicate years with observer data and open circles indicate years whose bycatch rates are borrowed from the closest year with observer data. See **Figs. 1A and S8** for maps of the regional strata.



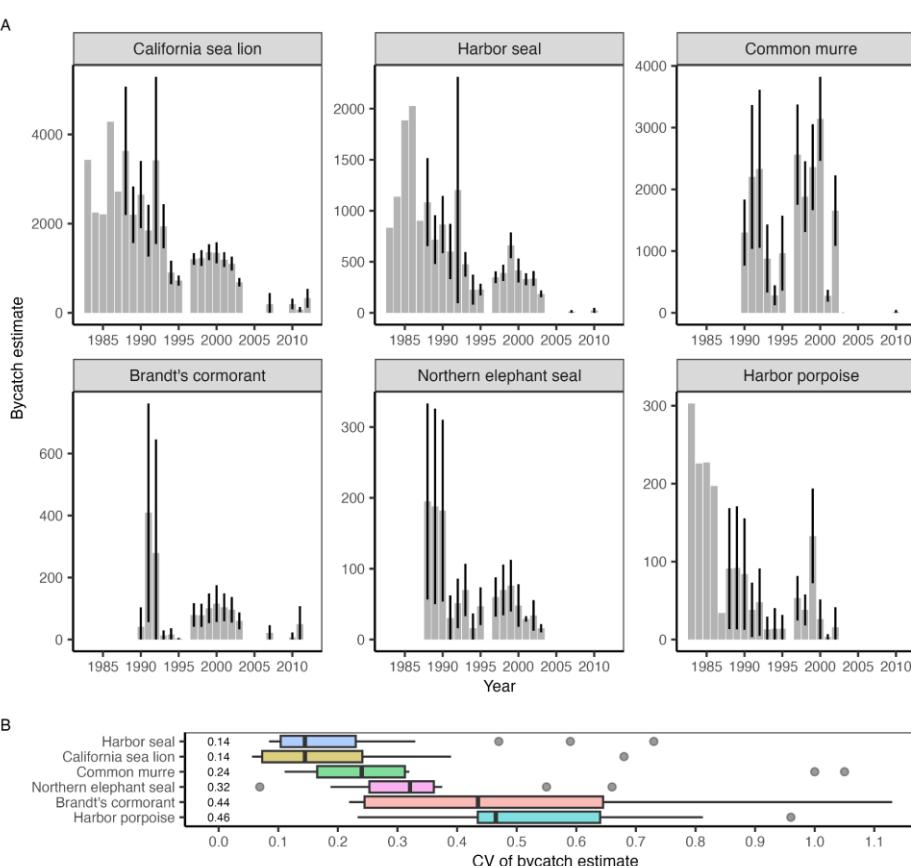


Fig. S10. Estimates of (A) annual bycatch from historical studies and (B) the uncertainty of these estimates expressed as the coefficient of variation (CV). In (A), error bars indicate 95% confidence intervals. See [Table 4S1](#) for the sources of these estimates. [In \(B\), the median CV of the bycatch estimates for each species is printed on the far left.](#) In the boxplots, the solid line indicates the median, the box indicates the interquartile range (IQR; 25th to 75th percentiles), the whiskers indicate 1.5 times the IQR, and points indicate outliers.

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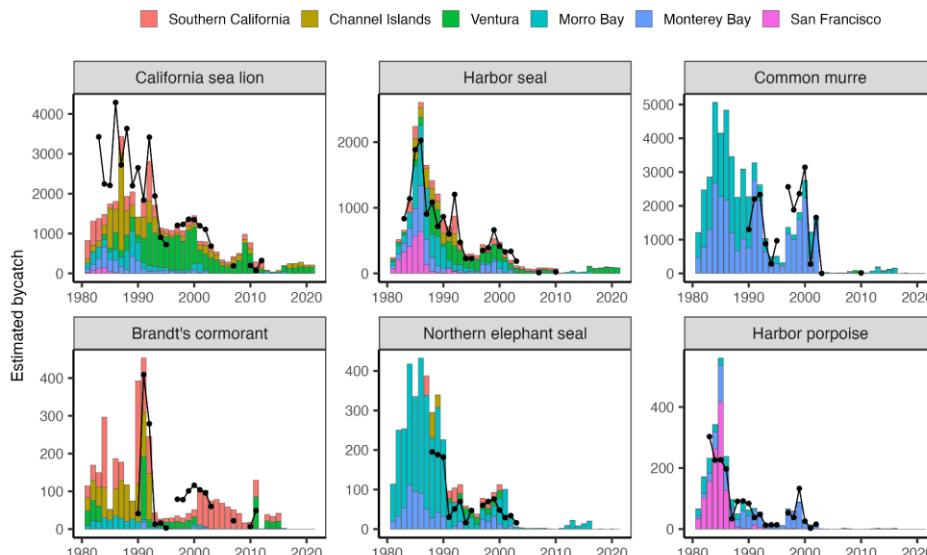
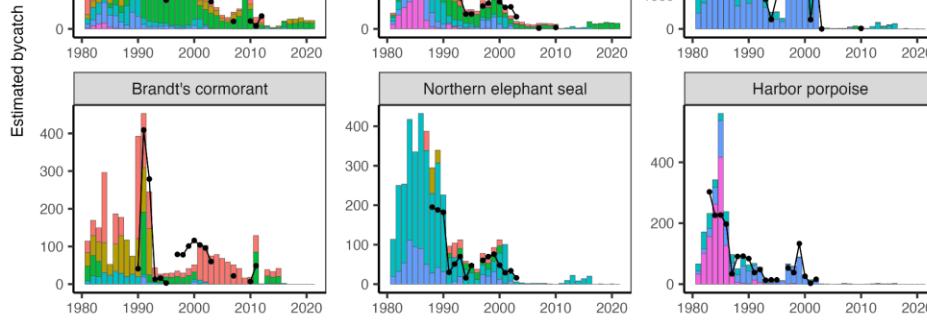
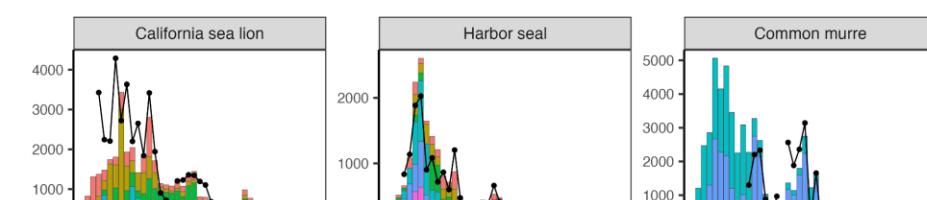


Fig. S11. A comparison of estimates of annual bycatch from our study (bars) and historical studies (points and lines). Potential reasons for these differences are explored in **Fig. S12.**

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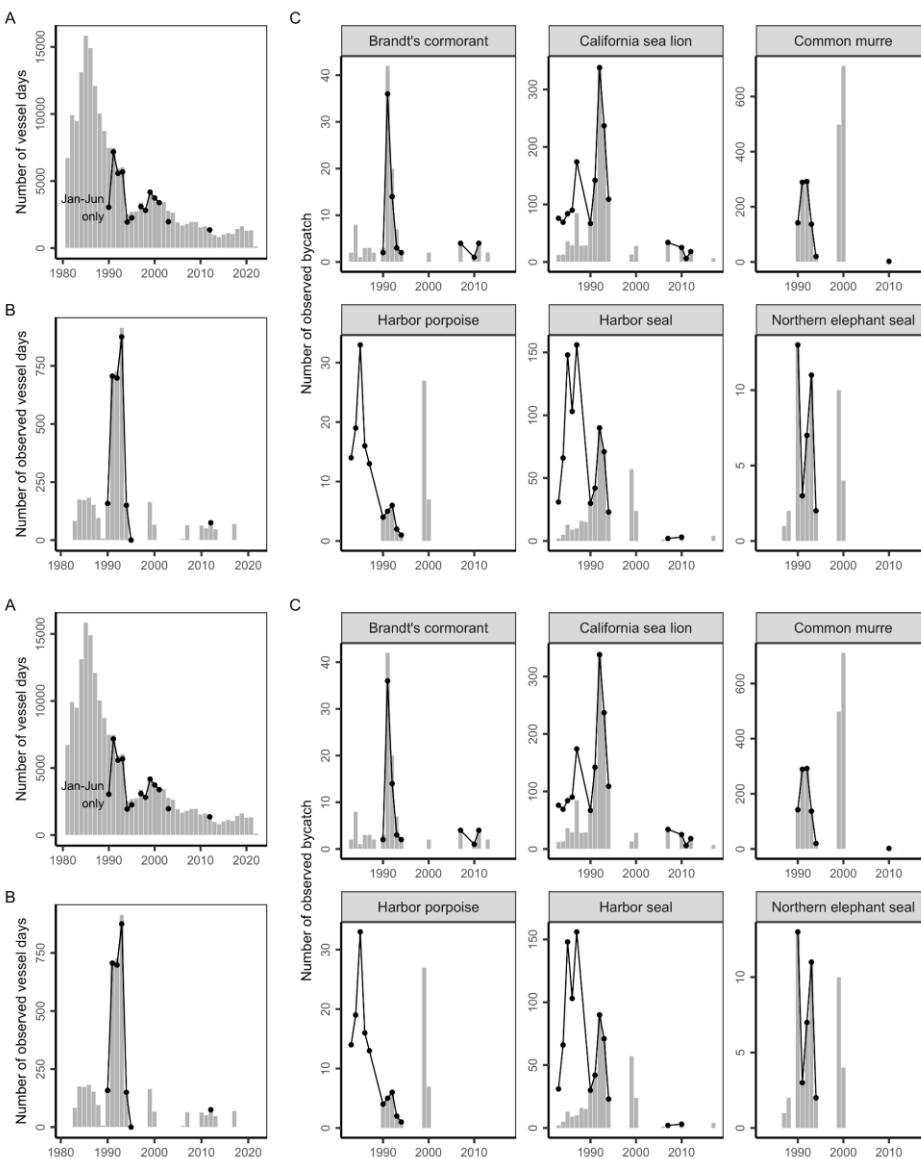
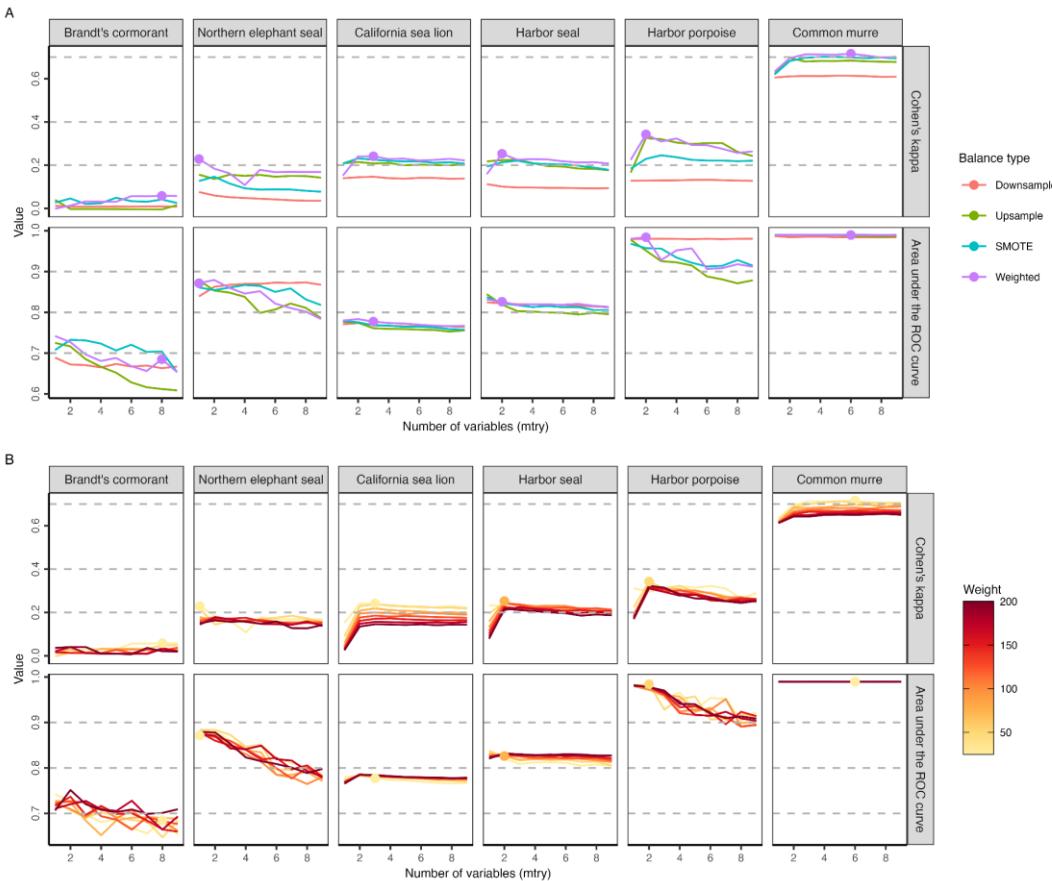


Fig. S12. A comparison of historical (A) fishing effort (number of vessel days); (B) observer coverage (number of observed vessel days); and (C) observer records (number of observed bycatch) derived in our

analysis (bars) and reported in historical studies (points and lines). These time series represent the key inputs into the ratio estimation analysis and help to explain the differences between the estimates of annual bycatch derived in our study as compared to historical studies (see Fig. S11). See Table 4S1 for additional details on historical studies.



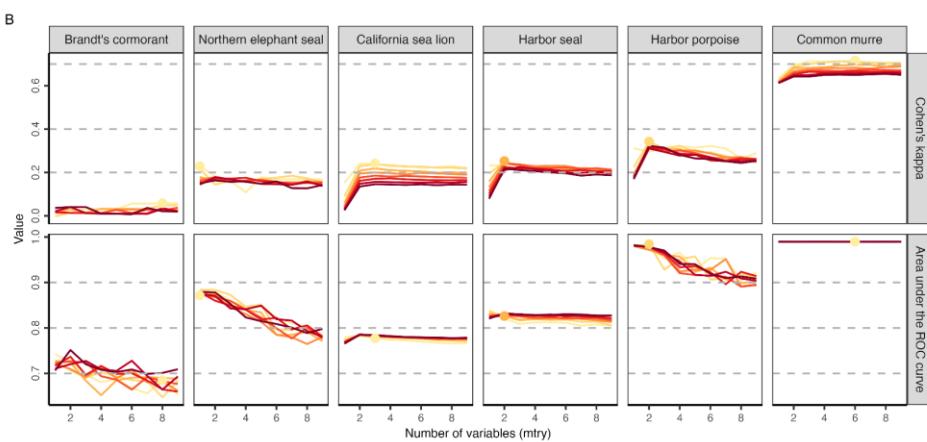
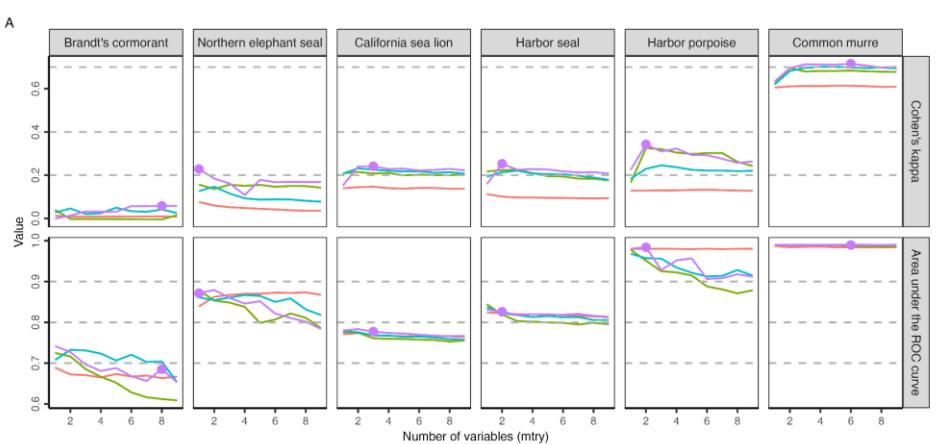
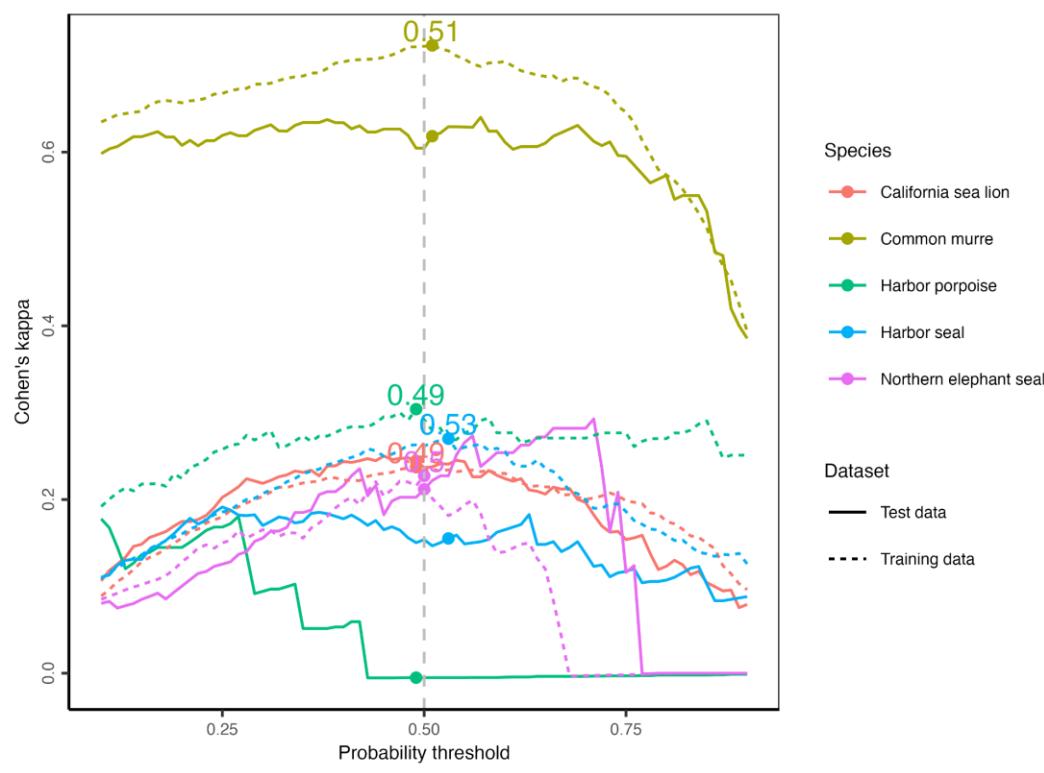
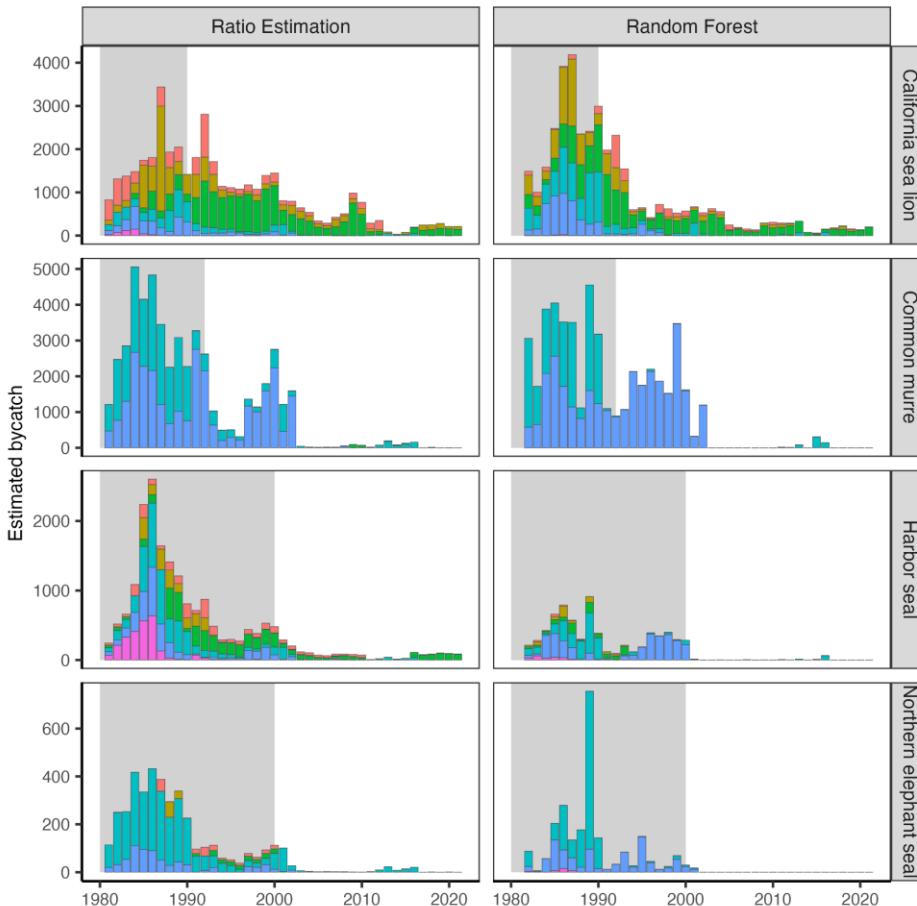


Fig. S13. A comparison of the model performance as measured through cross-validation on the training dataset between (A) balanced random forest modeling approaches and (B) weighted random forest modeling approaches for all candidate bycatch species (i.e., observer records > 50). We evaluated the model performance using Cohen's kappa and area under the receiver operator curve (ROC) and selected the model with the highest Cohen's kappa as the best performing model (labeled in the plot). We excluded Brandt's cormorant in the model prediction as their Cohen's kappa (<0.1) is too low to produce reliable predictions.

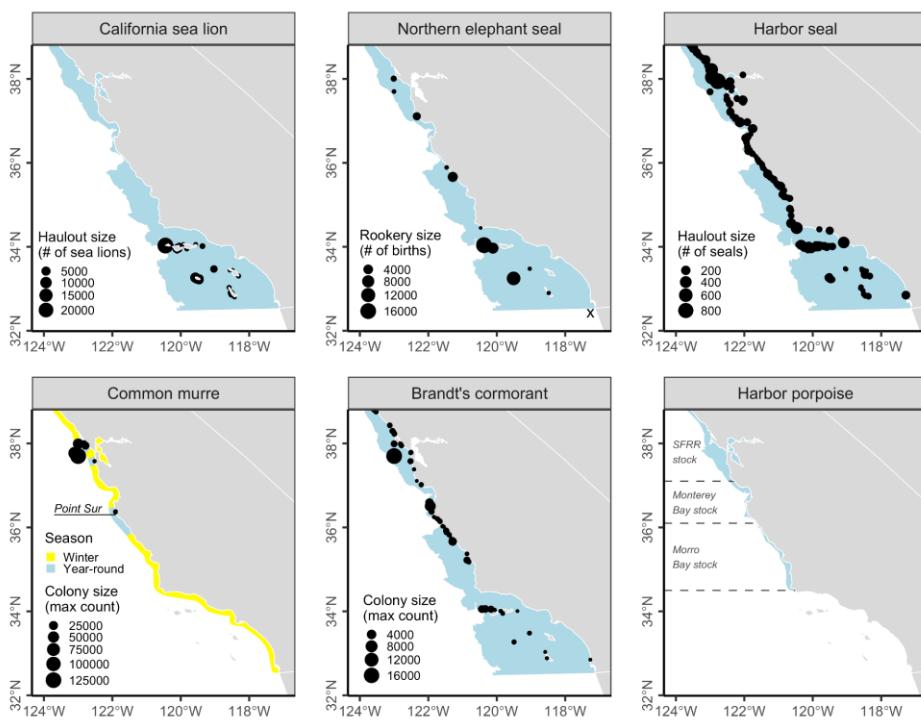


11559 Fig. S14.

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13 Southern California Ventura Monterey Bay
14 Channel Islands Morro Bay San Francisco
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4661 **Fig. S14.** A comparison of estimated bycatch numbers between ratio estimation and random forest
4662 stratified by regions (Fig. S8). The major differences between years are highlighted in gray.
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1565 Illustration of the methods used to select a probability threshold for classifying a logged set as having or
1566 not having bycatch. We selected the probability threshold that maximizes Cohen's kappa when applied to
1567 the training data as the optimal threshold (labeled in plot). We highlight the performance of this threshold
1568 when used on the independent test data to illustrate performance on out-of-sample data.
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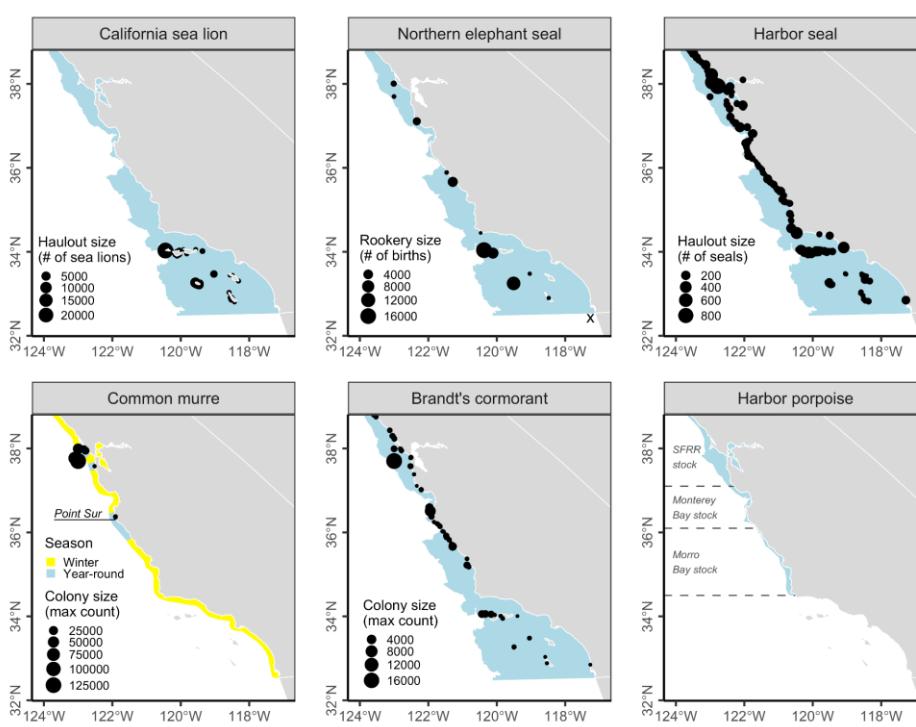


Fig. S15. Ranges of the six study species and information on haulout and colony size for selected species.

Blue colors indicate year-round ranges and yellow colors (common murre only) indicate winter ranges.

The range maps for all species except harbor porpoise are from the California Wildlife Habitat

Relationship System ([CDFW, 2021](#)). Harbor porpoises occur in waters less than 50

fathoms (92 meters) deep north of Point Conception ([Forney et al., 2014](#)). SFRR

indicates the San Francisco-Russian River harbor porpoise stock. Common murre and Brandt's cormorant

colony counts are from the California Seabird Colony Database ([CDFW, 2010](#)). There are

no historical breeding records for common murre south of Point Sur except for at the Prince Island colony

in Cuyler Harbor of San Miguel Island. Harbor seal haulout counts are from the CDFW Harbor Seal GIS

dataset ([CDFW, 2014](#)).

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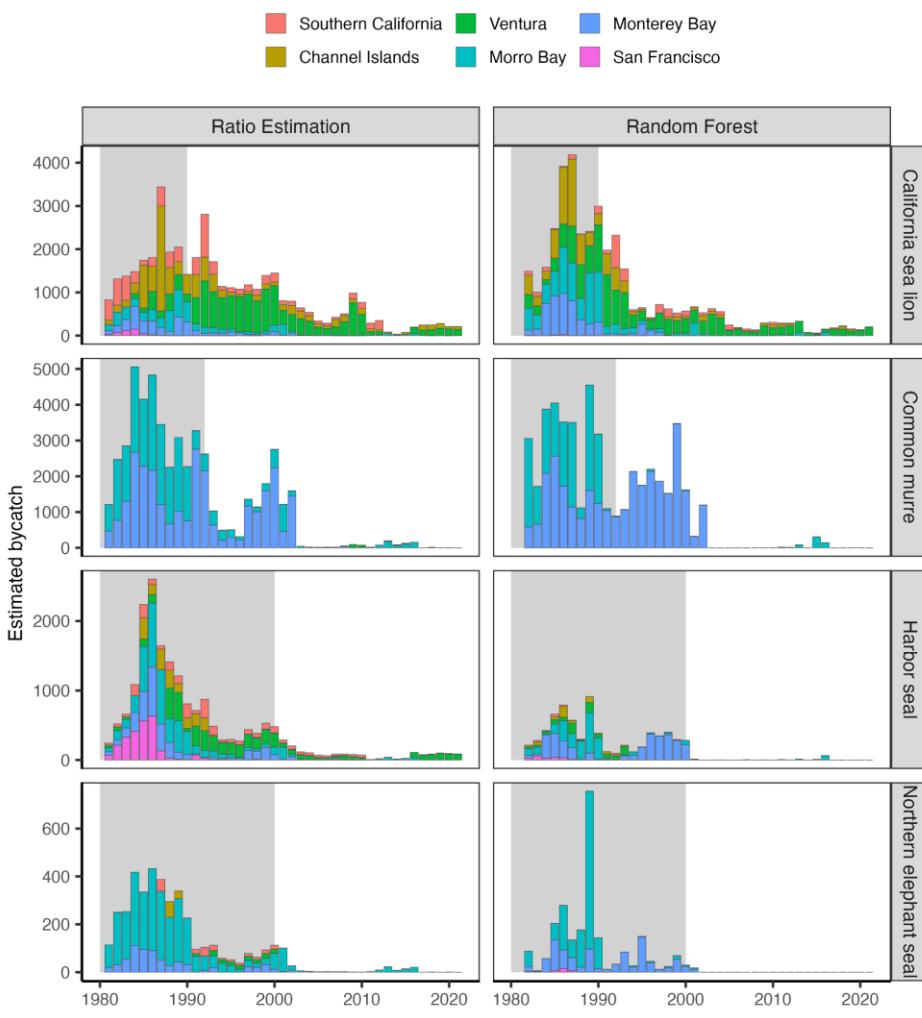


Fig. S16.

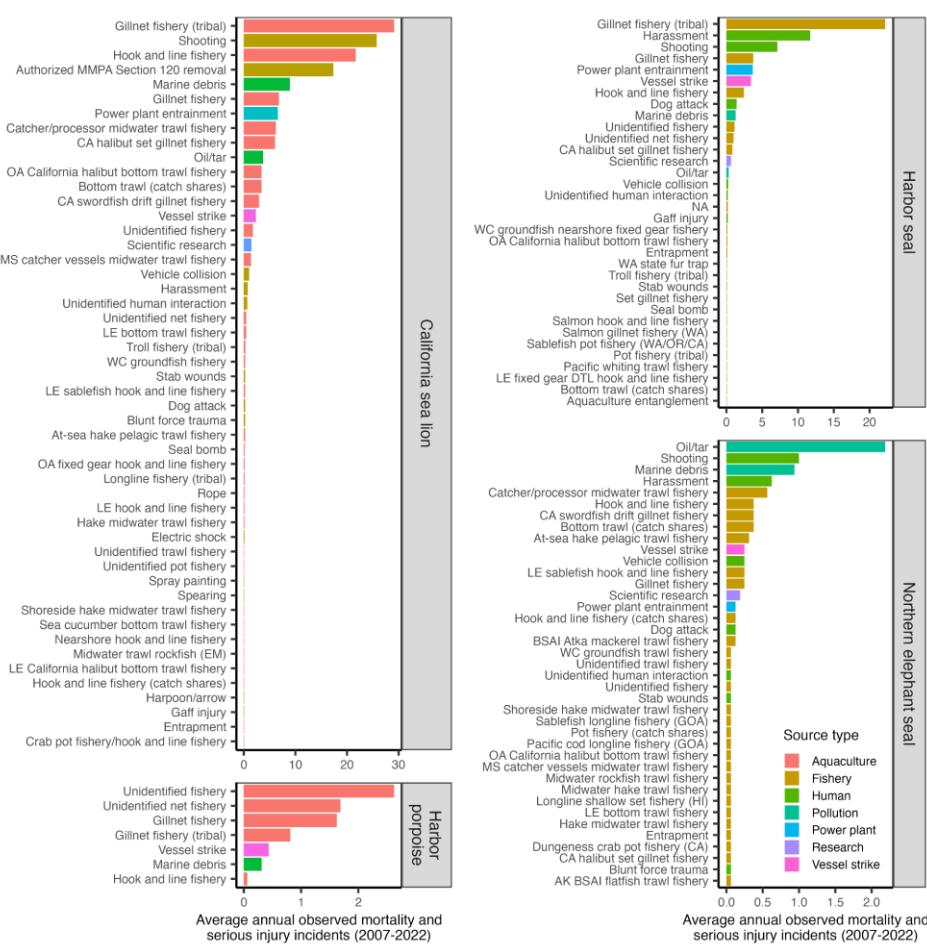


Fig. S16. A comparison of estimated bycatch numbers between ratio estimation and random forest stratified by regions (Fig. S8). The major differences between years are highlighted in gray.

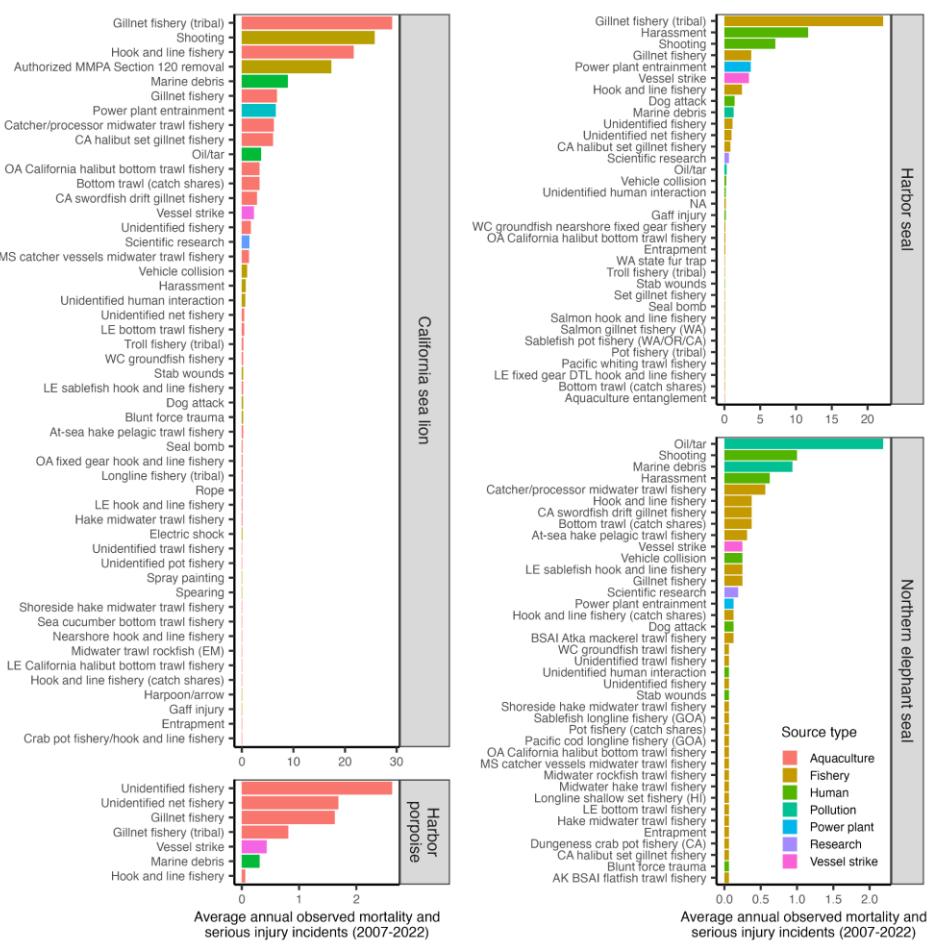


Fig. S17. Average annual observed mortality and serious injury incidents by source on the entire U.S. West Coast between 2007-2022 ([Carretta, 2023](#)), ([Carretta, 2023](#)).

West Coast between 2007-2022 ([Carretta, 2023](#)), ([Carretta, 2023](#)).

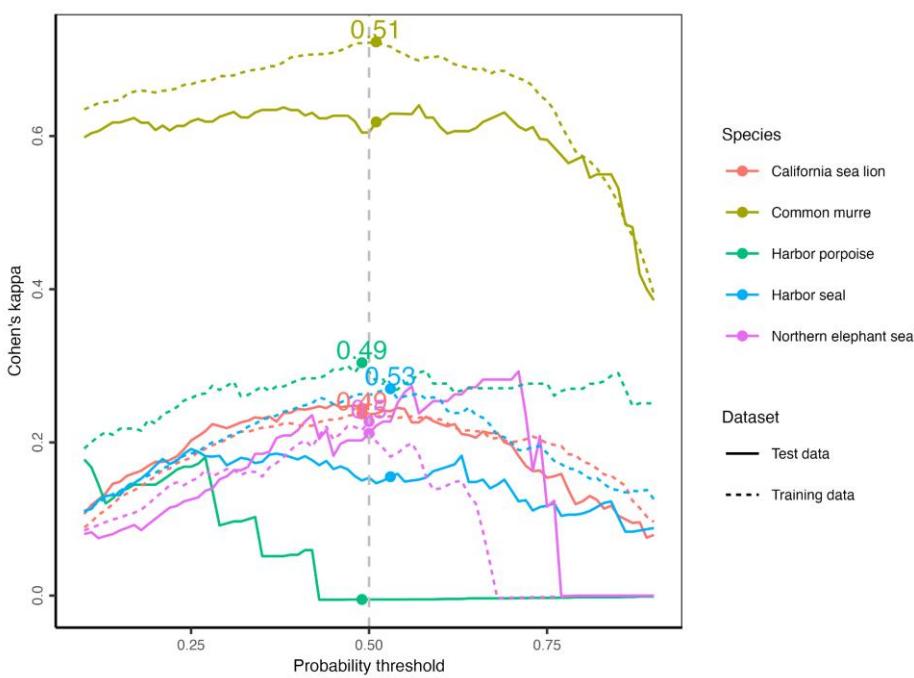


Fig. S17. Illustration of the methods used to select a probability threshold for classifying a logged set as having or not having bycatch. We selected the probability threshold that maximizes Cohen's kappa when applied to the training data as the optimal threshold (labeled in plot). We highlight the performance of this threshold when used on the independent test data to illustrate performance on out-of-sample data.

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56 1 **Estimates and drivers of protected species bycatch in the**
7 2 **California set gillnet fishery**
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1213 3 **Abstract**
1415 4 The identification of efficient management strategies that reduce protected species bycatch while also
16 5 minimizing impacts on fishing livelihoods is a global conservation challenge. Identifying such strategies
17 6 requires understanding levels of bycatch relative to management targets as well as the relationship
18 7 between bycatch risk and potential management levers. In this study, we use ratio estimators to
19 8 reconstruct bycatch of select marine mammal and seabird species in the California $\geq 3.5"$ set gillnet
20 9 fishery from 1981-2022 and random forest models to identify drivers and hotspots of bycatch risk. We
21 10 find that bycatch has dropped precipitously since the 1980s as a result of management, but at significant
22 11 costs to fisheries participation and revenues. Recent marine mammal bycatch ranges from 0.1% to 4.0%
23 12 of the potential biological removal and marine mammal populations are recovering. Spatial-temporal
24 13 drivers of bycatch risk were more important than fishing-related drivers of risk, suggesting that spatial-
25 14 temporal closures would be more effective than mesh size or soak time restrictions at limiting bycatch.
26
27 15 For each species, we identified 2-5 hotspots of elevated bycatch risk as candidates for temporary seasonal
28 16 closures. Bycatch risk for harbor seal (*Phoca vitulina*) and California sea lion (*Zalophus californianus*),
29 17 the species with the greatest bycatch risk, is especially high from April 1st to June 15th, suggesting that
30 18 hotspot closures during this 2.5-month time period could be particularly efficient. Our study also
31 19 highlights the value of competing multiple modeling approaches to identify methods that best predict rare
32 20 bycatch events.
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34 2135
36 22 **Keywords:** gillnet, bycatch, marine mammals, seabirds, area-based management
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5 23 **1. Introduction**

6 24 Bycatch, the accidental capture of non-target species in fisheries, presents a significant
7 25 conservation and economic challenge (Crowder and Murawski, 1998; Soykan et al., 2008). Bycatch of
8 26 large-bodied, slow-growing, low-productivity species such as marine mammals, sea turtles, and seabirds
9 27 (Crowder and Murawski, 1998; Read et al., 2006; Soykan et al., 2008) is of particularly high concern, as
10 28 the mortality of just a few individuals in these vulnerable populations, often recovering from historical
11 29 exploitation, can threaten population collapse and even extinction (Geijer and Read, 2013; Read et al.,
12 30 2006). As a result, many countries have established strict mandates to limit bycatch of vulnerable species,
13 31 which can result in fisheries closures and other severe restrictions (Crowder and Murawski, 1998; Senko
14 32 et al., 2014). These management disruptions can have serious social, cultural, and economic impacts on
15 33 fishing communities (Senko et al., 2014). Due to the negative ecological, economic, and social
16 34 consequences of fishery bycatch, bycatch avoidance is an important objective for global fishery
17 35 management. A sustainably managed fishery with low bycatch can not only provide ecological benefits,
18 36 but also social and economic benefits by providing a sustainable source of income, food, and nutrition.
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 To guide effective bycatch reduction policies, it is important to understand the magnitude of
historical and recent bycatch as well as the drivers of bycatch in a fishery. Estimates of total bycatch are
needed to determine whether bycatch exceeds management targets or is on pace to exceed targets in the
near future (Bjørge et al., 2013; Geijer and Read, 2013; Read et al., 2006). Historical bycatch estimates
offer insights into the effectiveness of past management interventions, which provide useful benchmarks
for adapting management in response to recent bycatch levels and trends. Understanding the drivers of
bycatch risk is critical to guiding effective and efficient management adaptations. For example,
determining whether bycatch is concentrated within specific areas or seasons can support the design of
time-area closures that prevent fishing when and where risk is high while maintaining fishing
opportunities elsewhere (Lewison et al., 2014; O’Keefe et al., 2023; Soykan et al., 2008). Similarly, gear,
soak time, or time of day restrictions can be used to curb bycatch if there are strong relationships between
bycatch risk and gear or other characteristics of fishing (O’Keefe et al., 2023). Without this information,
bycatch management must be precautionary to guarantee compliance with protected species legislation,
which could forego considerable fisheries yields.

 Because observer programs, which place trained scientists on fishing vessels to collect bycatch
data, are costly and rarely cover all fishing trips, various analytical approaches have been developed to
estimate unobserved bycatch and to evaluate drivers of bycatch risk. Ratio estimation, a design-based
approach that assumes that the rate of bycatch in observed fishing trips is proportional to the rate for all

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4 57 fishing trips, is one of the most common strategies (Cochran, 1977; Stock et al., 2019). The reliability of
5 58 this simple approach increases if there are sufficient data to support estimates within meaningful spatial or
6 59 temporal strata (e.g., regions, depth zones, seasons). However, ratio estimation can produce biased
7 60 estimates if other factors (e.g., gear type, soak time, time of day) influence bycatch rates (ICES, 2007), if
8 61 the observed trips are not representative of the unobserved trips, or if low sample sizes lead to spuriously
9 62 low or high bycatch rates within a stratum (Martin et al., 2005; McCracken, 2004; Ortiz and Arocha,
10 63 2004; Rochet and Trenkel, 2005). Model-based approaches, which use either statistical (e.g., generalized
11 64 linear models, generalized additive models) or machine learning (e.g., random forests, boosted regression
12 65 trees) models to estimate bycatch, can overcome many of these limitations by incorporating a wider suite
13 66 of covariates and by allowing for non-linear relationships and are generally thought to produce better
14 67 bycatch estimates (Stock et al., 2019). Additionally, model-based approaches can support management by
15 68 identifying drivers of bycatch risk and by predicting detailed hotspots of risk (Long et al., 2024; Lopez et
16 69 al., 2024; Stock et al., 2019).
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30 71 The California set gillnet fishery would benefit from updated bycatch estimation due to concerns
31 72 about the fishery's impact on protected marine mammals, which have led some conservation
32 73 organizations to call for the fishery's closure (Birch et al., 2023; Birch and Shester, 2023). The fishery
33 74 occurs in southern California and targets California halibut (*Paralichthys californicus*), white seabass
34 75 (*Atractoscion nobilis*), and Pacific angel shark (*Squatina californica*), among other species. It is currently
35 76 listed as a Category II fishery under the U.S. Marine Mammal Protection Act (MMPA), indicating that it
36 77 presents a medium bycatch threat to protected marine species (NOAA, 2024). Bycatch of marine mammal
37 78 and seabird species, including harbor porpoise (*Phocoena phocoena*), southern sea otter (*Enhydra lutris*
38 79 *nereis*), and common murre (*Uria aalge*) was high during the 1980s and 1990s, prompting large-scale
39 80 management interventions (Forney et al., 2001; Julian and Beeson, 1998). The fishery has also impacted
40 81 pinniped species such as California sea lion (*Zalophus californianus*), harbor seal (*Phoca vitulina*), and
41 82 northern elephant seal (*Mirounga angustirostris*). Total bycatch in the fishery has not been estimated
42 83 since 2012 (Carretta et al., 2014) and some conservation groups are concerned that bycatch remains an
43 84 issue (Birch et al., 2023; Birch and Shester, 2023).
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56 86 A number of management actions have been taken to reduce bycatch in the California set gillnet
57 87 fishery. During the 1980s, high bycatch of southern sea otters and common murres in central California
58 88 (Barlow et al., 1994) led to a depth restriction that closed fishing inside of 40 fathoms (73 m) in 1987
59 89 (Forney et al., 2001). This restriction shut down the fishery in the San Francisco area, effectively pushing
60 90 it south of Pigeon Point and into Monterey Bay and Morro Bay (**Fig. 1A-2**). In 1990, the state adopted
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4 91 Proposition 132 (CA Secretary of State, 1990), which went into effect in 1994 and banned the fishery in
5 mainland state waters (0-3 nautical miles) and in waters within 1 nautical mile or 70 fathoms of depth,
6 whichever is less restrictive, around the Channel Islands to further reduce bycatch of protected species
7 (FGC §8610.1-8610.16). In 2002, the state expanded the existing depth restriction, closing fishing inside
8 of 60 fathoms (110 m) to avoid the harbor porpoise population in Central California (14 CCR §104.1).
9 This effectively closed the fishery in Monterey Bay and Morro Bay (**Fig. 1A-4**). Currently, the fishery
10 only operates in southern California (south of Point Arguello) outside 3 nautical miles from the mainland
11 and outside 1 nautical mile or shallower than 70 fathoms (whichever is less) from the Channel Islands.
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14 100 Although these regulations are believed to have reduced bycatch in the set gillnet fishery
15 (Carretta et al., 2014), they have also greatly reduced fishery participation and revenues. The
16 implementation of the 40 fathom depth restriction in 1987 triggered a precipitous decline in participation
17 from ~400 vessels in 1987 to ~100 vessels in 1994. Since then, participation has continued to decline,
18 with ~40 vessels active in 2022, and the vast majority (>90%) of landings coming from just 13 vessels
19 (CDFW, 2023) (**Fig. 1B**). Fishing effort has similarly decreased from an estimated ~15,000 fishing trips
20 in 1987 to 1,000 trips in 2022 (**Fig. 1C**). This reduction in effort has significantly reduced bycatch levels
21 (Carretta et al., 2014) but at large costs to fishery revenues. Fleetwide revenues decreased from US\$15
22 million in 1987 to US\$1 million in 2022 (**Fig. 1C**; both values in 2022 dollars). Despite declining fishing
23 effort and bycatch, conservation groups are lobbying for additional restrictions, including permanent
24 closure, to further avoid bycatch (Birch et al., 2023; Birch and Shester, 2023). There is thus great need for
25 scientific guidance on management regulations that are likely to provide conservation benefits while also
26 avoiding unnecessary burdens on the fishing industry.
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In this study, we reconstruct the bycatch of select marine mammals and seabirds in the California
set gillnet fishery from 1981-2022 and identify drivers of bycatch that could be used to refine
management to more efficiently reduce bycatch, where efficient management achieves conservation
objectives while minimizing impacts on fishing opportunities. We focus on six protected species
encountered in the set gillnet fishery: California sea lion, harbor seal, northern elephant seal, harbor
porpoise, common murre, and Brandt's cormorant (*Phalacrocorax penicillatus*). Southern sea otter,
despite being one of the original species of management concern, is not included in this analysis due to
data limitations (**Fig. 2A**). We use ratio estimation methods to reconstruct historical bycatch levels and
compare recent bycatch levels to management targets. These methods, which have been used to estimate
bycatch in the fishery at various points in the past (**Table S1**), provide a complete time series of bycatch
estimates using methods approved for stock assessment and management. We then use random forest

models to evaluate drivers of bycatch risk and to make predictions of spatial bycatch risk for four species of concern; Brandt's cormorant and harbor porpoise were excluded from this portion of the analysis due to poor model performance (**Table 1**). Based on these results, we make recommendations for how management could more efficiently manage bycatch risk through measures such as seasonal or spatial closures, depth restrictions, or gear restrictions.

2. Methods

2.1 Overview

We used a design-based ratio estimation approach to estimate bycatch of select marine mammal and seabird species in the California set gillnet fishery from 1981-2022. Briefly, the ratio estimation approach estimates total bycatch by applying the bycatch rates (i.e., bycatch per trip) on the sample of fishing trips with trained observers (*section 2.4.1*) to all fishing trips, which are documented in the logbooks maintained by all fishing vessels (*section 2.4.2*). Although model-based approaches, including the random forest approach used in the second portion of our analysis, are generally thought to produce better estimates of bycatch than design-based approaches (Stock et al., 2019)(see *section 2.6.4* and the supplemental information for more details)(Stock et al., 2019)evaluate drivers of bycatch and map hotspots of bycatch risk for four of the six evaluated species. We used a random forest approach, a machine learning method increasingly used in ecology, because of its high predictive skill for rare events, ability to model non-linear relationships, and insensitivity to collinear or unimportant predictor variables relative to classical regression techniques such as generalized linear models or generalized additive models (Cutler et al., 2007; Prasad et al., 2006). All analysis was done in R (R Core Team, 2024) and all code and non-confidential data are available on GitHub here: [add link post double-blind peer review].

2.2 Study area

Our study area spans southern and central California, U.S.A. from the U.S.-Mexico border (32°N) to Point Reyes (38°N), which is just north of San Francisco Bay (**Fig. 1A**). The study area falls within the California Current Ecosystem, a highly productive eastern boundary upwelling system spanning from the southern tip of the Baja Peninsula, Mexico (28°N) to the U.S.-Canada border (48°N). Seasonal upwelling of cold, nutrient-rich water fuels populations of krill, squid, sardines, and other low trophic species that are fed upon by larger fishes, sea turtles, seabirds, and marine mammals. The ecosystem also supports important commercial and recreational fisheries and other human uses that benefit tens of millions of

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4 154 people living along the U.S. West Coast. We assessed bycatch within seven regions of the study area,
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6 155 which are shown in **Fig. 1A** and explained in detail in *section 2.5*.
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9 156 2.3 The fishery
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11 157 We defined the fishery using the definition in the MMPA List of Fisheries (NOAA, 2023): the
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13 158 ≥3.5 inch mesh set gillnet fishery targeting California halibut, white seabass, Pacific angel shark, and
14 other species. Although this definition deviates from historical studies, which frequently focused on the
15 portion of the fishery using mesh sizes larger than 8.0 or 8.5 inches (**Table S1**), the MMPA definition
16 provides the legal basis for bycatch management and is more consistent with historical regulations.
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18 161 Specifically, a minimum mesh size of 3.5 inches was set for white seabass in 1941, though it was
19 increased to 6.0 inches in 1988 (FGC §8623(d)). Since 1989, California halibut and Pacific angel shark
20 have been targeted using a minimum mesh size of 8.5 inches (FGC §8625(a)). The set gillnet fishery
21 principally excluded by this definition is that for Pacific herring (*Clupea pallasii*), which occurs in
22 California's four largest herring spawning areas — San Francisco Bay, Tomales Bay, Humboldt Bay, and
23 Crescent City Harbor — using mesh sizes of 2.0 to 2.5 inches (CDFW, 2019).
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31 168 2.4 Data
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34 169 Our analysis relies on two fisheries-dependent datasets: logbook data and observer data. All
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36 170 gillnet vessels are required to submit logbooks documenting when, where, and how they fished and how
37 much catch was retained. Thus, logbooks characterize all fishing trips. However, because logbooks are
38 self-reported and reported discards are unverifiable, logbooks likely underreport discarded bycatch,
39 especially the bycatch of protected species. As a result, information from observer programs, which place
40 trained observers on a sample of fishing trips (0-16.7% in this fishery; **Fig. 2B; Table S2**), are required to
41 inform estimates of bycatch for the unobserved trips recorded in vessel logbooks, which constitute the
42 majority of fishing effort (83.3-100% in this fishery; **Fig. 2B; Table S2**). Thus, the estimation of total
43 bycatch through ratio estimation depends on both the observer and logbook data. In contrast, the
44 evaluation of bycatch drivers and hotspots with the random forest models uses only the observer data, as
45 these are the only data to accurately record protected species bycatch when it occurs. These datasets are
46 described in detail below.
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55 181 2.4.1 Observer data
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57 182 We received observer data from 1983 to 2017 from the California Department of Fish and
58 Wildlife (CDFW). There was observer coverage in the California set gillnet fishery from 1983-1995
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4 184 (coastwide), 1999-2000 (Monterey Bay area only), 2010-2013 (south of Point Conception only), and in
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6 185 2017 (south of Point Conception only) (**Fig. 2**). The observer program was run by CDFW from 1983-
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8 186 1989 and by the National Oceanic and Atmospheric Administration (NOAA) from 1990 onwards. The
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10 187 percentage of annual fishing trips with onboard observers has varied over time, ranging from 0.3% of
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12 188 trips in 2006 to 16.7% of trips in 1993 (**Fig. 2B; Table S2**). Observers collected information on the
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14 189 amount and fate of catch (kept, discarded, or damaged), the length composition of the catch, the location
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16 190 and time of the catch, and characteristics of the gear used to target the catch (**Fig. S2**). We developed a
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18 191 series of simple assumptions to impute missing values for a few key variables (GPS coordinates, fishing
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20 192 depth, soak hour, mesh size) used to describe gillnet sets documented in the observer data (**Fig. S3**; see
supplemental methods for details).

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23 195 Although historical reports document low levels of observer coverage in Morro Bay, Monterey
24 Bay, and San Francisco in the 1980s (**Table S1**), the data that we received from CDFW excluded most of
25 these observations. We recovered a small portion of the missing raw data – observations from Monterey
26 Bay from 1987-1989 (**Fig. 2**) – from original CDFW data sheets that were given to a colleague at the
27 Southwest Fisheries Science Center during the late 1990s for a reanalysis of historical bycatch rates in
28 that region (Forney et al., 2001). We extracted summaries of set-level bycatch rates from historical reports
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30 199 (**Table S1**) for years and regions missing raw data to support the ratio estimation analysis (**Table S3**). We
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32 200 converted set-level bycatch rates to trip-level bycatch rates assuming an average of 3 sets per trip (**Table**
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34 201 **S3**), as indicated by the observer data (**Fig. S4**). **Fig. 2C** illustrates the coverage of the available,
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36 202 recovered, and lost observer data.

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41 206 For each species of marine mammal, seabird, and sea turtle documented in the observer data, we
42 calculated the total number of captures observed in the California set gillnet fishery (**Fig. 2**). Throughout
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44 207 this analysis, we focus on the six species with more than 50 observations: common murre (2,381),
45 California sea lion (1,372), harbor seal (519), Brandt's cormorant (118), northern elephant seal (78), and
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47 209 harbor porpoise (97). Unfortunately, this excluded southern sea otter, which was a species of significant
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49 210 conservation concern in the 1980s, but whose bycatch was only documented in the lost observer data.
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52 212 2.4.2 Logbook data
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55 213 We received logbook data from the commercial gillnet fishery from 1981 to 2022 from CDFW
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57 214 (**Fig. 1**). All California gillnet vessels are required to submit logbooks documenting all of their fishing
58 trips; as a result, these logbooks represent all fishing effort associated with the fishery. These data
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60 216 describe vessel information (vessel name, unique identifier, permit number); when (date), where

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4 217 (statistical reporting block; **Fig. 1A**), and how long (hours) a vessel fished; what fish it targeted; what
5 type of gear it used (drift or set gillnet) and characteristics of this gear (length, mesh size, fishing depth;
6 **Fig. S2**); what species it caught; and the amount (number and/or weight) and fate of this catch (kept,
7 released, or lost, including the identity of predators preying on released fish). We attempted to identify
8 individual fishing sets within the logbook data as the unique combination of vessel administration
9 information (vessel name, either vessel id or boat number, permit number), where, when, and how long a
10 vessel fished (block id, date, and fishing hours), and characteristics of the gear (net length, mesh size, and
11 fishing depth). This analysis revealed an average of 1 set per trip, which is inconsistent with the 3 sets per
12 trip documented in the more accurate observer data. We term these unique identifiers “pseudo-sets” and
13 view them as roughly equivalent to a fishing trip (**Fig. S4**). We developed a series of simple assumptions
14 to impute missing or unrealistic values for a few key variables (fishing depth, soak hour, mesh size) used
15 to describe gillnet pseudo-sets documented in the logbook data (**Fig. S5 & S6**; see supplemental methods
16 for details).

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27 230 2.4.3 Sea surface temperature
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29 231 Because sea surface temperature (SST) is a common driver of the distributions of both target and
30 bycatch species (Hazen et al., 2018), we used SST as an environmental covariate in the random forest
31 models described below. We derived the SST associated with each set documented in the observer and
32 trip documented in the logbook data using the NOAA 1/4° Daily Optimum Interpolation Sea Surface
33 Temperature (OISST) dataset, which interpolates observations from different monitoring platforms (e.g.,
34 satellites, ships, buoys, and Argo floats) to provide a globally complete grid of SST from September 1,
35 39 1981 to present (Huang et al., 2021). For sets reported in the observer data, we extracted the SST at the
40 reported GPS location on the reported day of fishing. For trips reported in logbooks, we calculated the
41 average SST in the reported block on the reported day of fishing.
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46 240 2.5 Ratio estimation
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48 241 We estimated annual bycatch for each study species using ratio estimators. Ratio estimators
49 assume that the rate of bycatch in observed fishing trips is proportional to the rate of bycatch within all
50 fishing trips within a given stratum (Cochran, 1977). This assumption requires that the characteristics of
51 observed trips do not systematically differ from the characteristics of all trips, which was confirmed by a
52 two-sided Kolmogorov-Smirnov test for six key traits (i.e., day of year, depth, latitude, mesh size,
53 distance from shore, soak time) (**Fig. S7**). We used trips rather than sets as the sampling unit given the
54 inability to identify unique sets in the logbook data (**Fig. S4B**). This is valid because the number of gillnet
55 sets per fishing trip (median: 3 sets/trip; interquartile range: 2-4 sets/trip) has been consistent through time
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4 249 (**Fig. S4A**). Under this approach, the bycatch rate for species s in stratum i ($r_{s,i}$) – where, in this case,
5 strata are defined by years and regions (see next paragraph) – is thus calculated as:
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$$r_{s,i} = \frac{k_{s,i}}{d_{s,i}}$$

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14 254 Where $k_{s,i}$ is the total number of individuals of species s captured in observed trips occurring in strata i
15 and $d_{s,i}$ is the total number of observed trips occurring in strata i . The total estimate of bycatch of species s
16 in strata i ($m_{s,i}$) is then calculated by multiplying the bycatch rate ($r_{s,i}$) by the total number of trips to have
17 occurred in the strata (D_i):
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25 261 Where the total number of trips (D_i) is derived from the logbook data.
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29 263 We calculated annual bycatch estimates using a seven-region stratification scheme (**Figs. 1A, S8**).
30 264 This stratification scheme combines the scheme by Diamond and Hanan (1986) and Julian (1993) for
31 areas north and south of Point Conception, respectively. Although early efforts to estimate bycatch in the
32 California set gillnet fishery often stratified estimates by region and season (**Table S1**), later efforts found
33 that observer coverage was often too limited to employ complex temporal stratification and that estimates
34 between temporally stratified and unstratified approaches were generally similar (**Table S1**). Stratum-
35 specific bycatch rates for years without observer coverage in the stratum are borrowed from the closest
36 year (forwards or backwards) with observer coverage in the stratum (**Fig. 2C & S9**), as has been the
37 practice in previous studies. We collated annual bycatch estimates from past studies (**Table S1**) for
38 comparison with our updated estimates (**Figs. S10-S12**).
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48 274 Although there are methods for estimating the uncertainty of bycatch estimates generated through
49 ratio estimation (Julian and Beeson, 1998), we were unable to implement these methods because they rely
50 on bootstrap procedures that sample from the bycatch rates of observed trips. Because these procedures
51 require raw observer data, we cannot use them for (1) years where summary values from historical reports
52 are used because the raw data have been lost or (2) years without observer data from within one of the
53 fished strata. As a result, only 6 of the 42 evaluated years had the data required to estimate uncertainty:
54 279 2006, 2007, 2010, 2011, 2012, and 2017 (**Fig. 1AC; Fig. 2BC**). An exploration of the uncertainty
55 estimates generated in historical reports with access to the lost data (**Table S1**) suggests that the median
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$$m_{s,i} = D_i * r_{s,i}$$

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4 282 coefficient of variation for estimates of annual bycatch estimates ranges from a low of 0.14 for harbor seal
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6 283 to a high 0.47 for harbor porpoise (**Fig. S10B**).
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9 285 We evaluated the sustainability of recent estimated marine mammal bycatch by comparing it to
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11 286 the potential biological removal (PBR) for each stock, which is defined under the MMPA as the
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13 287 maximum number of animals, not including natural mortalities, that may be removed from a marine
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15 288 mammal stock while allowing that stock to reach or maintain its maximum sustainable population. We
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17 289 extracted each PBR from its most recent stock assessment (**Fig. 7**) and compared it to the average
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19 290 estimated catch over the last 10 years (2013-2022). A fishery is managed based on its classification into
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21 291 one of three categories: Category 1 fisheries cause annual mortality and serious injury (M/SI) greater than
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23 292 50% of the PBR, Category II fisheries cause annual M/SI between 1 and 50% of the PBR, and Category
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25 293 III fisheries cause annual M/SI less than 1% of the PBR. A fishery is considered to be approaching the
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27 294 MMPA's "zero mortality rate goal" (ZMRG) when annual M/SI is below 10% of the PBR.
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31 295 2.6 Random forest modeling
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34 296 2.6.1 Model training
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300 We used random forest classification models trained on the observer data to identify drivers of
301 bycatch risk for each of the six study species. We used a random forest approach, a machine learning
302 method that ensembles predictions from hundreds of decision trees, rather than a classical regression
303 method (e.g., generalized linear or additive models) because of their comparatively high predictive skill
304 for rare events, ability to model non-linear relationships, and insensitivity to collinear or unimportant
305 predictor variables (Cutler et al., 2007; Prasad et al., 2006). We considered nine attributes of fishing as
306 potential drivers of bycatch risk: haul depth (fathoms), mesh size (inches), soak time (hours), latitude
307 ($^{\circ}$ N), longitude ($^{\circ}$ W), distance from shore (km), Julian day, sea surface temperature ($^{\circ}$ C), and whether the
308 fishing occurs near an island (i.e., within 10 km of island coast). These attributes were selected based on
309 their demonstrated relationship to bycatch risk in other papers (e.g., (Bettoli and Scholten, 2006; Bjørge et
310 al., 2013; Kroetz et al., 2020)) and their availability in the observer data or their ability to be derived
311 through remote sensing (i.e., distance from shore, temperature, island area). They represent a range of
312 spatial (latitude, longitude, distance from shore, depth, island area), temporal (Julian day), environmental
313 (temperature), and fishing-related (soak time, mesh size) attributes. For each species, we classified an
observed set as having (1) or not having (0) bycatch, and trained a classification model assuming a
Bernoulli distribution in the response variable.

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4 314 Because bycatch of protected species is rare, observed fishing sets show strong class imbalance
5 towards sets without bycatch compared to sets with bycatch. To illustrate, the percent of observed sets
6 with bycatch is as follows: California sea lion (1.02%), common murre (0.43%), harbor seal (0.43%),
7 Brandt's cormorant (0.09%), northern elephant seal (0.076%), and harbor porpoise (0.074%). Therefore,
8 without a proper sample balancing method, predictions are likely to be biased towards the majority class
9 (sets without bycatch), leading to an underestimation of bycatch risk. For this reason, we considered four
10 approaches for accounting for class imbalance resulting from bycatch rarity.
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18 322 The first three approaches employ different sample balancing methods: (1) downsampling, (2)
19 upsampling, and (3) synthetic minority over-sampling (SMOTE), which uses a mixture of down and
20 upsampling (More and Rana, 2017). The downsampling approach randomly removes observations of the
21 majority group (sets without bycatch) to obtain equal representation of the majority and minority (sets
22 with bycatch) group. The upsampling approach randomly samples observations from the minority group
23 with replacement to obtain equal representation with the majority group. The synthetic minority over-
24 sampling (SMOTE) approach both up-samples the minority group and down-samples the majority group.
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26 327 It up-samples the minority class by synthesizing new cases from its nearest five neighbors and down-
27 samples the majority class by randomly drawing samples from that group. We created each balanced
28 dataset using the *themis* package in R (Hvitfeldt, 2023) and fit random forest models to these datasets
29 using the *randomForests* package in R (Liaw and Wiener, 2002).
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38 334 The fourth approach employs weighted random forests, which use observation weighting rather
39 than sample balancing to elevate the importance of the minority class. In this “cost-sensitive” learning
40 approach (More and Rana, 2017), higher weights are assigned to minority observations so that the model
41 receives a higher penalty for misclassifying these observations, helping to reduce bias towards the
42 majority class. We evaluated multiple weighting schemes to optimize the predictive skill of this approach.
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44 338 Specifically, we assigned majority observations (sets without bycatch) a weight of 1 and assigned
45 minority observations (sets with bycatch) weights of 25 to 200 in increments of 25. Thus, a total of eight
46 candidate weighted random forest models were evaluated as described below. We fit the weighted
47 random forest model using the *ranger* package in R (Wright and Ziegler, 2017).
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55 344 We trained each of the eleven candidate models (three balanced random forest models, eight
56 weighted random forest models) on 80% of the observer data, withholding the remaining 20% for model
57 testing. In training the models, we performed a grid search to identify the “mtry” hyperparameter – the
58 number of variables to randomly sample as candidates at each node split – that maximizes Cohen’s kappa
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under 10-fold cross validation (**Fig. S13**). While accuracy measures the proportion of correctly classified categorizations, Cohen's kappa measures the proportion of correct classifications while accounting for the probability of being correct by chance and is a better measure of predictive skill, especially for imbalanced datasets (Cohen, 1968). Although there are no definitive rules for interpreting Cohen's kappa, general guidelines suggest that values above 0.7 are "excellent", 0.4-0.7 are "good", 0.2-0.4 are "fair", and below 0.2 are "poor" (Fleiss et al., 2013; Landis and Koch, 1977). We identified the best fitting model as the model generating the highest Cohen's kappa on the training data. We applied this model to the test dataset for an independent evaluation of its predictive power. We only evaluated four species (California sea lion, common murre, harbor seal and northern elephant seal) whose best models exhibited "fair" or better performance in their training dataset and close to "fair" performance on the test dataset for the rest of the analysis (**Table 1**).

2.6.2 Model evaluation

We evaluated the drivers of bycatch risk for each species by inspecting the variable importance and the marginal effects of each variable as estimated in the best fitting model. Variable importance was evaluated as the total decrease in node impurities from splitting on the variable averaged over all trees. The impurity measure is corrected when building the model to reduce its bias towards continuous variables (Nembrini et al., 2018). Marginal effects measure the impact of the changes in one variable on the response variable while all other variables are held constant. The marginal effects plots provide the scientific basis for our discussions of management regulations that could effectively and efficiently reduce bycatch risk.

2.6.3 Mapping spatial bycatch risk

To generate maps of spatial bycatch risk, we used the best fitting model to predict risk to a 0.02° grid spanning southern California ($32\text{-}35^{\circ}\text{N}$ and $117\text{-}121^{\circ}\text{W}$), the only area where the fishery can operate under current regulations. Conceptually, our metric of spatial bycatch risk represents the probability of bycatch at a given location under recent average conditions. We derived this metric by first predicting the probability of bycatch on every calendar day (Julian day 1 to 365) for each grid cell. The nine input variables for making these predictions were derived as follows: (1) *latitude*, (2) *longitude*, (3) *distance from shore*, (4) *depth*, and (5) *island proximity* (i.e., whether the location was within 10 km of an island) were derived based on the centroid of the grid cell; (6) *soak time* was set to 24 hours, the logbook mode; (7) *mesh size* was set to 8.5 inches, the logbook mode; (8) *Julian day* was the input day; (9) *sea surface temperature* (SST) was the average SST for that day and grid cell from 2010-2022. We then calculated the weighted average of the daily risk weighting by the amount of logbook entries (a metric of fishing

effort) occurring on that Julian day from 2010-2022. We cropped the predictions to areas within 20 km of shore as this is the farthest offshore that the fishery has operated (**Fig. S7**). We delineated bycatch hotspots as areas falling within the 95th (California sea lion, harbor seal) or 99th (common murre, northern elephant seal) percentile contour of spatial risk for each species. A higher threshold was used for common murre and northern elephant seal because they exhibit a large number of very low risk cells.

2.6.4 Bycatch estimation

We explored using the random forests models to estimate annual bycatch (1981-2022) in the fishery but found them to be unsuitable for this specific case study. Although model-based approaches generally perform better than design-based approaches at estimating bycatch (reference), we found that our random forest models underpredicted bycatch risk in the 1980s and 1990s relative to the ratio estimator (**Fig. S14**). This is most likely because of the loss of observer data from the northern strata during this time period, fishery largely operated in the northern portion of the region. See the supplemental information for details on the estimation of bycatch using the random forest models.

3. Results

3.1 Ratio estimation

In general, estimated bycatch peaked in the mid-1980s, steadily declined following the 40-fathom depth restriction implemented in 1987, with a temporary increase in the late-1990s followed by continued decline (**Fig. 3**). Bycatch peaked, in order of decreasing magnitude, at 5,059 common murre in 1984, 3,437 California sea lion in 1987, 2,605 harbor seal in 1986, 560 harbor porpoise in 1985, 453 Brandt's cormorant in 1991, and 432 northern elephant seal in 1986 (**Fig. 3**). This pattern reflects trends in fishing effort, which also declined after a peak in 1985, with a brief expansion in the late-1990s followed by continued decline (**Fig. 1C**). Estimated bycatch of harbor porpoise, northern elephant seal, and common murre declined especially sharply following the 2002 exclusion of fishing from waters shallower than 60 fathoms. Slight differences between our estimates of annual bycatch and those from historical studies (**Fig. S11**) are driven by a mixture of differences in our methods and input data (**Fig. S12**).

The sustainability of recent estimated marine mammal bycatch was evaluated as a percentage of the potential biological removal (PBR) of each stock. Based on this sustainability reference point, bycatch concerns, in order of decreasing threat, are as follows: harbor seal (65 individuals per year = 4.0% of a PBR of 1,641 individuals), harbor porpoise (1 individual per year = 1.5% of a PBR of 65 individuals in

the Morro Bay stock, where the bycatch took place), California sea lion (194 individuals per year = 1.4% of a PBR of 14,011 individuals), and northern elephant seal (7 individuals per year = 0.1% of a PBR of 5,122 individuals) (**Fig. 7**). The assessment that bycatch during the last 10 years poses the greatest risk to harbor seals is supported by the fact that the harbor seal stock size has been stable or declining in recent years while all of the other marine mammal populations have been undergoing sustained population growth (**Fig. 7**). The sustainability of estimated seabird bycatch is more difficult to evaluate given more limited population monitoring data (**Fig. 7**) and the lack of legally binding reference points for defining allowed incidental take. However, increasing Brandt's cormorant nests from 1980 to 2020 (**Fig. 7**) and steeply reduced bycatch of common murre (**Fig. 3**) suggests low risks posed to these species.

3.2 Random forest modeling

3.2.1 Model performance

The best-fitting model performed well for all species, except for Brandt's cormorant and harbor porpoise, which exhibited poor performance (Cohen's kappa less than 0.2) and were therefore excluded from further consideration. Weighted random forest models performed best for California sea lion, common murre, northern elephant seal, and harbor seal with case weights of 25, 25, 25, and 75, respectively (**Table 1; Fig. S13**). For common murre, Cohen's kappa was 0.71, indicating "good" performance, while for harbor seal, California sea lion, and northern elephant seal, Cohen's kappa was 0.25, 0.24, and 0.23, respectively, indicating "fair" performance. Model performance was positively correlated with the frequency of bycatch observations, i.e., species with more observed bycatch events produced models with greater skill (**Table 1**; $r^2 = 0.64$ for Cohen's kappa for training data). Cohen's kappa was positively correlated with the area under the receiver operator curve (AUC) ($r^2 = 0.68$ for the training dataset), indicating minimal tradeoffs in using this metric for model selection (**Table 1**).

3.2.2 Drivers of bycatch risk

The importance of the evaluated explanatory variables in determining bycatch risk varied by species but some general patterns emerged (**Fig. 4**). In general, spatial (latitude, longitude, depth, and distance from shore) and temporal (Julian day) variables were more influential than variables associated with the environment (sea surface temperature) or the fishing methodology (soak time, mesh size). Whether fishing occurred close to an island (a spatial variable) was the exception, as it was consistently the least important variable. Sea surface temperature, which is closely related to space and time, was generally more important than soak time and always more important than mesh size.

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4 441 The species exhibit a mixture of similar and dissimilar responses to the explanatory variables
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6 442 (**Fig. 5**). California sea lion and harbor seal exhibit similar responses in bycatch risk. Both species have
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8 443 higher bycatch risk in shallower depths in nearshore areas with a spike in risk occurring around 34°N
9 latitude, including some deeper offshore areas (**Fig. 6**). They also exhibited a pronounced increase in risk
10 during the spring, lasting approximately from Apr 1 (90th day of the year) to June 15 (166th day of the
11 year). They are infrequently caught in nets with mesh sizes smaller than 8.5 inches, though the use of
12 such nets is rare (**Fig. S5E**). Variability in bycatch risk for common murre and northern elephant seal is
13 most strongly determined by latitude and longitude (**Fig. 5**), with the only area of elevated risk in
14 contemporary fishing grounds occurring just north of Point Conception (**Fig. 6**). For all four species,
15 bycatch risk exhibits an asymptotic relationship with soak time, though the shapes of these relationships
16 differ by species. The species exhibit complex and variable relationships to temperature.
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24 452 3.2.3 Maps of bycatch risk
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26 453 The species exhibited different patterns of spatial bycatch risk. California sea lion bycatch risk is
27 predicted to be highest in four areas: (1) on the northern coasts of the northern Channel Islands, especially
28 on the northern coast of Santa Rosa; (2) a small nearshore area west of Santa Barbara; (3) the eastern
29 coast of Santa Cruz Island; and (4) the northwestern shores of Santa Catalina and San Clemente Islands;
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31 456 (**Fig. 6**). Harbor seal bycatch risk is predicted to be highest in four areas: (1) the sliver of nearshore area
32 stretching from Santa Barbara to Point Sal; (2) the eastern coasts of Santa Cruz Island; (3) a broad coastal
33 area near Point Mugu; and (4) the sliver of nearshore area stretching from Point Mugu to Point Vicente
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35 459 (**Fig. 6**). Common murre bycatch risk is predicted to be negligible throughout most of southern California
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37 460 (**Fig. 6**). It is only predicted to be high in a small patch near Point Sal and even there, the maximum risk
38 index is much lower than for the other evaluated species. Like common murre, northern elephant seal
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40 462 bycatch risk is also predicted to be negligible throughout most of southern California except in the region
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42 463 near Point Sal (**Fig.6**).
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48 465 4. Discussion
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51 466 Our study provides the first update to total estimates of protected species bycatch in the
52 California set gillnet fishery since 2012. We find that bycatch, once high and unsustainable for some
53 species (Forney et al., 2021, 2001), is now well below the “zero mortality rate goal” (ZMRG) of 10% of
54 the potential biological removal (50 C.F.R. § 229.2). Recent marine mammal bycatch estimates range
55 from 0.1-4.0% of their potential biological removals and common murre bycatch has been effectively
56 eliminated. All of the evaluated populations, including the once declining and heavily depleted Morro
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Our study provides the first update to total estimates of protected species bycatch in the California set gillnet fishery since 2012. We find that bycatch, once high and unsustainable for some species (Forney et al., 2021, 2001), is now well below the “zero mortality rate goal” (ZMRG) of 10% of the potential biological removal (50 C.F.R. § 229.2). Recent marine mammal bycatch estimates range from 0.1-4.0% of their potential biological removals and common murre bycatch has been effectively eliminated. All of the evaluated populations, including the once declining and heavily depleted Morro Bay, have shown significant improvements in bycatch rates. The Morro Bay population, which was once heavily depleted, has shown a remarkable recovery, with bycatch rates now well below the ZMRG. The California sea lion and harbor seal populations have also shown improvements, with bycatch rates now well below the ZMRG. The northern elephant seal population has shown a significant reduction in bycatch rates, with bycatch rates now well below the ZMRG. The common murre population has shown a significant reduction in bycatch rates, with bycatch rates now well below the ZMRG.

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4 472 Bay harbor porpoise population, are growing or stable. These advances, while directly attributable to
5 management interventions, are more due to reductions in fishing effort (i.e., fewer fishing trips) than to
6 reductions in bycatch rates (i.e., lower bycatch per fishing trip). This highlights a steep tradeoff between
7 conservation and fisheries objectives under the current management regime: while populations of
8 protected species have undergone sustained growth, fishing opportunities and revenues have undergone
9 prolonged declines. Despite this, there have been calls for more bycatch-motivated restrictions to the
10 fishery (Birch et al., 2023; Birch and Shester, 2023). Our results indicate that current fishing operations
11 do not pose a threat to the evaluated species, which suggests that current management is sufficient at
12 limiting bycatch. However, targeting management toward spatial-temporal bycatch hotspots could
13 improve economic outcomes while keeping bycatch low.
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23 483 Our results suggest that spatial-temporal management could more efficiently and effectively
24 manage bycatch risk than gear modifications or soak time regulations. Specifically, bycatch rates for
25 California sea lions and harbor seals are greatest from April 1 to June 15, suggesting that a 2.5 month
26 seasonal closure of bycatch hotspots for these two species could prevent bycatch while allowing the
27 opening of less risky but currently closed areas to fishing. These hotspots are predicted to
28 disproportionately contribute to bycatch yet are of minor fishing importance (**Fig. 6B**) suggesting that the
29 loss of fishing opportunities in these areas during brief seasonal closures could be easily made up by
30 opening areas of low predicted bycatch risk. As a result, seasonal closures could broaden fishing
31 opportunities while continuing to meet bycatch avoidance objectives. However, we caution that such
32 hotspot closures could exacerbate bycatch problems if fishing effort is displaced and concentrated in areas
33 of secondarily high risk (Free et al., 2023). Therefore, monitoring of fishing effort and bycatch rates are
34 important for verifying that seasonal closures achieve their conservation and fisheries objectives.
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36 491 Additionally, any changes in current management strategies must take into account spatial-temporal
37 patterns of bycatch and relative sensitivity of each species.
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41 498 Hotspots of bycatch risk are aligned with the location of known haulouts, breeding colonies, and
42 foraging grounds. High California seal lion bycatch risk around the northern Channel Islands is likely
43 related to the large haulouts of sea lions in that area (**Fig. S15**). Similarly, hotspots of harbor seal bycatch
44 risk correspond to the locations of large harbor seal haulouts on Santa Cruz Island and near Point Mugu
45 (**Fig. S15**). The absence of common murre bycatch risk in southern California is consistent with the
46 distribution of the species, which has no breeding colonies or permanent foraging grounds in southern
47 California (**Fig. S15**). Similarly, the negligible risk for northern elephant seals is consistent with the
48 phenology of their migrations. Although northern elephant seals breed on the Channel Islands and near
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4 506 San Simeon/Cambria (**Fig. S15**) from December to March, they disperse to their distant foraging grounds
5 (males to Alaska and females to oceanic waters far West of California) before the fishing season peaks
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7 508 from April to June, significantly reducing their vulnerability to the gillnet fishery. Finally, the elimination
8 of harbor porpoise bycatch is consistent with the confinement of the fishery to southern California where
9 harbor porpoise do not occur (**Fig. S15**).
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14 512 The completion and safekeeping of accurate fisheries-dependent data is fundamental to producing
15 accurate bycatch assessments and effective management strategies. While summaries of historical data
16 facilitated reliable bycatch estimates through ratio estimation, the loss of raw observer data from the
17 514 1980s likely impeded our ability to accurately estimate bycatch in the northern strata using random
18 forests, an approach often thought to be more accurate than ratio estimation (Stock et al., 2019). Notably,
19 515 the lost data document a period when fishing was allowed in shallower, more inshore, and more northern
20 waters (**Fig. 1A**). Recovering this data would enhance our ability to assess the drivers of bycatch in the
21 516 northern region and could provide insights for re-evaluating previous management strategies.
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23 517 Furthermore, missing meta-data on critical gear characteristics (e.g., mesh size, net length, net height, net
24 518 material; **Fig. S3**) in the available observer data also limited our ability to identify the potential for these
25 management levers to reduce bycatch risk. Ensuring the complete documentation of gear characteristics,
26 519 perhaps by prioritizing characteristics known to impact bycatch risk in other gillnet fisheries (Northridge
27 520 et al., 2017), is important to maximizing the utility of expensive, and sometimes controversial, observer
28 programs (Suuronen and Gilman, 2020). Finally, the ability to delineate individual sets in the logbooks
29 521 and improved documentation of the characteristics of logged sets would enhance future bycatch estimates
30 522 by allowing sets to be the sampling unit and by avoiding assumptions about missing data, respectively.
31 523 This could be achieved by redesigning logbooks, training fishers on completing logbooks, expanding
32 524 electronic monitoring, and/or demonstrating that better data can actually lead to fewer restrictions.
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Our results highlight the importance of considering multiple modeling approaches when estimating and evaluating rare bycatch events. Although model-based methods (e.g., random forests) for estimating bycatch are often preferred to sample-based methods (e.g., ratio estimators) (Stock et al., 2019), we find complementary value in using both approaches. While ratio estimation generated more reliable bycatch estimates due to its ability to leverage both raw and summarized data, the random forest model provided the empirical basis for assessing drivers of bycatch risk. Furthermore, our results highlight the value of considering multiple sample balancing approaches when evaluating bycatch using model-based methods, as the specification of the best performing model varied by species. Recent efforts to estimate bycatch in West Coast fisheries using random forests have used only a single sample

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4 540 balancing technique (e.g., Carretta, 2023); we encourage future efforts to compete multiple approaches to
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6 541 optimize estimates of rare bycatch events (More and Rana, 2017).
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9 543 The continued recovery of protected species will require management of stressors besides
10 fisheries bycatch, some of which may present even larger threats (Avila et al., 2018; Oldach et al., 2022).
11 544 For example, in Summer 2023, there were over 1,000 statewide strandings of California sea lions and
12 other pinnipeds attributed to domoic acid toxicosis resulting from an intense bloom of harmful diatoms in
13 the *Pseudo-nitzschia* genus (SCCOOS, 2023; Smith et al., 2023). Harmful algal blooms are increasing in
14 frequency, duration, and intensity on the West Coast (Hallegraeff et al., 2021) as a result of ocean
15 warming and eutrophication (McKibben et al., 2017) suggesting that, in the long-term, curbing climate
16 change and nutrient runoff may be the most important actions for stemming mortality for recovering
17 pinniped populations. Furthermore, harassment and shooting are some of the most commonly observed
18 sources of mortality and serious injury for California sea lions, harbor seals, and northern elephant seals
19 (Carretta, 2023) (**Fig. S16**), suggesting the need for greater outreach and enforcement to prevent these
20 gratuitous forms of mortality. Finally, it is important to understand the bycatch contributions of other
21 fisheries, many of which report higher levels of *observed* bycatch (**Fig. S16**) yet are not as heavily
22 prosecuted as the California set gillnet fishery. Modeling studies similar to this one are needed to
23 determine whether higher apparent bycatch in these fisheries is due to higher observer coverage or higher
24 bycatch rates. However, the sustained recovery of the evaluated populations suggests that total bycatch
25 across all fisheries is low.
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29 561 There will always be tradeoffs between maximizing fishing opportunities and minimizing bycatch
30 of protected species (Samhouri et al., 2021). As a result, managers often seek to implement regulations
31 that maximize fishing outcomes while keeping bycatch below legally defined sustainable reference points
32 (Kirby and Ward, 2014). The identification of such strategies is seldom straightforward and depends on
33 substantial investments in data and scientific enterprises. Notably, they depend on monitoring populations
34 of protected species to support the assessment of their status and levels of allowable incidental take and
35 monitoring bycatch in key fisheries to support assessments of total bycatch, drivers of bycatch, and the
36 effectiveness of past management interventions (Kirby and Ward, 2014; Punt et al., 2021). In the absence
37 of such data, management must often be precautionary to ensure compliance with protected species
38 legislation (Punt et al., 2021). We illustrate the potential return on investment of supporting such
39 scientific enterprises as our results show that past management interventions have been successful at
40 reducing bycatch in the California set gillnet fishery well below target levels, opening the door for more
41 efficient restrictions and negating the need for unnecessary precaution. The continued demonstration that
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4 574 monitoring programs can generate better outcomes for businesses could facilitate increased public support
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6 575 and funding to identify win-win scenarios for fisheries and conservation in more fisheries.
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- 7 577 Avila, I.C., Kaschner, K., Dormann, C.F., 2018. Current global risks to marine mammals: Taking
8 578 stock of the threats. *Biol. Conserv.* 221, 44–58.
9 579 <https://doi.org/10.1016/j.biocon.2018.02.021>
- 10 580 Barlow, J., Baird, R.W., Heyning, J.E., Wynne, K., Manville, A.M., Lowry, L.F., Hanan, D.,
11 581 Sease, J., Burkanov, V., 1994. A Review of Cetacean and Pinniped Mortality in Coastal
12 582 Fisheries Along the West Coast of the USA and Canada and the East Coast of the
13 583 Russian Federation. *Rep. Int. Whal. Comm.* 15, 405–426.
- 14 584 Bettoli, P.W., Scholten, G.D., 2006. Bycatch rates and initial mortality of paddlefish in a
15 585 commercial gillnet fishery. *Fish. Res.* 77, 343–347.
16 586 <https://doi.org/10.1016/j.fishres.2005.11.008>
- 17 587 Birch, C., Blacow-Draeger, A., Geoff Shester, PH.D., Webb, S., Purcell, E., 2023. The Net
18 588 Consequence: Impacts of Set Gillnets on California Ocean Biodiversity. Zenodo.
19 589 <https://doi.org/10.5281/ZENODO.7971874>
- 20 590 Birch, C., Shester, G., 2023. Underreporting of Marine Mammal Takes in the California Set
21 591 Gillnet Fishery Underscores the Need for Observers. *Oceana*.
- 22 592 Bjørge, A., Skern-Mauritzen, M., Rossman, M.C., 2013. Estimated bycatch of harbour porpoise
23 593 (*Phocoena phocoena*) in two coastal gillnet fisheries in Norway, 2006–2008. Mitigation
24 594 and implications for conservation. *Biol. Conserv.* 161, 164–173.
25 595 <https://doi.org/10.1016/j.biocon.2013.03.009>
- 26 596 CA Secretary of State, 1990. Proposition 132. Marine Resources. Initiative Constitutional
27 597 Amendment.
- 28 598 Cameron, G.A., Forney, K.A., 2000. Preliminary Estimates of Cetacean Mortality in
29 599 California/Oregon Gillnet Fisheries for 1999 (International Whaling Commission Working
30 600 Paper No. SC/52/O24).
- 31 601 Cameron, G.A., Forney, K.A., 1999. Preliminary Estimates of Cetacean Mortality in the
32 602 California Gillnet Fisheries for 1997 and 1998 (No. SC/51/04). Southwest Fisheries
33 603 Science Center.
- 34 604 Capitolo, P.J., McChesney, G.J., Bechaver, C.A., Rhoades, S.J., Shore, A., Carter, H.R.,
35 605 Parker, M.W., Eigner, L.E., 2012. Breeding population trends of Brandt's and double-
36 606 crested cormorants, Point Sur to Point Mugu, California, 1979–2011.
- 37 607 Carretta, J.V., 2023. Sources of human-related injury and mortality for U.S. Pacific West Coast
38 608 marine mammal stock assessments, 2017–2021. <https://doi.org/10.25923/QWF2-9B97>
- 39 609 Carretta, J.V., 2002. Preliminary estimates of cetacean mortality in California gillnet fisheries for
40 610 2001 (No. SC/54/SM12). International Whaling Commission, Scientific Committee,
41 611 Shimonoseki, Japan.
- 42 612 Carretta, J.V., 2001. Preliminary estimates of cetacean mortality in California gillnet fisheries for
43 613 2000 (SC/53/SM9). International Whaling Commission, Scientific Committee, London,
44 614 UK., London, UK.
- 45 615 Carretta, J.V., Chivers, S.J., 2004. Preliminary estimates of marine mammal mortality and
46 616 biological sampling of cetaceans in California gillnet fisheries for 2003 (No. SC/56/SM1).
- 47 617 Carretta, J.V., Chivers, S.J., 2003. Preliminary estimates of marine mammal mortality and
48 618 biological sampling of cetaceans in California gillnet fisheries for 2002 (No. SC/55/SM3).
- 49 619 Carretta, J.V., Enriquez, L., 2012a. Marine mammal and seabird bycatch in California gillnet
50 620 fisheries in 2010 (Administrative Report No. LJ-12-01). NOAA Southwest Fisheries
51 621 Science Center, La Jolla, CA.
- 52 622 Carretta, J.V., Enriquez, L., 2012b. Marine mammal and seabird bycatch in California gillnet
53 623 fisheries in 2011 (NOAA Technical Memorandum NMFS No. NOAA-TM-NMFS-SWFSC-
54 624 500). NOAA Southwest Fisheries Science Center, La Jolla, CA.

1
2
3
4 625 Carretta, J.V., Enriquez, L., 2009. Marine mammal and seabird bycatch in observed California
5 626 commercial fisheries in 2007 (Administrative Report No. LJ-09-01). NOAA Southwest
6 627 Fisheries Science Center, La Jolla, CA.
7 628 Carretta, J.V., Enriquez, L., Villafana, C., 2014. Marine mammal, sea turtle, and seabird bycatch
8 629 in California gillnet fisheries in 2012 (NOAA Technical Memorandum NMFS No. NOAA-
10 630 TM-NMFS-SWFSC-526). NOAA Southwest Fisheries Science Center, La Jolla, CA.
11 631 Carretta, J.V., Oleson, E.M., Forney, K.A., Muto, M.M., Weller, D.W., Lang, A.R., Baker, J.,
12 632 Hanson, B., Orr, A.J., Barlow, J., Moore, J.E., Brownell Jr, R.L., 2022. U.S. Pacific
13 633 marine mammal stock assessments: 2021 (NOAA Technical Memorandum No. NOAA-
14 634 TM-NMFS-SWFSC-663). NOAA Southwest Fisheries Science Center, La Jolla, CA.
15 635 Carter, H.R., 2001. Population Trends of the Common Murre (*Uria aalge californica*), in: Biology
16 636 and Conservation of the Common Murre in California, Oregon, Washington, and British
17 637 Columbia Volume 1: Natural History and Population Trends.
18 638 CDFW, 2023. Evaluating Bycatch in the California Halibut Set Gill Net Fishery. California
19 639 Department of Fish and Wildlife, Sacramento, CA.
20 640 CDFW, 2021. California Wildlife Habitat Relationship System.
21 641 CDFW, 2019. California Pacific Herring Fishery Management Plan.
22 642 CDFW, 2014. Harbor Seals [ds106] GIS Dataset.
23 643 CDFW, 2010. Seabird Colonies: California, 2010.
24 644 CDFW, 2002. Bathymetry Project: 25m bathymetry dataset.
25 645 Cochran, W.G., 1977. Sampling Techniques. John Wiley and Sons.
26 646 Cohen, J., 1968. Weighted kappa: Nominal scale agreement provision for scaled disagreement
27 647 or partial credit. *Psychol. Bull.* 70, 213–220. <https://doi.org/10.1037/h0026256>
28 648 Condylos, S., 2023. priceR: Economics and Pricing Tools.
29 649 Crowder, L.B., Murawski, S.A., 1998. Fisheries Bycatch: Implications for Management.
30 650 *Fisheries* 23, 8–17. [https://doi.org/10.1577/1548-8446\(1998\)023<0008:FBIFM>2.0.CO;2](https://doi.org/10.1577/1548-8446(1998)023<0008:FBIFM>2.0.CO;2)
31 651 Cutler, D.R., Edwards, T.C., Beard, K.H., Cutler, A., Hess, K.T., Gibson, J., Lawler, J.J., 2007.
32 652 Random Forests for Classification in Ecology. *Ecology* 88, 2783–2792.
33 653 Diamond, S.L., Hanan, D.A., 1986. An Estimate of Harbor Porpoise Mortality in California Set
34 654 Net Fisheries, April 1, 1983 through March 31, 1984 (Administrative Report No. SWR-
35 655 86-16). NOAA Southwest Fisheries Science Center, La Jolla, CA.
36 656 Fleiss, J.L., Levin, B., Paik, M.C., 2013. Statistical Methods for Rates and Proportions. John
37 657 Wiley & Sons.
38 658 Forney, K.A., Benson, S.R., Cameron, G.A., 2001. Central California gillnet effort and bycatch of
39 659 sensitive species, 1990-1998, in: Melvin, E., Parrish, J.K. (Eds.), Seabird Bycatch:
40 660 Trends, Roadblocks, and Solutions. Alaska Sea Grant, University of Alaska Fairbanks,
41 661 pp. 141–160. <https://doi.org/10.4027/sbtrs.2001.08>
42 662 Forney, K.A., Carretta, J.V., Benson, S.R., 2014. Preliminary estimates of harbor porpoise
43 663 abundance in Pacific Coast waters of California, Oregon and Washington, 2007-2012.
44 664 Forney, K.A., Moore, J.E., Barlow, J., Carretta, J.V., Benson, S.R., 2021. A multidecadal
45 665 Bayesian trend analysis of harbor porpoise (*Phocoena phocoena*) populations off
46 666 California relative to past fishery bycatch. *Mar. Mammal Sci.* 37, 546–560.
47 667 <https://doi.org/10.1111/mms.12764>
48 668 Free, C.M., Bellquist, L.F., Forney, K.A., Humberstone, J., Kauer, K., Lee, Q., Liu, O.R.,
49 669 Samhouri, J.F., Wilson, J.R., Bradley, D., 2023. Static management presents a simple
50 670 solution to a dynamic fishery and conservation challenge. *Biol. Conserv.* 285, 110249.
51 671 <https://doi.org/10.1016/j.biocon.2023.110249>
52 672 Freeman, E.A., Moisen, G.G., 2008. A comparison of the performance of threshold criteria for
53 673 binary classification in terms of predicted prevalence and kappa. *Ecol. Model.* 217, 48–
54 674 58. <https://doi.org/10.1016/j.ecolmodel.2008.05.015>
55 675 Geijer, C.K.A., Read, A.J., 2013. Mitigation of marine mammal bycatch in U.S. fisheries since
56
57
58
59
60
61
62
63
64
65

- 1
2
3
- 4 676 1994. *Biol. Conserv.* 159, 54–60. <https://doi.org/10.1016/j.biocon.2012.11.009>
- 5 677 Hallegraeff, G.M., Anderson, D.M., Belin, C., Bottein, M.-Y.D., Bresnan, E., Chinain, M.,
6 Enevoldsen, H., Iwataki, M., Karlson, B., McKenzie, C.H., Sunesen, I., Pitcher, G.C.,
7 Provoost, P., Richardson, A., Schweibold, L., Tester, P.A., Trainer, V.L., Yñiguez, A.T.,
8 Zingone, A., 2021. Perceived global increase in algal blooms is attributable to intensified
9 monitoring and emerging bloom impacts. *Commun. Earth Environ.* 2, 117.
10 https://doi.org/10.1038/s43247-021-00178-8
- 11 682 Hanan, D.A., Diamond, S.L., 1989. Estimates of Sea Lion, Harbor Seal, and Harbor Porpoise
12 Mortalities in California Set Net Fisheries for the 1986-87 Fishing Year.
13 684 Hanan, D.A., Diamond, S.L., Scholl, J.P., 1988. Estimates of Sea Lion and Harbor Seal
14 Mortalities in California Set Net Fisheries for 1983, 1984, and 1985.
15 686 Hanan, D.A., Diamond, S.L., Scholl, J.P., 1987. An Estimate of Harbor Porpoise Mortalities in
16 California Set Net Fisheries, April 1, 1985 through March 31, 1986 (Administrative
17 Report No. SWR 87-5). NOAA Southwest Fisheries Science Center, La Jolla, CA.
18 688 Hanan, D.A., Diamond, S.L., Scholl, J.P., 1986. An Estimate of Harbor Porpoise Mortality in
19 California Set Net Fisheries April 1, 1984 through March 31, 1985 (Administrative Report
20 No. SWR 86-16). NOAA Southwest Fisheries Science Center, La Jolla, CA.
21 690 Hazen, E.L., Scales, K.L., Maxwell, S.M., Briscoe, D.K., Welch, H., Bograd, S.J., Bailey, H.,
22 Benson, S.R., Eguchi, T., Dewar, H., Kohin, S., Costa, D.P., Crowder, L.B., Lewison,
23 R.L., 2018. A dynamic ocean management tool to reduce bycatch and support
24 sustainable fisheries. *Sci. Adv.* 4, eaar3001. <https://doi.org/10.1126/sciadv.aar3001>
- 25 694 Huang, B., Liu, C., Banzon, V., Freeman, E., Graham, G., Hankins, B., Smith, T., Zhang, H.-M.,
26 695 2021. Improvements of the Daily Optimum Interpolation Sea Surface Temperature
27 (DOISST) Version 2.1. *J. Clim.* 34, 2923–2939. <https://doi.org/10.1175/JCLI-D-20-0166.1>
- 28 697 Hvitfeldt, E., 2023. themis: Extra Recipes Steps for Dealing with Unbalanced Data. R package.
29 701 ICES, 2007. Report of the Workshop on Discard Raising Procedures (ICES CM No.
30 703 2007ACFM:06). ICES, San Sebastian, Spain.
- 31 704 Julian, F., 1994. Pinniped and cetacean mortality in California gillnet fisheries: preliminary
32 705 estimates for 1993 (International Whaling Commission No. SC/46/0). NOAA Southwest
33 706 Fisheries Science Center, La Jolla, CA.
- 34 707 Julian, F., 1993. Pinniped and cetacean mortality in California gillnet fisheries: preliminary
35 708 estimates for 1992.
- 36 709 Julian, F., Beeson, M., 1998. Estimates of marine mammal, turtle, and seabird mortality for two
37 710 California gillnet fisheries: 1990-1995. *Fish. Bull.* 96, 271–284.
- 38 711 Kirby, D.S., Ward, P., 2014. Standards for the effective management of fisheries bycatch. *Mar.
39 712 Policy* 44, 419–426. <https://doi.org/10.1016/j.marpol.2013.10.008>
- 40 713 Konno, E.S., 1990. Effort Estimates of Gill Net Fisheries in California that Incidentally Catch
41 714 Marine Mammals, for the 1987-88 Fishing Year. NOAA Southwest Fisheries Science
42 715 Center, Terminal Island, CA.
- 43 716 Kroetz, A.M., Mathers, A.N., Carlson, J.K., 2020. Evaluating protected species bycatch in the
44 717 U.S. Southeast Gillnet Fishery. *Fish. Res.* 228, 105573.
45 718 https://doi.org/10.1016/j.fishres.2020.105573
- 46 719 Laake, J.L., Lowry, M.S., DeLong, R.L., Melin, S.R., Carretta, J.V., 2018. Population growth and
47 720 status of California sea lions. *J. Wildl. Manag.* 82, 583–595.
48 721 https://doi.org/10.1002/jwmg.21405
- 49 722 Landis, J.R., Koch, G.G., 1977. The Measurement of Observer Agreement for Categorical Data.
50 723 *Biometrics* 33, 159–174. <https://doi.org/10.2307/2529310>
- 51 724 Lennert, C., Kruse, S., Beeson, M., Barlow, J., 1994. Estimates of incidental marine mammal
52 725 bycatch in California gillnet fisheries for July through December, 1990 (Report of the
53 726 International Whaling Commission No. SC/43/O 3).
- 54
55
56
57
58
59
60
61
62
63
64
65

- 1
2
3
- 4 727 Lewison, R.L., Crowder, L.B., Wallace, B.P., Moore, J.E., Cox, T., Zydelis, R., McDonald, S.,
5 728 DiMatteo, A., Dunn, D.C., Kot, C.Y., Bjorkland, R., Kelez, S., Soykan, C., Stewart, K.R.,
6 729 Sims, M., Boustany, A., Read, A.J., Halpin, P., Nichols, W.J., Safina, C., 2014. Global
7 730 patterns of marine mammal, seabird, and sea turtle bycatch reveal taxa-specific and
8 731 cumulative megafauna hotspots. Proc. Natl. Acad. Sci. 111, 5271–5276.
9 732 <https://doi.org/10.1073/pnas.1318960111>
- 10 733 Liaw, A., Wiener, M., 2002. Classification and Regression by randomForest. R News 2, 18–22.
- 11 734 Long, C.A., Ahrens, R.N.M., Jones, T.T., Siders, Z.A., 2024. A machine learning approach for
12 735 protected species bycatch estimation. Front. Mar. Sci. 11.
13 736 <https://doi.org/10.3389/fmars.2024.1331292>
- 14 737 Lopez, J., Griffiths, S., Wallace, B.P., Cáceres, V., Rodríguez, L.H., Abrego, M., Alfaro-
15 738 Shigueto, J., Andraka, S., Brito, M.J., Bustos, L.C., Cari, I., Carvajal, J.M., Clavijo, L.,
16 739 Cocas, L., Paz, N. de, Herrera, M., Mangel, J.C., Pérez-Huaripata, M., Piedra, R.,
17 740 Dávila, J.A.Q., Rendón, L., Rguez-Baron, J.M., Santana, H., Suárez, J., Veelenturf, C.,
18 741 Vega, R., Zárate, P., 2024. Vulnerability of the Critically Endangered leatherback turtle to
19 742 fisheries bycatch in the eastern Pacific Ocean. I. A machine-learning species distribution
20 743 model. Endanger. Species Res. 53, 271–293. <https://doi.org/10.3354/esr01288>
- 21 744 Lowry, M., Condit, R., Hatfield, B., Allen, S.G., Berger, R., Morris, P.A., Le Boeuf, B.J., Reiter,
22 745 J., 2014. Abundance, Distribution, and Population Growth of the Northern Elephant Seal
23 746 (*Mirounga angustirostris*) in the United States from 1991 to 2010. Aquat. Mamm. 40, 20–
24 747 31. <https://doi.org/10.1578/AM.40.1.2014.20>
- 25 748 Lowry, M.S., 2021. Abundance and distribution of pinnipeds at the Channel Islands in southern
26 749 California, central and northern California, and southern Oregon during summer 2016–
27 750 2019 (NOAA Technical Memorandum NMFS No. NOAA-TM-NMFS-SWFSC-656).
28 751 NOAA Southwest Fisheries Science Center, La Jolla, CA.
- 29 752 Manel, S., Williams, H.C., Ormerod, S. j., 2001. Evaluating presence–absence models in
30 753 ecology: the need to account for prevalence. J. Appl. Ecol. 38, 921–931.
31 754 <https://doi.org/10.1046/j.1365-2664.2001.00647.x>
- 32 755 Martin, T.G., Wintle, B.A., Rhodes, J.R., Kuhnert, P.M., Field, S.A., Low-Choy, S.J., Tyre, A.J.,
33 756 Possingham, H.P., 2005. Zero tolerance ecology: improving ecological inference by
34 757 modelling the source of zero observations. Ecol. Lett. 8, 1235–1246.
35 758 <https://doi.org/10.1111/j.1461-0248.2005.00826.x>
- 36 759 McCracken, M.L., 2004. Modeling a Very Rare Event to Estimate Sea Turtle Bycatch: Lessons
37 760 Learned (NOAA Technical Memorandum No. NMFS-PIFSC-3). Pacific Islands Fisheries
38 761 Science Center.
- 39 762 McKibben, S.M., Peterson, W., Wood, A.M., Trainer, V.L., Hunter, M., White, A.E., 2017.
40 763 Climatic regulation of the neurotoxin domoic acid. Proc. Natl. Acad. Sci. 114, 239–244.
41 764 <https://doi.org/10.1073/pnas.1606798114>
- 42 765 More, A.S., Rana, D.P., 2017. Review of random forest classification techniques to resolve data
43 766 imbalance, in: 2017 1st International Conference on Intelligent Systems and Information
44 767 Management (ICISIM). Presented at the 2017 1st International Conference on Intelligent
45 768 Systems and Information Management (ICISIM), IEEE, Aurangabad, pp. 72–78.
46 769 <https://doi.org/10.1109/ICISIM.2017.8122151>
- 47 770 Nembrini, S., König, I.R., Wright, M.N., 2018. The revival of the Gini importance? Bioinformatics
48 771 34, 3711–3718. <https://doi.org/10.1093/bioinformatics/bty373>
- 49 772 NOAA, 2024. CA Halibut, White Seabass and Other Species Set Gillnet (>3.5 in mesh) - MMPA
50 773 List of Fisheries [WWW Document]. NOAA. URL
51 774 <https://www.fisheries.noaa.gov/national/marine-mammal-protection/ca-halibut-white-seabass-and-other-species-set-gillnet-35-mesh> (accessed 6.4.24).
- 52 775 NOAA, 2023. List of Fisheries for 2024, Federal Register.
- 53 776 Northridge, S., Coram, A., Kingston, A., Crawford, R., 2017. Disentangling the causes of
54 777
- 55
56
57
58
59
60
61
62
63
64
65

1
2
3

4 778 protected-species bycatch in gillnet fisheries. Conserv. Biol. J. Soc. Conserv. Biol. 31,
5 779 686–695. <https://doi.org/10.1111/cobi.12741>

6 780 O'Keefe, C.E., Cadrin, S.X., Glemarec, G., Rouxel, Y., 2023. Efficacy of Time-Area Fishing
7 781 Restrictions and Gear-Switching as Solutions for Reducing Seabird Bycatch in Gillnet
8 782 Fisheries. Rev. Fish. Sci. Aquac. 31, 29–46.
9 783 <https://doi.org/10.1080/23308249.2021.1988051>

10 784 Oldach, E., Killeen, H., Shukla, P., Brauer, E., Carter, N., Fields, J., Thomsen, A., Cooper, C.,
11 785 Mellinger, L., Wang, K., Hendrickson, C., Neumann, A., Bøving, P.S., Fangue, N., 2022.
12 786 Managed and unmanaged whale mortality in the California Current Ecosystem. Mar.
13 787 Policy 140, 105039. <https://doi.org/10.1016/j.marpol.2022.105039>

14 788 Ortiz, M., Arocha, F., 2004. Alternative error distribution models for standardization of catch
15 789 rates of non-target species from a pelagic longline fishery: billfish species in the
16 790 Venezuelan tuna longline fishery. Fish. Res., Models in Fisheries Research: GLMs,
17 791 GAMS and GLMMs 70, 275–297. <https://doi.org/10.1016/j.fishres.2004.08.028>

18 792 Perkins, P., Barlow, J., Beeson, M., 1994. Report on Pinniped and Cetacean Mortality in
19 793 California Gillnet Fisheries: 1988-1990 (Administrative Report No. LJ-94-11). NOAA
20 794 Southwest Fisheries Science Center, La Jolla, CA.

21 795 Perkins, P., Barlow, J., Beeson, M., 1992a. Report on Pinniped and Cetacean Mortality in
22 796 California Gillnet Fisheries: 1990-1991 (Administrative Report No. LJ-92-14). NOAA
23 797 Southwest Fisheries Science Center, La Jolla, CA.

24 798 Perkins, P., Barlow, J., Beeson, M., 1992b. Pinniped and cetacean mortality in California gillnet
25 799 fisheries: 1991 (No. SC/44/SM14).

26 800 Prasad, A.M., Iverson, L.R., Liaw, A., 2006. Newer Classification and Regression Tree
27 801 Techniques: Bagging and Random Forests for Ecological Prediction. Ecosystems 9,
28 802 181–199. <https://doi.org/10.1007/s10021-005-0054-1>

29 803 Punt, A.E., Siple, M.C., Francis, T.B., Hammond, P.S., Heinemann, D., Long, K.J., Moore, J.,
30 804 Sepúlveda, M., Reeves, R.R., Sigurðsson, G.M., Víkingsson, G., Wade, P.R., Williams,
31 805 R., Zerbini, A.N., 2021. Can we manage marine mammal bycatch effectively in low-data
32 806 environments? J. Appl. Ecol. 58, 596–607. <https://doi.org/10.1111/1365-2664.13816>

33 807 R Core Team (2024). R: A language and environment for statistical computing. R Foundation for
34 808 Statistical Computing, Vienna, Austria. URL <https://www.R-project.org/>

35 809 Read, A.J., Drinker, P., Northridge, S., 2006. Bycatch of Marine Mammals in U.S. and Global
36 810 Fisheries. Conserv. Biol. 20, 163–169. <https://doi.org/10.1111/j.1523-1739.2006.00338.x>

37 811 Rochet, M.-J., Trenkel, V.M., 2005. Factors for the variability of discards: assumptions and field
38 812 evidence. Can. J. Fish. Aquat. Sci. 62, 224–235. <https://doi.org/10.1139/f04-185>

39 813 Samhouri, J.F., Feist, B.E., Fisher, M.C., Liu, O., Woodman, S.M., Abrahms, B., Forney, K.A.,
40 814 Hazen, E.L., Lawson, D., Redfern, J., Saez, L.E., 2021. Marine heatwave challenges
41 815 solutions to human–wildlife conflict. Proc. R. Soc. B Biol. Sci. 288, 20211607.
42 816 <https://doi.org/10.1098/rspb.2021.1607>

43 817 SCCOOS, 2023. California HAB Bulletin: May-July 2023 [WWW Document]. URL
44 818 <https://sccoos.org/california-hab-bulletin/may-2023/> (accessed 5.31.24).

45 819 Senko, J., White, E.R., Heppell, S.S., Gerber, L.R., 2014. Comparing bycatch mitigation
46 820 strategies for vulnerable marine megafauna. Anim. Conserv. 17, 5–18.
47 821 <https://doi.org/10.1111/acv.12051>

48 822 Smith, J., Cram, J.A., Berndt, M.P., Hoard, V., Shultz, D., Deming, A.C., 2023. Quantifying the
49 823 linkages between California sea lion (*Zalophus californianus*) strandings and particulate
50 824 domoic acid concentrations at piers across Southern California. Front. Mar. Sci. 10,
51 825 1278293. <https://doi.org/10.3389/fmars.2023.1278293>

52 826 Soykan, C., Moore, J., Zydelis, R., Crowder, L., Safina, C., Lewison, R., 2008. Why study
53 827 bycatch? An introduction to the Theme Section on fisheries bycatch. Endanger. Species
54 828 Res. 5, 91–102. <https://doi.org/10.3354/esr00175>

61
62
63
64
65

- 1
2
3
4 829 Stock, B.C., Ward, E.J., Thorson, J.T., Jannot, J.E., Semmens, B.X., 2019. The utility of spatial
5 830 model-based estimators of unobserved bycatch. ICES J. Mar. Sci. 76, 255–267.
6 831 <https://doi.org/doi:10.1093/icesjms/fsy153>
7 832 Suuronen, P., Gilman, E., 2020. Monitoring and managing fisheries discards: New technologies
8 833 and approaches. Mar. Policy 116, 103554. <https://doi.org/10.1016/j.marpol.2019.103554>
9 834 Wright, M.N., Ziegler, A., 2017. ranger: A Fast Implementation of Random Forests for High
10 835 Dimensional Data in C++ and R. J. Stat. Softw. 77, 1–17.
11 836 <https://doi.org/10.18637/jss.v077.i01>
12
13
14
15
16
17
18
19
20
21
22
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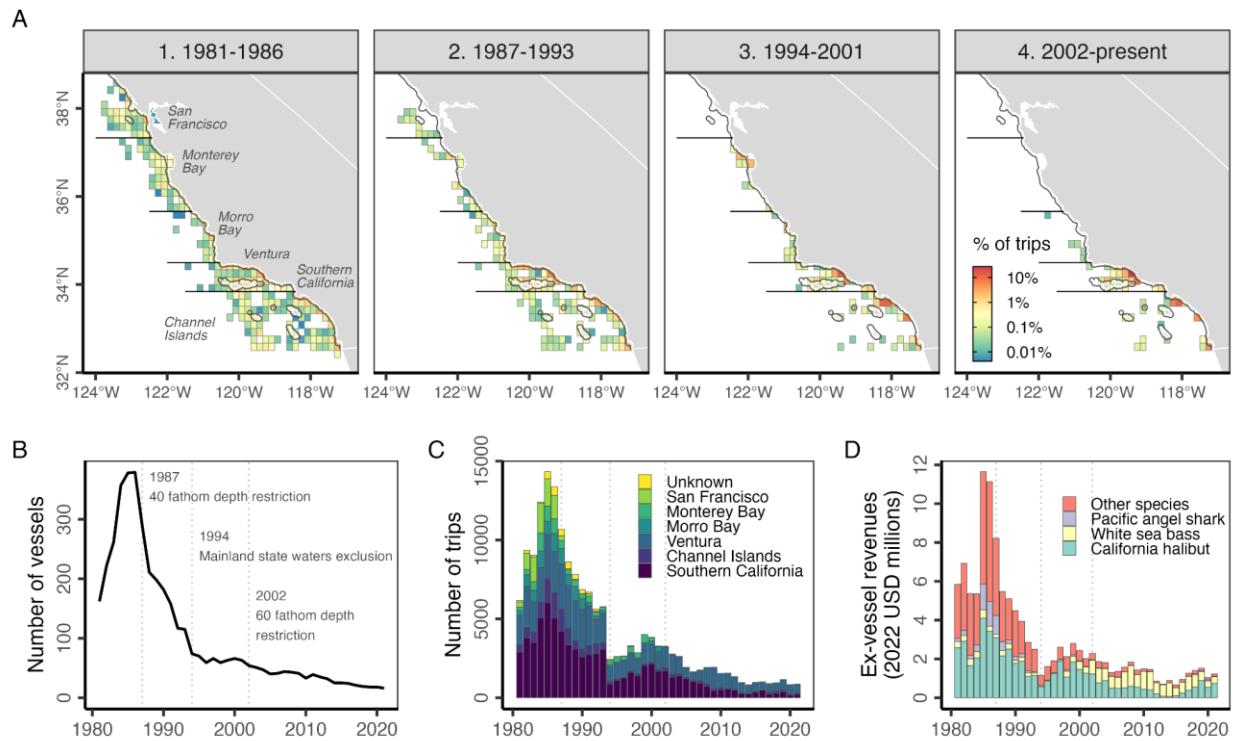
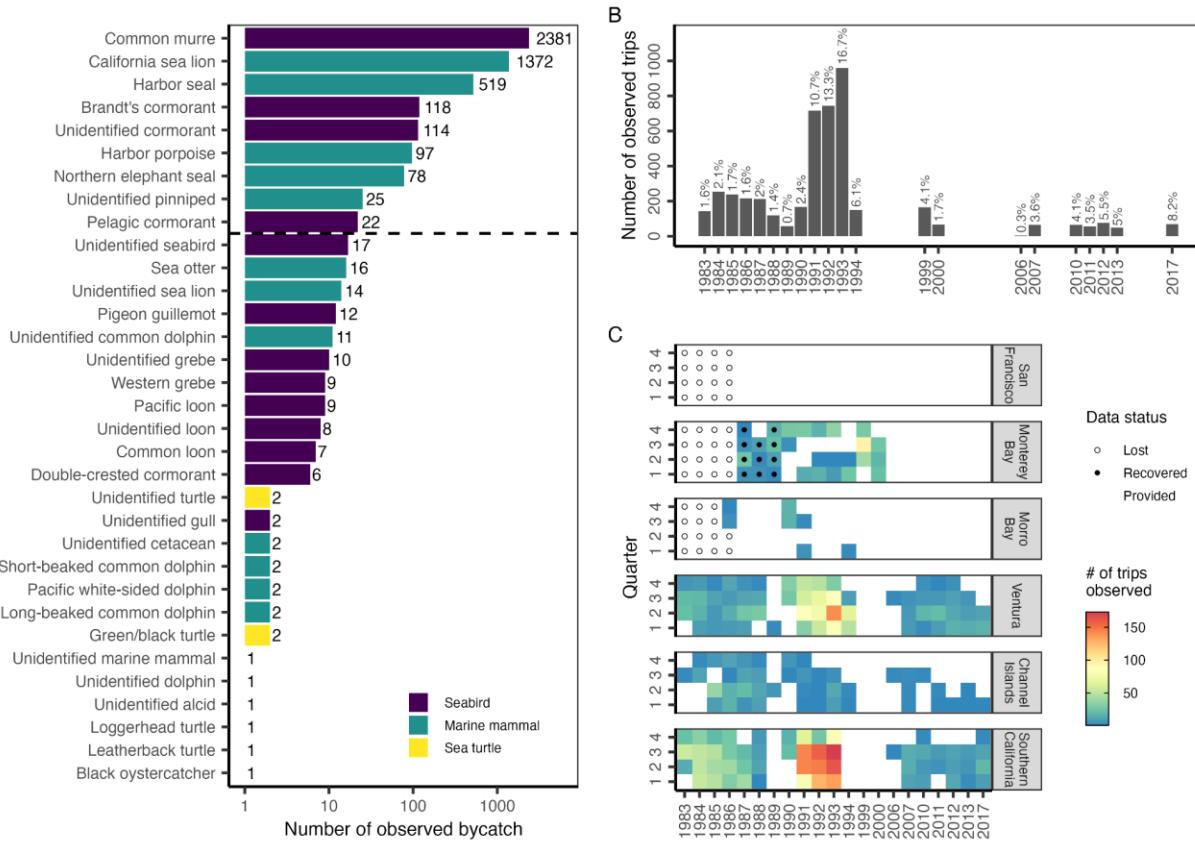


Fig. 1. History of the California $\geq 3.5"$ set gillnet fishery. Panel A shows the spatial history of fishing effort during four regulatory periods. Trips are reported by the 10×10 minute ($\sim 18 \times 18$ km) statistical blocks used for fisheries catch and effort reporting. The horizontal lines delineate geographical strata used in the ratio estimation analysis; strata are labeled in the first plot. The thin coastal line marks state waters (less than 3 nautical miles from the coast). Blocks visited by fewer than three vessels during each regulatory period are hidden to maintain confidentiality and comply with the “rule-of-three.” The other panels show time series of fisheries (B) participation, (C) effort, and (D) revenues. Vertical lines mark years in which major regulations, labeled in Panel B, were implemented; these define the regulatory periods used in Panel A. These regulations became operative on April 15, 1987; January 1, 1994; and April 26, 2002. See Fig. S1 and the supplemental methods for details on estimating ex-vessel revenues from the fishery.



31 850
32 851 **Fig. 2.** History of observer coverage in the California $\geq 3.5"$ set gillnet fishery. Panel A shows the bycatch
33 852 of marine mammals, seabirds, and sea turtles recorded by observers from 1983–2017. We focus on species
34 853 with ≥ 50 observations, which are delineated by the horizontal dashed line. Note log-scale on x-axis. Panel
35 854 B shows the number of observed trips (vessel-days) over time. The dark labels show the estimated percent
36 855 of trips that were observed. Panel C shows the number of observed trips across the spatial (region) and
37 856 temporal (quarters) strata considered in the ratio estimation analysis. See Fig. S8 for a map of the spatial
38 857 strata. Quarters are defined as: 1 = JFM (winter), 2 = AMJ (spring), 3 = JAS (summer), and 4 = OND
39 858 (fall).

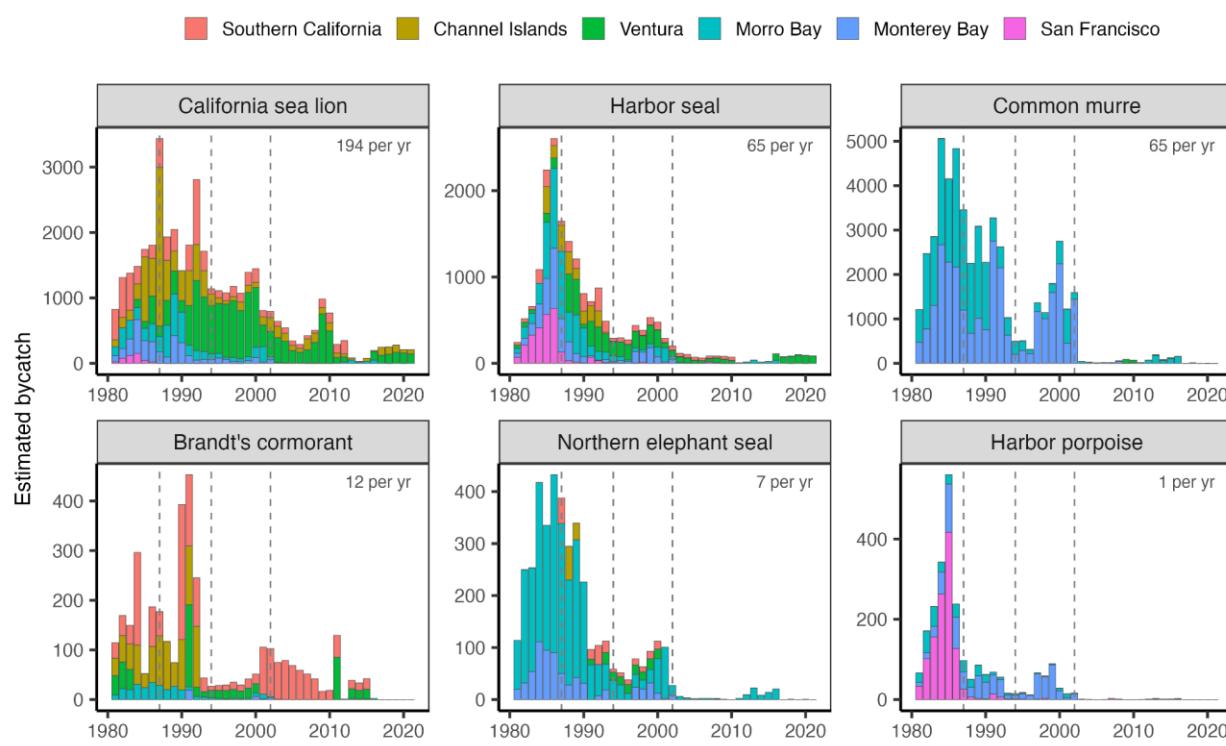


Fig. 3. Estimated bycatch in the California $\geq 3.5"$ set gillnet fishery from 1981-2021 predicted using the ratio estimation approach. Average estimated annual bycatch rates for the last 10 years (2012-2021) from the ratio estimator are marked in the top-right corner. Species are listed in order of decreasing recent bycatch rates.

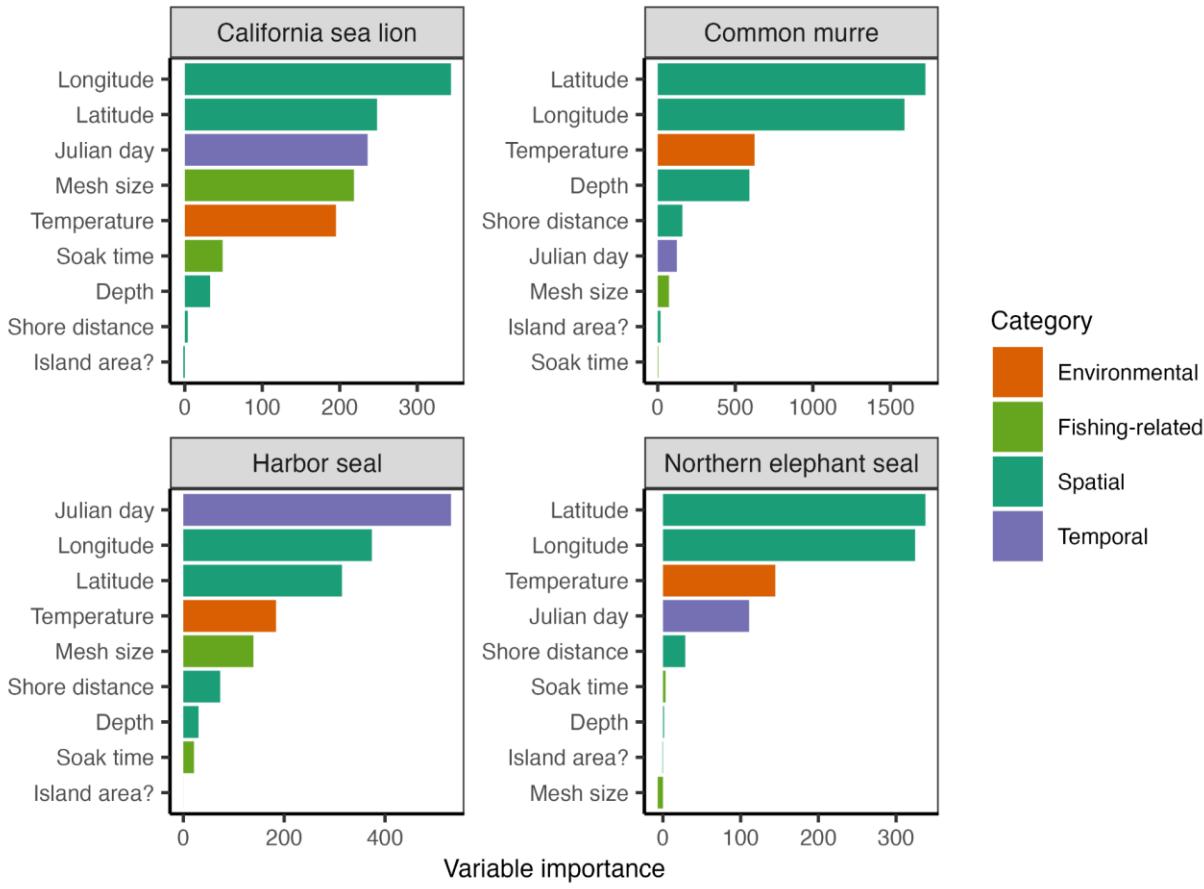
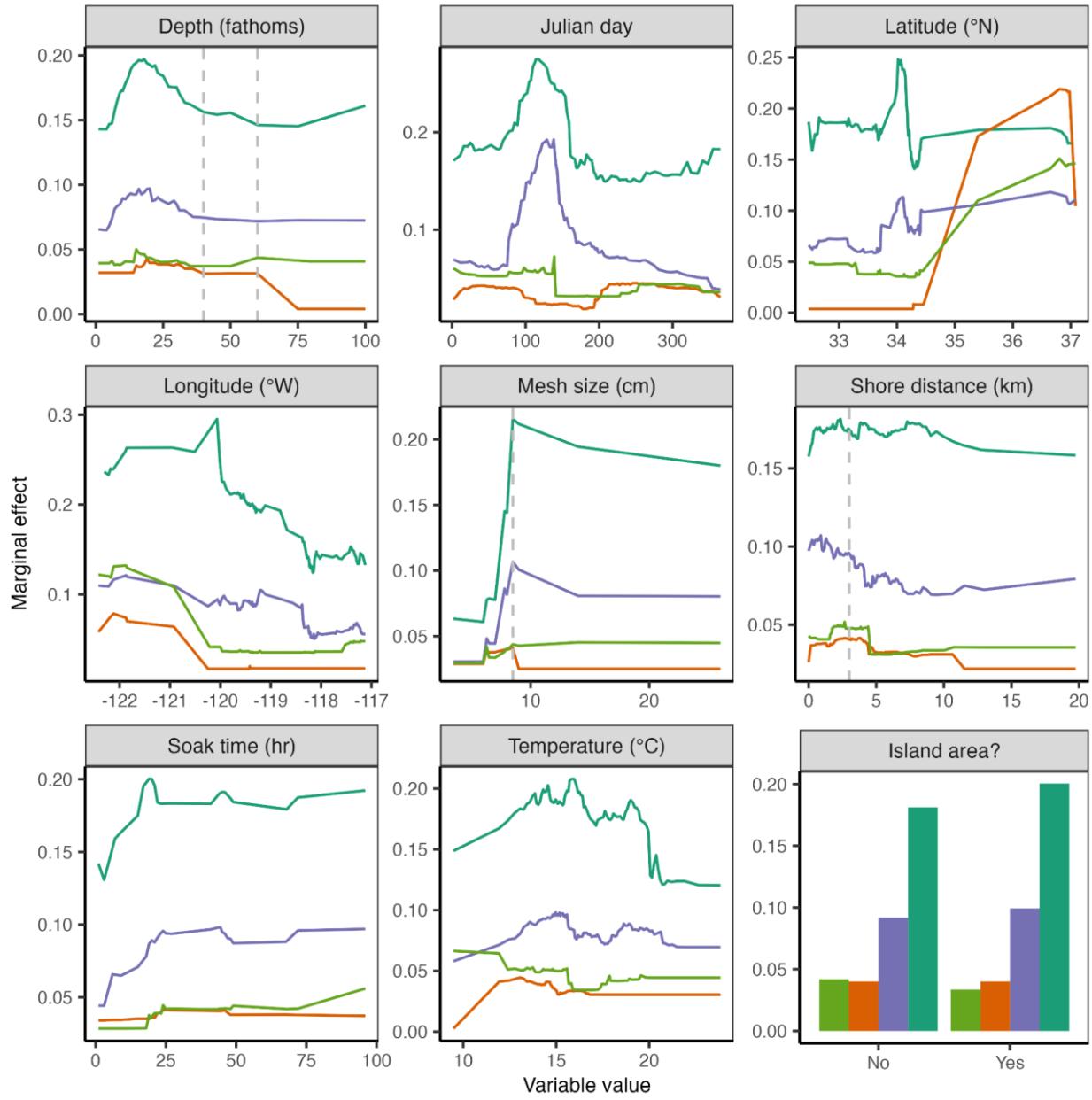


Fig. 4. Importance of the evaluated explanatory variables in the final random forest model for the study species with acceptable model performance. With the bias towards continuous variable corrected, variable importance is measured as the total decrease in node impurities from splitting on the variable averaged over all trees. Explanatory variables are colored based on the category of the variable.



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870 **Fig. 5.** Marginal effect of the evaluated explanatory variables on bycatch risk as estimated by the best
871 fitting random forest model for the four study species with acceptable model performance. The marginal
872 effect of each variable represents the effect of the variable when the other variables are held at their mean
873 values. The dashed lines indicate 40 and 60 in Depth (fathoms), 8.5 in mesh size (cm), and 3 in shore
874 distance (km).

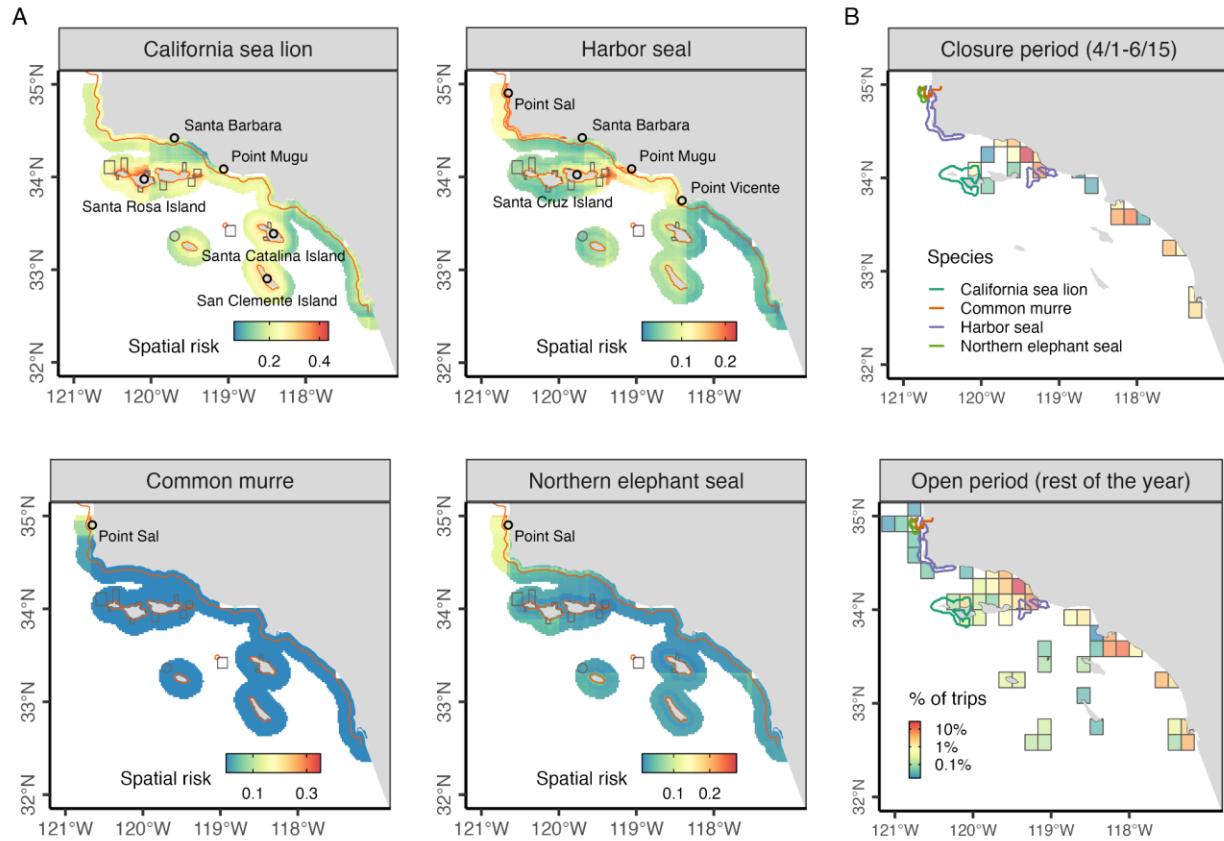


Fig. 6. The (A) spatial bycatch risk relative to current management areas and (B) spatial bycatch hotspots relative to recent fishing effort. Panel A shows the average spatial bycatch risk as estimated by the best fitting random forest model for the four study species with acceptable model performance. The spatial bycatch risk represents the probability of bycatch at a given location under recent (2010-2021) average conditions. Key landmarks for delineating bycatch hotspots are labeled in each panel. The thin orange coastal line marks the nearshore areas from which gillnet fishing is excluded: within 3 nautical miles of the mainland and within 1 nautical mile or shallower than 70 fathoms (whichever is closer to shore) from the Channel Islands. The gray polygons indicate the locations of California Marine Protected Areas, where all set gillnet fisheries are excluded. Spatial bycatch risk is shown only for southern California, as this is the only area where the fishery can operate under current regulations. Panel B shows hotspots of bycatch risk relative to recent fishing effort (2002-2022; see Fig 1A) during the proposed closed and open periods.

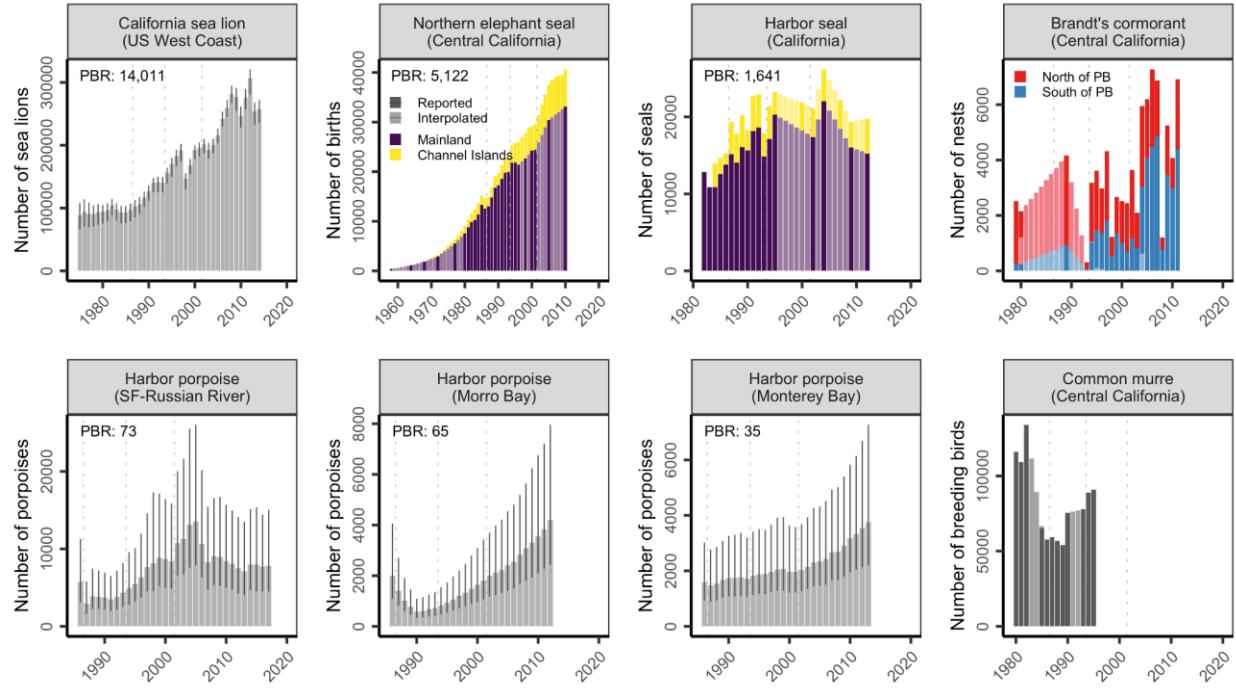


Fig. 7. Estimated abundance of populations of the six study species. The potential biological removal (PBR) indicates the maximum number of animals, not including natural mortalities, that may be removed from a marine mammal stock while allowing the stock to reach or maintain its optimum sustainable population. PBRs are only calculated for marine mammals. Error bars indicate 95% confidence or credible intervals. Years without reported values were filled using linear interpolation. Vertical lines mark years in which major bycatch regulations were implemented. Population estimates are from the following sources: California sea lion (Laake et al., 2018), northern elephant seal (Carretta et al., 2022), harbor seal (Carretta et al., 2022), harbor porpoise (Forney et al., 2021), Brandt's cormorant (Capitolo et al., 2012), and common murre (Carter, 2001). Data from Carretta et al. (2022) were graphically digitized.

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4 898 **Table 1.** Performance of the best fitting random forest model by species.
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Species	# of bycatch observed	Best model	Mtry	Training data		Test data	
				Kappa	AUC	Kappa	AUC
California sea lion	1372	Weighted-25	3	0.24	0.78	0.23	0.78
Harbor seal	519	Weighted-75	2	0.25	0.83	0.15	0.81
Harbor porpoise	97	Weighted-50	2	0.34	0.98	-0.005	0.98
Common murre	2381	Weighted-25	6	0.71	0.99	0.61	0.97
Brandt's cormorant	118	Weighted-25	8	0.06	0.68	0.07	0.63
Northern elephant seal	78	Weighted-25	1	0.23	0.87	0.21	0.86

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6 900 Supplemental Information
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10 901 Imputing missing values in observed and logbooks data
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12 902 We developed a series of simple assumptions to impute missing values for a few key variables
13 (GPS coordinates, fishing depth, soak hour, mesh size) reported in the observer data (**Fig. S3A; Table**
14 **S4**). We assigned missing GPS coordinates using the median coordinates for observed trips within the
15 statistical block most frequently visited by the vessel – in order of preference – that week, month, or year
16 based on the logbook data (described below). We derived missing fishing depths by extracting depths
17 from 25-meter resolution bathymetry data (CDFW, 2002) (**Fig. S3B**). We reassigned missing soak hours
18 the mode value for a vessel and target species (**Fig S3C**). We reassigned missing mesh sizes the mode for
19 – in order of preference – the vessel and target species, the target species, or all vessels (**Fig. S3DE**). We
20 assigned each GPS coordinate to the nearest statistical reporting block (see **Fig. 1A**), which allows points
21 erroneously falling on land to be assigned a likely statistical block. We derived the distance from shore, a
22 covariate used to explain bycatch rates in the random forest model, as the distance of each set to the
23 nearest point on shore.
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25 914 We developed a series of simple assumptions to impute missing or unrealistic values for a few
26 key variables (fishing depth, soak hour, mesh size) reported in the logbook data (**Fig. S5A; Table S4**).
27 915 We reassigned both missing (including 0 values) and unrealistic fishing depths, which we defined as
28 depths exceeding the maximum depth in the reported fishing block, the median depth of the fishing block
29 (**Fig. S5B**). We computed the median and maximum depths of each fishing block using 25-meter
30 resolution bathymetry data (CDFW, 2002). We reassigned missing soak hours (including 0 values) the
31 mode value for a vessel. We capped rare and unlikely soak times exceeding 96 hours (4 days) at 96 hours;
32 however, such soak times could theoretically occur during rough weather when it is unsafe to haul gear
33 (**Fig. S5C**). We reassigned missing (including 0 values) and unrealistic mesh sizes, which we defined as
34 mesh sizes exceeding 20 inches, using a hierarchical procedure (**Fig. S5DE**). For logbooks with both
35 vessel identification and target species information, we assigned the mesh size most commonly used by
36 the vessel when targeting that target species. For logbooks with only target species information (no vessel
37 identification), we assigned the mesh size most commonly used when targeting that target species across
38 all vessels (**Figs. S5 & S6**). We derived the distance from shore, a covariate used to explain bycatch rates
39 in the random forest model, as the median distance from shore of observed trips within the reported block
40 given that exact locations are not reported.
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4 930 Bycatch estimation using the random forest models
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7 931 We used the best fitting model to generate annual estimates of protected species bycatch from
8 932 1981 to 2022 by predicting whether “pseudo-sets” recorded in logbooks were likely to have captured each
9 933 study species and assuming median numbers of sets and caught animals for “pseudo-sets” with bycatch.
10 934 We predict to pseudo-sets rather than trips because the random forest model is trained on set-level
11 935 covariates in the observer data. We used the best fitting model for each species to estimate the probability
12 936 that a logged pseudo-set included bycatch of a species then categorized the pseudo-set as with or without
13 937 bycatch using a species-specific probability threshold. We derived the species-specific probability
14 938 thresholds as the threshold that maximizes Cohen’s kappa when applied to the training datasets (**Fig.**
15 939 **S17**). We selected the probability threshold based on Cohen’s kappa rather than the area under the
16 940 receiver operator curve (AUC) because (1) the models were tuned and selected based on Cohen’s kappa
17 941 and (2) simulation work shows that deriving thresholds based on AUC tends to overestimates the
18 942 prevalence of rare events while it underestimates the prevalence of common events (Freeman and Moisen,
19 943 2008; Manel et al., 2001). We summed the number of pseudo-sets predicted to have bycatch each year,
20 944 converted this sum to “true sets” assuming three sets per pseudo-set (**Fig. S4AB**), and multiplied this sum
21 945 by the median number of captures when a capture occurs to generate estimates of the total number of
22 946 captured animals (**Fig. S4C**). We opted not to employ a more complex two-stage or hurdle model
23 947 approach, where a second model estimates the number of captured individuals when bycatch occurs,
24 948 given the rarity of bycatch events larger than one for all species but common murre (**Fig. S4C**).
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27 949 The random forest models estimate trends in bycatch that are similar to the estimates from the
28 950 ratio estimator from 2000-2022 (**Fig. 3**). However, the estimates produced by the two approaches diverge
29 951 from 1981-2000 by various extents. While they generally agree for California sea lion back to 2000, the
30 952 random forest model underpredicts bycatch relative to the ratio estimator in the late 1990s and
31 953 overpredicts in the 1980s and early 1990s (**Fig. 3**). While the approaches generally agree for harbor seal
32 954 back to 1995, the random forest model underpredicts bycatch relative to the ratio estimator before 1995,
33 955 especially in the Channel Islands and Ventura strata (**Fig. S14**). For common murre, the random forest
34 956 model overpredicts bycatch relative to the ratio estimator in the mid- to late-1990s and underpredicts
35 957 relative to the ratio estimator in earlier years, especially in Morro and Monterey Bays. These
36 958 underpredictions likely occur because of the unequal impacts of lost data from the northern strata in the
37 959 1980s (**Fig. 2**). Unlike the random forest models, the ratio estimators are able to use summarized observer
38 960 data for this region and time period from old reports. As a result, the ratio estimators can learn from
39 961 observations from this region and time period while the random forest models are blind to data from this
40 962 region and period. Thus, the random forest models are likely to underpredict risk in early years in
41 963 northern strata because they largely learned from late years in southern California, where bycatch risk was
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lower. For this reason, we recommend the use of bycatch estimates from the ratio estimators over the random forest models until a time when the 1980s observer data is rediscovered.

Comparison to bycatch estimates from historical reports

Our estimates of annual bycatch are generally aligned with estimates from historical studies (**Fig. S11**). Slight differences between our estimates and those from historical studies are driven by a mixture of differences in our methods and input data. While we apply a consistent approach for defining the fishery, stratifying the data, and estimating bycatch, historical studies have employed variable fishery definitions, stratification schemes, and estimation methodologies (**Table S1**). First, we consistently defined the fishery as using $\geq 3.5"$ set gillnets, while historical studies have considered set gillnets $\geq 8"$, $\geq 8.5"$, or of unspecified sizes (**Table S1**). Furthermore, it is unclear whether historical studies filtered out fishing sets based on the reported target species and whether this decision was consistent. We do not define the fishery based on reported target species given extreme heterogeneity in how this information is reported. Differences in fishery definitions, as well as differences in data cleaning methods, likely lead to the slight differences in effort and observed bycatch attributed to the fishery (**Fig. S12**), which inevitably causes differences in bycatch estimates. Second, we used a consistent six-region stratification scheme, while historical studies used a mixture of stratification schemes ranging from no stratification to spatial stratification to spatial-temporal stratification (**Table S1**). This impacts the ratio estimators and the magnitudes of bycatch. Finally, we used vessel days (trips) as the sample unit, whereas historical studies oscillated between trips and sets as the preferred sample unit (**Table S1**). This is problematic not only because of the lack of consistency but also because sets cannot be uniquely identified in the logbook data.

Identifying set gillnet landing receipts and revenues

We used landing receipts (a.k.a., fish tickets) to estimate ex-vessel revenues generated by the California $\geq 3.5"$ set gillnet fishery from 1981-2022. Among other information, landing receipts report the date, value, species, and gear of commercial landings. We identified landing receipts associated with the $\geq 3.5"$ set gillnet fishery through a multi-step filtering process. First, we filtered the landing receipts to the five gear types that could include $\geq 3.5"$ mesh set gillnets: trammel nets, set gillnets, small-mesh set gillnets, large-mesh set gillnets, or entangling nets (**Fig. S1A**). Entangling nets, which encompass both set and drift gillnets, were a widely used gear type from 1984-1993. As a result, this filter retained many swordfish landings and other landings associated with drift gillnets. It also retained many herring landings and other landings associated with set gillnets with mesh sizes smaller than 3.5 inches. To remove landing receipts associated with drift gillnets and set gillnets with mesh sizes smaller than 3.5 inches, we

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4 995 used gillnet logbooks to identify landing receipts associated with known set gillnet vessels and logged set
5 gillnet trips. We began by further filtering to only include vessels documented as using $\geq 3.5"$ set gillnets
6 in the gillnet logbooks (**Fig. S1B**). After this filter, a large amount of swordfish and herring landings
7 remained, indicating that many $\geq 3.5"$ set gillnet vessels use other gears. Thus, to further tie landing
8 receipts with known $\geq 3.5"$ set gillnet trips, we explored four related approaches for linking landing
9 receipts to logged $\geq 3.5"$ set gillnet trips. The first approach was the most strict and only considered
10 landing receipts reported on the exact day of logged $\geq 3.5"$ set gillnet trips (**Fig. S1C**). This filter
11 eliminated swordfish and herring landings but is likely to be overly restrictive. The date of landing may
12 differ from the date of fishing because of misreporting, multi-day trips, or delayed sales. Thus, we
13 explored three progressively less restrictive rules, which attributed landing receipts recorded within one
14 (**Fig. S1D**), two (**Fig. S1E**), or three (**Fig. S1F**) days of logged $\geq 3.5"$ set gillnet trips to the fishery. We
15 selected the landing receipts associated with the 3-day buffer as the final set of landing receipts associated
16 with the fishery because it effectively eliminated landings of species not associated with the $\geq 3.5"$ set
17 gillnet fishery (i.e., swordfish and herring) while being inclusive-within-reason of potential $\geq 3.5"$ set
18 gillnet fishery landings. Finally, we adjusted daily ex-vessel landings values for inflation by converting
19 all values to January 1, 2022 US dollars using the *priceR* package in R (Condylios, 2023).
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33 1011 Mapping species ranges
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35 1012 We mapped the range of the study species using range maps from the California Wildlife Habitat
36 Relationships (CWHR) System (CDFW, 2021) (**Fig. S15**). The CWHR ranges were developed by
37 1013 species-specific experts. Range maps were not developed for harbor porpoise as part of the CWHR effort.
38 1014 We developed a range map for harbor porpoise assuming that harbor porpoise occur primarily in waters
39 shallower than 50 fathoms (92 meters) north of Point Conception (Forney et al., 2014). Harbor seal
40 1015 haulouts were mapped using the CDFW Harbor Seal Haulout GIS dataset (CDFW, 2014). CDFW
41 conducted aerial surveys of all known haulout sites in 2001, 2002, and 2003 and counted the number of
42 1016 harbor seals observed in aerial photographs of each site. We mapped northern elephant seal rookery size
43 1017 in 2010 using a database of counts developed by (Lowry et al., 2014). Counts were generated through a
44 review of ground and aerial photographic surveys. We mapped California sea lion haulouts using data
45 1018 from (Lowry, 2021). Haulouts were mapped in the Channels Islands between 2016-2019 using aerial
46 1019 photographic surveys. Sea lion haulouts occur along the California coast but were not mapped to single
47 1020 sites in this study and therefore not plotted in our range maps. We mapped seabird colonies using the
48 1021 2010 CDFW Seabird Colonies Database (CDFW, 2010). These data were collected as part of the Marine
49 1022 Life Protection Act (MLPA) planning process and report the maximum number of seabirds of 26 species
50 1023 at all known colonies.
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Supplemental Tables

Table S1. Historical bycatch estimation studies and their characteristics (the period column indicates month and year as MM/YY).

Study	Seasons	Period	Species	Fishery definition	Stratification scheme	Sample unit
(Barlow et al., 1994)	1983-87	(see papers)	Pinnipeds/cetaceans	(see included papers)	(see included papers)	(see papers)
(Hanan et al., 1988)	1983-85	Apr 1-Mar 31	Sea lion, harbor seals	Set nets for halibut/flounder/sharks	5 regions	Sets
(Diamond and Hanan, 1986)	1983	4/83 - 3/84	Harbor porpoise	$\geq 8.0"$ set nets for halibut/flounder	3 regions	Sets
(Hanan et al., 1986)	1984	4/84 - 3/85	Harbor porpoise	Set nets for halibut/flounder/sharks	3 regions	Sets
(Hanan et al., 1987)	1985	4/85 - 3/86	Harbor porpoise	Set nets (but not for croaker)	3 regions	Sets
(Hanan and Diamond, 1989)	1986	4/86 - 3/87	Sea lion, harbor seal, harbor porpoise	Set nets for halibut/flounder/sharks	5 regions, seasons	Sets
(Konno, 1990)	1987	4/87 - 3/88	Sea lion, harbor seal, harbor porpoise	Set nets for halibut/flounder/sharks	5 regions, seasons	Sets
(Perkins et al., 1994)	1988-90	4/88 - 3/89 4/89 - 12/89 1/90 - 12/90	Pinnipeds/cetaceans	Set nets for halibut/angel shark	3 regions	Vessel-day
(Lennert et al., 1994)	1990	7/90 - 12/90	Marine mammals	Set nets for halibut/angel shark	3 regions	Vessel-day
(Perkins et al., 1992a)	1990	7/90 - 6/91	Pinnipeds/cetaceans	$\geq 8.0"$ nets for halibut/angel shark	3 regions	Sets
(Perkins et al., 1992b)	1991	1/91 - 12/91	Pinnipeds/cetaceans	Set nets for halibut/angel shark	SCal/Ventura: Quarters Other strata: Annual	Vessel-day
(Julian, 1993)	1992	1/92 - 12/92	Pinnipeds/cetaceans	Set nets for halibut/angel shark	4 regions: quarterly	Vessel-day
(Julian, 1994)	1993	1/93 - 12/93	Pinnipeds/cetaceans	Set nets for halibut	4 regions	Vessel-day
(Julian and Beeson, 1998)	1990-95	7/90-12/90 1-12, 91-95	Mammals/seabirds/turtles	$\geq 8.5"$ nets for halibut/angel shark	SCal/Ventura: Quarters Other strata: Annual	Vessel-day
(Cameron and Forney, 1999)	1997-98	1/97 - 12/97 1/98 - 12/98	Cetaceans	$\geq 8.5"$ nets for halibut/angel shark	1997: Geographical+seasonal 1998: Geographical only	Vessel-day
(Cameron and Forney, 2000)	1999	1/99 - 12/99	Cetaceans	$\geq 8.5"$ nets for halibut/angel shark	Geographical+seasonal	Vessel-day
(Carretta, 2001)	2000	1/00 - 12/00	Cetaceans	$\geq 8.5"$ nets for halibut/angel shark	SCal/Ventura: Quarters Other strata: Annual	Vessel-day
(Carretta, 2002)	2001	1/01 - 12/01	Cetaceans	Set nets for halibut/angel shark	SCal/Ventura: Quarters Other strata: Annual	Vessel-day
(Carretta and Chivers, 2003)	2002	1/02 - 12/02	Cetaceans	Set nets for halibut/angel shark	SCal/Ventura: Quarters Other regions: Annual	Vessel-day
(Carretta and Chivers, 2004)	2003	1/03 - 12/03	Marine mammals	$\geq 8.5"$ nets for halibut/angel shark	SCal/Ventura: Quarters Other regions: Annual	Vessel-day
(Carretta and Enriquez, 2009)	2007	1/07 - 12/07	Mammals/seabirds	Set nets for halibut/white seabass	No stratification	Sets
(Carretta and Enriquez, 2012a)	2010	1/10 - 12/10	Mammals/seabirds	Set nets for halibut/white seabass	No stratification	Vessel-day
(Carretta and Enriquez, 2012b)	2011	1/11 - 12/11	Mammals/seabirds	Set nets for halibut/white seabass	No stratification	Sets
(Carretta et al., 2014)	2012	1/12 - 12/12	Mammals/seabirds/turtles	Set nets for halibut/white seabass	No stratification	Vessel-day

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21032**Table S2.** Number of fishing trips in the California 3.5 inch mesh set gillnet fishery by year.

Year	# of vessels	# of trips		Percent of trips	
		Total	Observed	Unobserved	Observed
1981	162	6139	0	6139	0.0%
1982	222	9218	0	9218	0.0%
1983	262	9012	143	8869	1.6%
1984	357	12374	255	12119	2.1%
1985	378	14314	238	14076	1.7%
1986	379	13336	217	13119	1.6%
1987	290	10667	213	10454	2.0%
1988	211	8585	120	8465	1.4%
1989	198	7811	58	7753	0.7%
1990	182	6836	167	6669	2.4%
1991	158	6668	716	5952	10.7%
1992	117	5611	744	4867	13.3%
1993	115	5754	959	4795	16.7%
1994	74	2455	150	2305	6.1%
1995	70	2616	0	2616	0.0%
1996	59	2654	0	2654	0.0%
1997	66	3310	0	3310	0.0%
1998	59	2889	0	2889	0.0%
1999	63	4026	165	3861	4.1%
2000	66	3828	66	3762	1.7%
2001	63	3289	0	3289	0.0%
2002	54	3395	0	3395	0.0%
2003	51	2779	0	2779	0.0%
2004	47	2627	0	2627	0.0%
2005	40	1930	0	1930	0.0%
2006	41	1658	5	1653	0.3%
2007	44	1797	65	1732	3.6%
2008	43	1936	0	1936	0.0%
2009	41	1934	0	1934	0.0%
2010	33	1544	64	1480	4.1%
2011	39	1575	55	1520	3.5%
2012	35	1374	76	1298	5.5%
2013	32	968	48	920	5.0%
2014	25	819	0	819	0.0%
2015	25	1014	0	1014	0.0%
2016	24	1077	0	1077	0.0%
2017	21	840	69	771	8.2%
2018	19	948	0	948	0.0%
2019	18	1151	0	1151	0.0%
2020	18	841	0	841	0.0%
2021	16	870	0	870	0.0%

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Table S3. Bycatch rates extracted from historical reports (* mark years with raw observer data where the summary values from the historical reports are not needed).

Reference	Species	Region	Year	# sets		Estimated # trips		Catch per trip	Raw data available?
				observed	observed	Catch			
Hanan et al. 1988	California sea lion	Channel Islands	1985	180	60.00	44	0.733	*	
Hanan & Diamond 1989	California sea lion	Channel Islands	1986	66	22.00	54	2.455	*	
Hanan et al. 1988	California sea lion	Monterey Bay	1983	22	7.33	12	1.636		
Hanan et al. 1988	California sea lion	Monterey Bay	1984	126	42.00	19	0.452		
Hanan et al. 1988	California sea lion	Monterey Bay	1985	49	16.33	5	0.306		
Hanan & Diamond 1989	California sea lion	Monterey Bay	1986	36	12.00	4	0.333		
Hanan et al. 1988	California sea lion	Morro Bay	1983	288	96.00	41	0.427		
Hanan et al. 1988	California sea lion	Morro Bay	1984	374	124.67	22	0.176		
Hanan et al. 1988	California sea lion	Morro Bay	1985	317	105.67	25	0.237		
Hanan & Diamond 1989	California sea lion	Morro Bay	1986	137	45.67	13	0.285	*	
Hanan et al. 1988	California sea lion	San Francisco	1983	158	52.67	4	0.076		
Hanan et al. 1988	California sea lion	San Francisco	1984	300	100.00	8	0.080		
Hanan et al. 1988	California sea lion	San Francisco	1985	348	116.00	3	0.026		
Hanan & Diamond 1989	California sea lion	San Francisco	1986	419	139.67	2	0.014		
Hanan et al. 1988	California sea lion	Southern California	1983	430	143.33	16	0.112	*	
Hanan et al. 1988	California sea lion	Southern California	1984	571	190.33	13	0.068	*	
Hanan et al. 1988	California sea lion	Southern California	1985	339	113.00	5	0.044	*	
Hanan & Diamond 1989	California sea lion	Southern California	1986	425	141.67	15	0.106	*	
Hanan et al. 1988	California sea lion	Ventura	1983	430	143.33	16	0.112	*	
Hanan et al. 1988	California sea lion	Ventura	1984	571	190.33	13	0.068	*	
Hanan et al. 1988	California sea lion	Ventura	1985	339	113.00	5	0.044	*	
Hanan & Diamond 1989	California sea lion	Ventura	1986	425	141.67	15	0.106	*	
Diamond & Hanan 1986	Harbor porpoise	Monterey Bay	1983	22	7.33	2	0.273		
Hanan et al. 1986	Harbor porpoise	Monterey Bay	1984	126	42.00	2	0.048		
Hanan et al. 1987	Harbor porpoise	Monterey Bay	1985	49	16.33	2	0.122		
Hanan & Diamond 1989	Harbor porpoise	Monterey Bay	1986	36	12.00	1	0.083		
Diamond & Hanan 1986	Harbor porpoise	Morro Bay	1983	288	96.00	7	0.073		
Hanan et al. 1986	Harbor porpoise	Morro Bay	1984	374	124.67	3	0.024		
Hanan et al. 1987	Harbor porpoise	Morro Bay	1985	317	105.67	3	0.028		
Hanan & Diamond 1989	Harbor porpoise	Morro Bay	1986	137	45.67	3	0.066	*	
Diamond & Hanan 1986	Harbor porpoise	San Francisco	1983	151	50.33	5	0.099		
Hanan et al. 1986	Harbor porpoise	San Francisco	1984	299	99.67	14	0.140		
Hanan et al. 1987	Harbor porpoise	San Francisco	1985	348	116.00	28	0.241		
Hanan & Diamond 1989	Harbor porpoise	San Francisco	1986	419	139.67	12	0.086		
Hanan et al. 1988	Harbor seal	Channel Islands	1985	180	60.00	13	0.217	*	
Hanan et al. 1988	Harbor seal	Monterey Bay	1983	22	7.33	0	0.000		
Hanan et al. 1988	Harbor seal	Monterey Bay	1984	126	42.00	10	0.238		
Hanan et al. 1988	Harbor seal	Monterey Bay	1985	49	16.33	7	0.429		
Hanan & Diamond 1989	Harbor seal	Monterey Bay	1986	36	12.00	9	0.750		
Hanan et al. 1988	Harbor seal	Morro Bay	1983	288	96.00	17	0.177		
Hanan et al. 1988	Harbor seal	Morro Bay	1984	374	124.67	29	0.233		
Hanan et al. 1988	Harbor seal	Morro Bay	1985	317	105.67	84	0.795		
Hanan & Diamond 1989	Harbor seal	Morro Bay	1986	137	45.67	25	0.547	*	
Hanan et al. 1988	Harbor seal	San Francisco	1983	158	52.67	11	0.209		
Hanan et al. 1988	Harbor seal	San Francisco	1984	300	100.00	22	0.220		
Hanan et al. 1988	Harbor seal	San Francisco	1985	348	116.00	38	0.328		
Hanan & Diamond 1989	Harbor seal	San Francisco	1986	419	139.67	60	0.430		
Hanan et al. 1988	Harbor seal	Southern California	1983	430	143.33	0	0.000	*	
Hanan et al. 1988	Harbor seal	Southern California	1984	571	190.33	4	0.021	*	
Hanan et al. 1988	Harbor seal	Southern California	1985	339	113.00	2	0.018	*	
Hanan & Diamond 1989	Harbor seal	Southern California	1986	425	141.67	8	0.056	*	
Hanan et al. 1988	Harbor seal	Ventura	1983	430	143.33	0	0.000	*	
Hanan et al. 1988	Harbor seal	Ventura	1984	571	190.33	4	0.021	*	
Hanan et al. 1988	Harbor seal	Ventura	1985	339	113.00	2	0.018	*	
Hanan & Diamond 1989	Harbor seal	Ventura	1986	425	141.67	8	0.056	*	

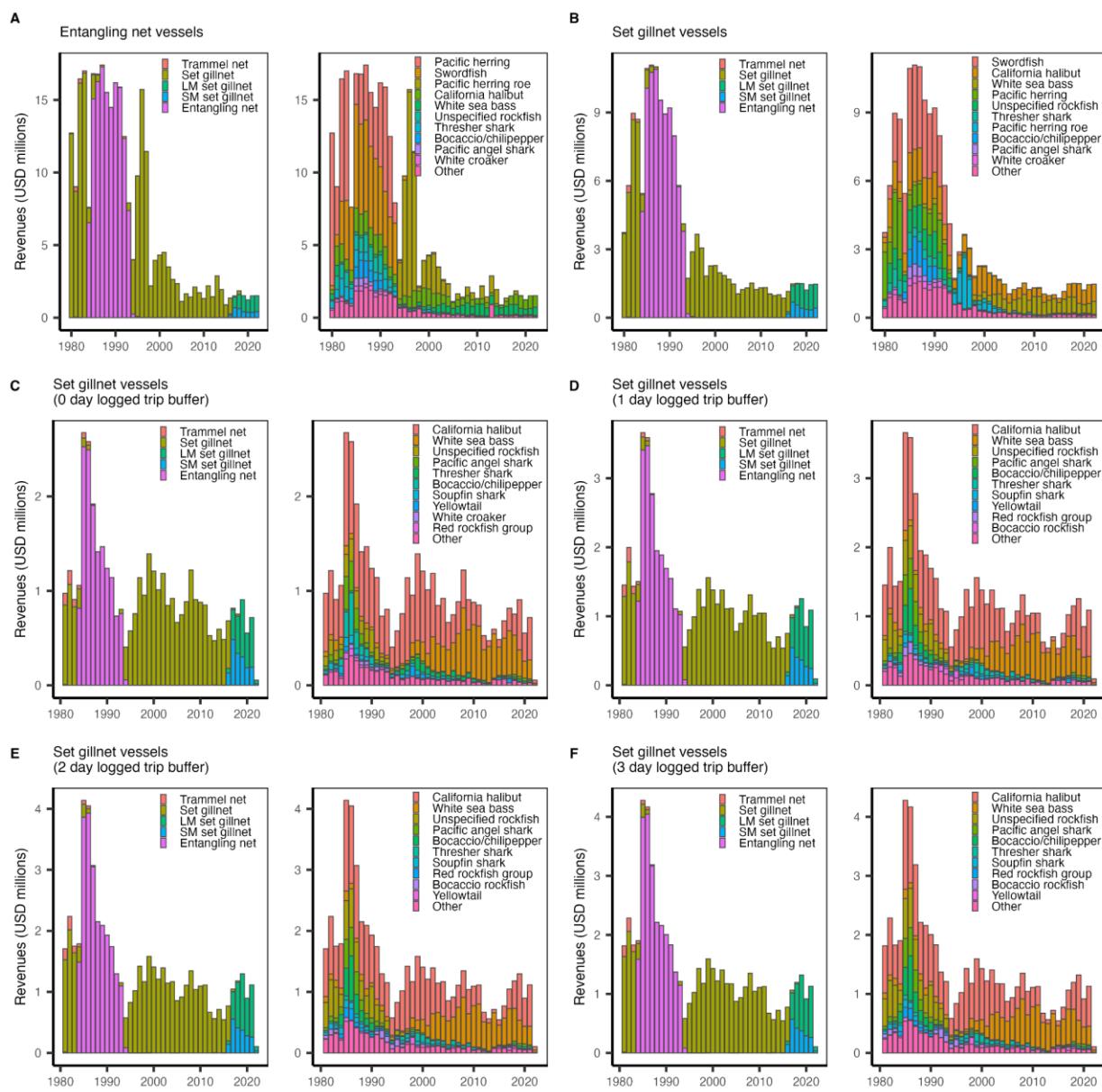
Table S4. Assumptions made throughout the analysis, their likelihood, and their potential impact on the results (*assumptions that are expected to be valid or true on average are not likely to impact the results).

Assumption	Likelihood and potential impact of assumption
Data analysis	
Ratio estimation	
1. Ratio estimation assumes that the rate of bycatch for observed fishing trips equals the rate for all fishing trips in a given stratum.	
2. We assumed that the bycatch rate in years without observer data was equal to the bycatch rate in the closest year with data.	
Bycatch estimation using the random for models*	
1. We assumed that a “pseudo-set” (roughly equivalent to a trip) is equivalent to 3 sets (Fig. S4AB).	
2. We assumed the number of captures per set is the median number of captures when a capture occurs (Fig. S4C).	
Data imputation	
Observer data	
1. We assigned missing GPS coordinates using the median coordinates for observed trips within the statistical block most frequently visited by the vessel – in order of preference – that week, month, or year based on the logbook data.	
2. We derived missing fishing depths by extracting depths from 25-meter resolution bathymetry data (Fig. S3B).	
3. We reassigned missing soak hours the mode value for a vessel and target species.	
4. We reassigned missing mesh sizes the mode for – in order of preference – the vessel and target species, the target species, or all vessels (Fig. S3DE).	
5. We assigned each GPS coordinate to the nearest statistical reporting block, which allows points erroneously falling on land to be assigned a likely statistical block.	
Logbook data	
1. We reassigned both missing and unrealistic fishing depths, defined as depths exceeding the maximum depth in the reported fishing block, the median depth of the fishing block (Fig. S5B).	
2. We reassigned missing soak hours the mode value for a vessel.	
3. We capped unlikely soak times exceeding 96 hours at 96 hours.	

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4. We reassigned missing and unrealistic (>20 inch) mesh sizes using a hierarchical procedure (**Fig. S5DE**). For logbooks with both vessel id and target species info, we assigned the mesh size most commonly used by the vessel when targeting that target species. For logbooks with only target species info (no vessel id), we assigned the mesh size most commonly used when targeting that target species across all vessels (**Figs. S5 & S6**).

1040
1041 *** This analysis is informational and only included in the supplemental information.*

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47
48 1044 **Fig. S1.** Estimated ex-vessel revenues generated by the California $\geq 3.5"$ set gillnet fishery as estimated
49 through six different filtration procedures. The filtration procedures examine the sum annual ex-vessel
50 revenues reported on landing receipts from (A) vessels using various reported entangling net gears; (B)
51 vessels using various entangling net gears that are known to use set gillnets based on logbooks; and (C-F)
52 vessels known to use set gillnets based on logbooks that are dated within various buffers of a logged set
53 gillnet trip. We adopted the final filter, which sums landing receipts date within 3 days of logged set
54 gillnet trip, as the best estimate of ex-vessel revenues for the fishery. Revenues have not been adjusted for
55 inflation (see Fig. 1D for the inflation adjusted ex-vessel revenues).
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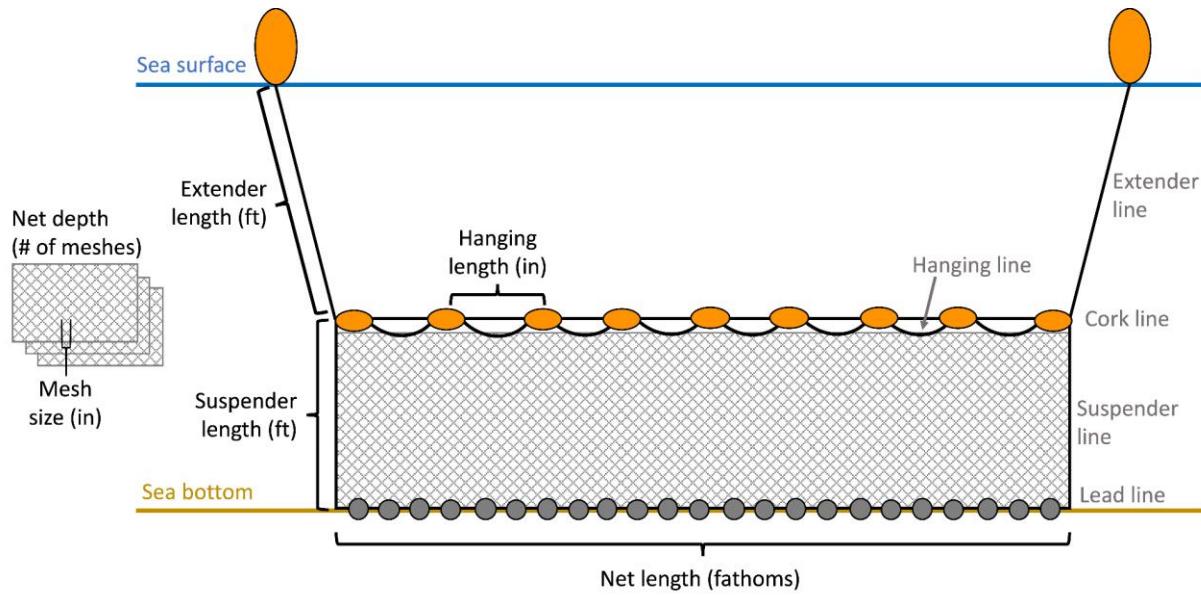


Fig. S2. Diagram of a typical California set gillnet illustrating the measurements reported in the logbook and observer data. Grey text indicates parts of the gillnet and black text indicates the reported measurements. The material, strength, and color of the net is also reported. Finally, the hanging ratio, a percentage that is calculated as the length of the mesh web divided by the length of the cork line, is reported.

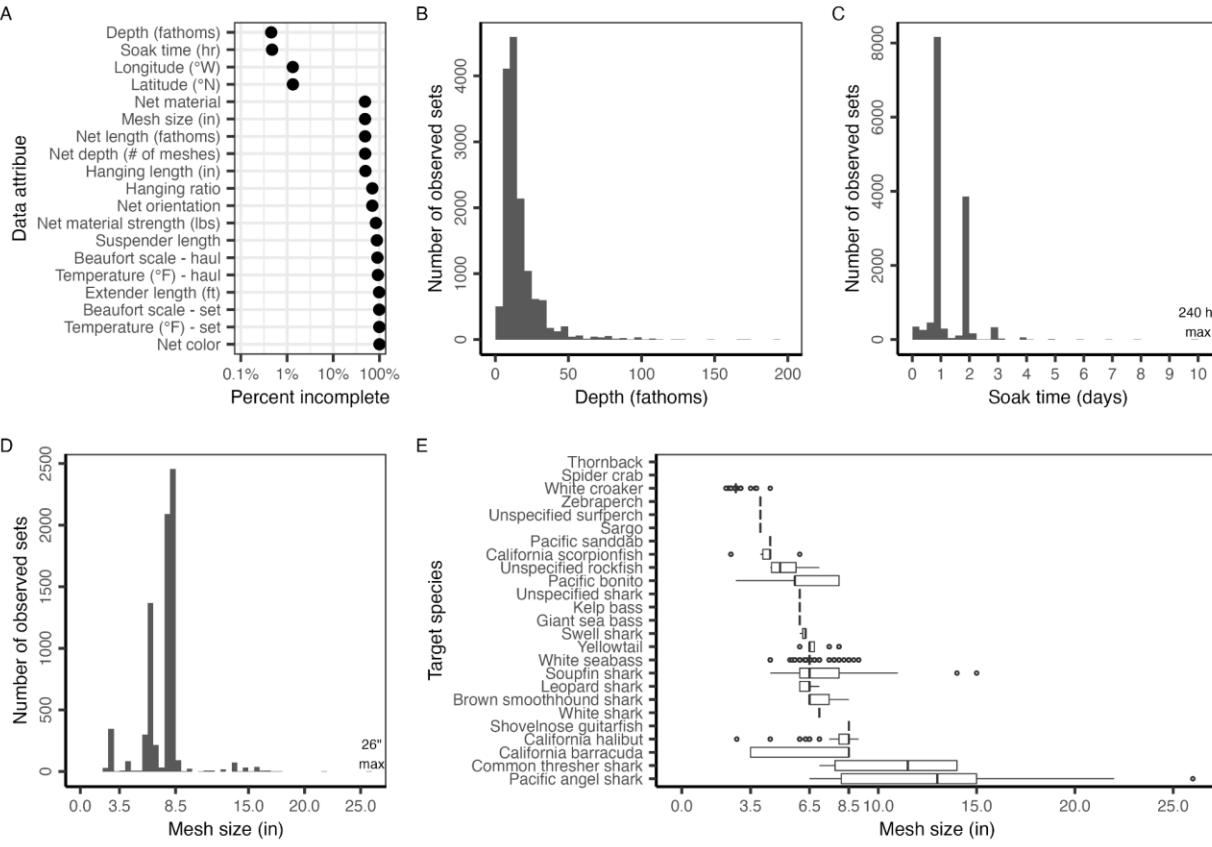
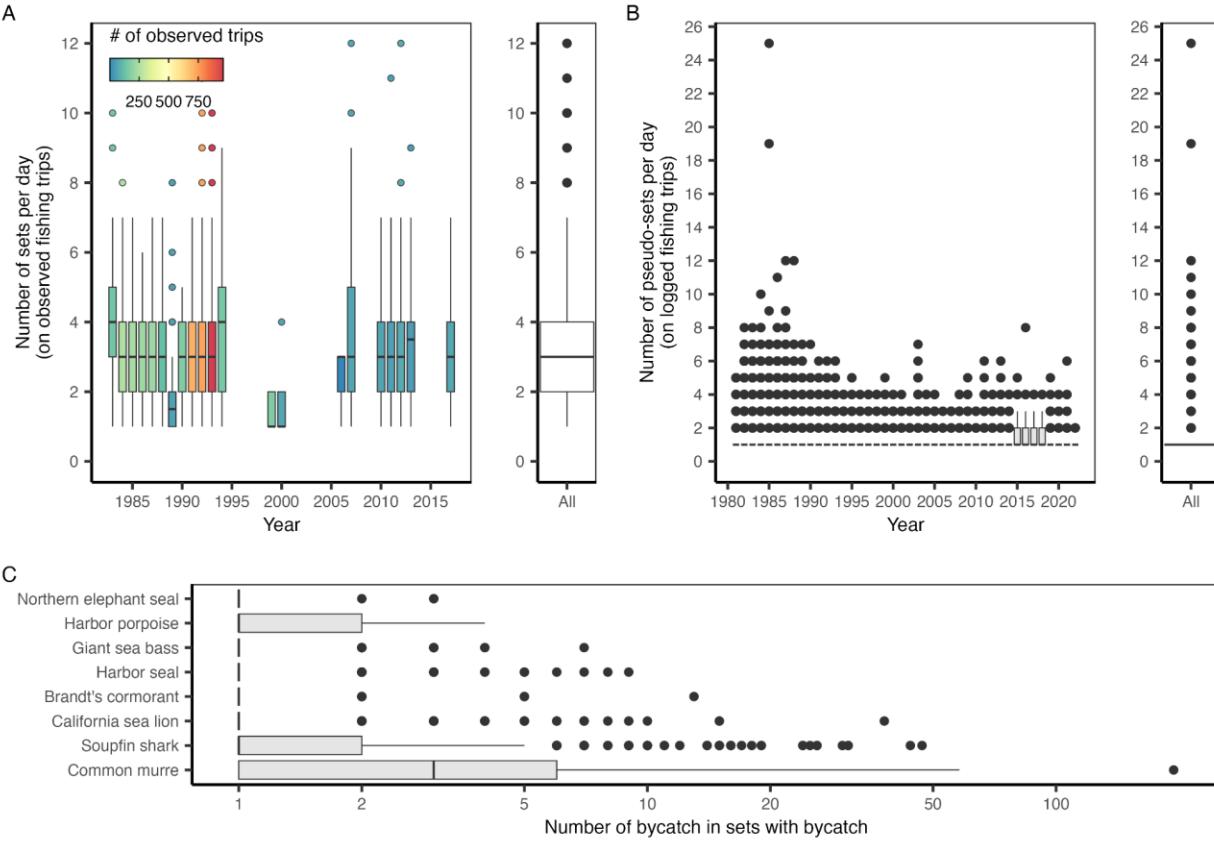


Fig. S3. Traits of the observed set gillnet metadata. Panel **A** shows the level of completeness of gillnet metadata. Panel **B** shows the distribution of reported depths. Panel **C** shows the distribution of reported soak times; the maximum reported soak time is 240 hours (10 days). Panel **D** shows the distribution of reported mesh sizes; the maximum reported mesh size is 26 inches. Panel **E** shows the distribution of reported mesh sizes by reported target species. In the boxplots, the solid line indicates the median, the box indicates the interquartile range (IQR; 25th to 75th percentiles), the whiskers indicate 1.5 times the IQR, and points indicate outliers.



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331067 **Fig. S4.** The (A) number of sets per observed fishing trip by year and overall; (B) number of pseudo-sets
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351068 per logged fishing trip by year and overall; and (C) number of bycatch in sets with bycatch of each of the
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371069 species of interest. In the boxplots, the solid line indicates the median, the box indicates the interquartile
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391070 range (IQR; 25th to 75th percentiles), the whiskers indicate 1.5 times the IQR, and points indicate
401071 outliers. In (A), the fill color indicates the number of observed fishing trips contributing to the annual
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421072 distribution.

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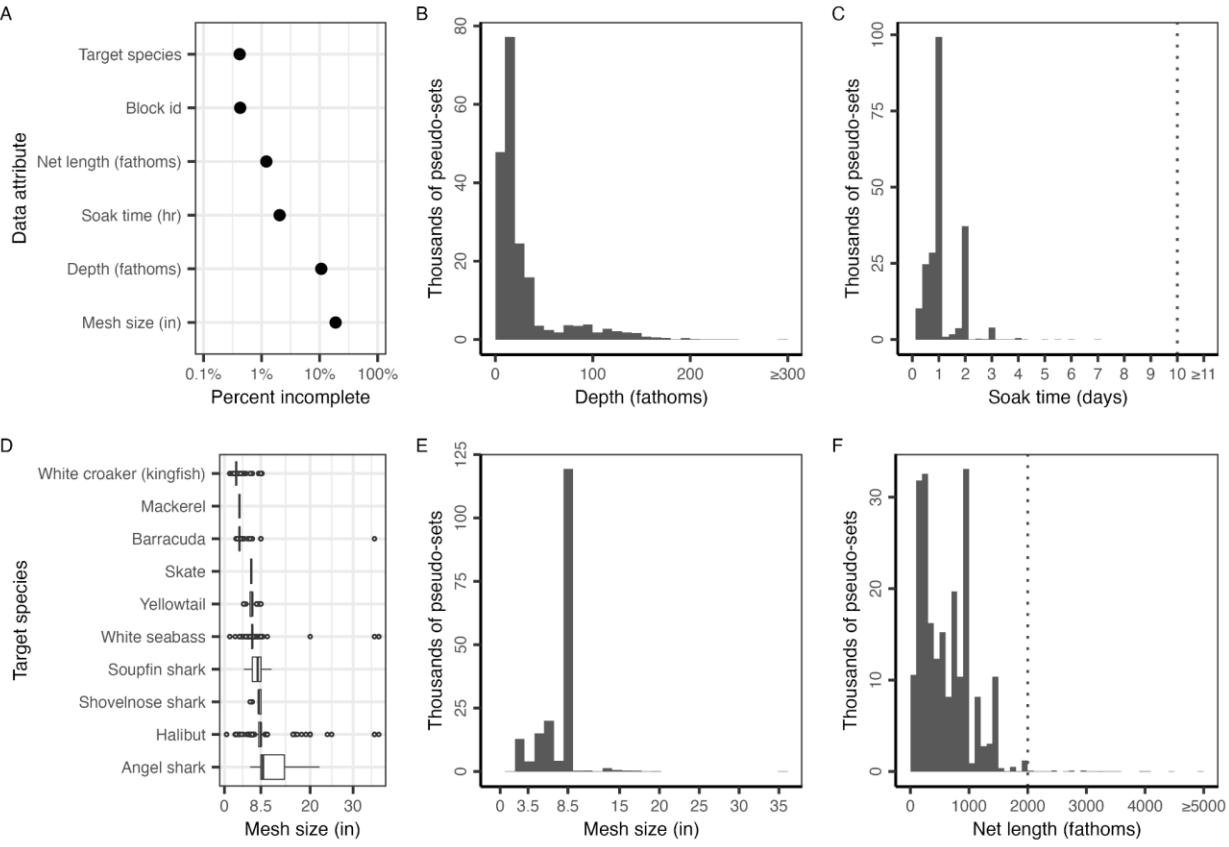


Fig. S5. Traits of the set gillnet sets documented in the logbook data. Panel **A** shows the level of completeness of gillnet logbook metadata. Panel **B** shows the distribution of reported depths. Panel **C** shows the distribution of reported soak times. The maximum reported soak time in the observer data is 10 days; rare values larger than this value were assumed to be unrealistic and were capped at the maximum. Panel **D** shows the distribution of reported mesh sizes by reported target species. In the boxplots, the solid line indicates the median, the box indicates the interquartile range (IQR; 25th to 75th percentiles), the whiskers indicate 1.5 times the IQR, and points indicate outliers. Panel **E** shows the distribution of reported mesh sizes. Panel **F** shows the distribution of reported net lengths.

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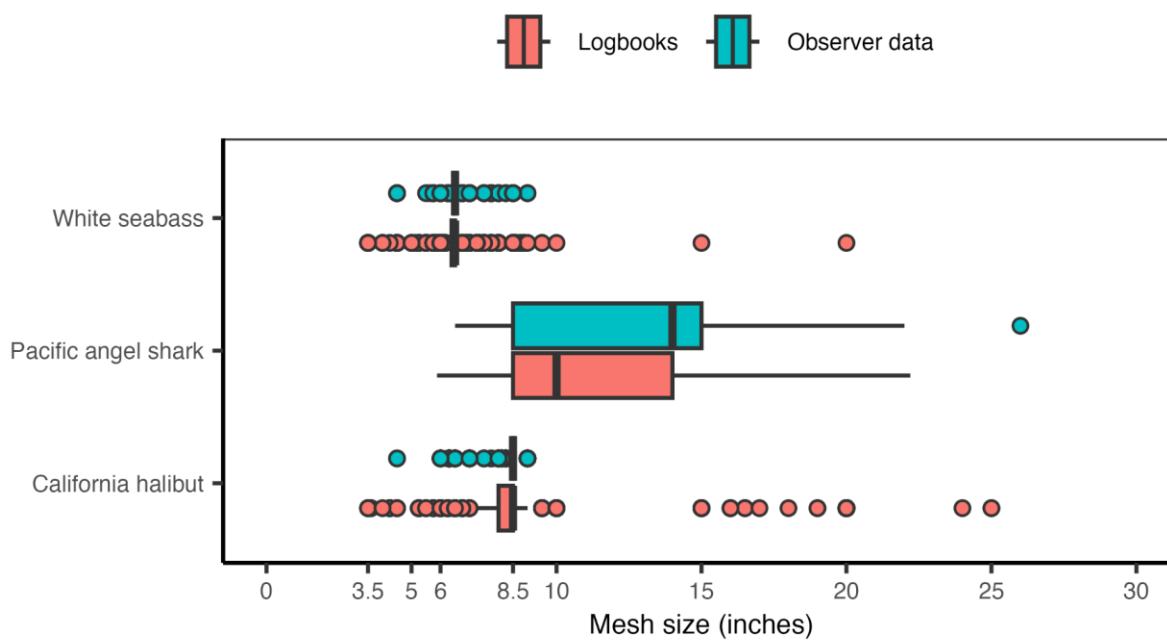
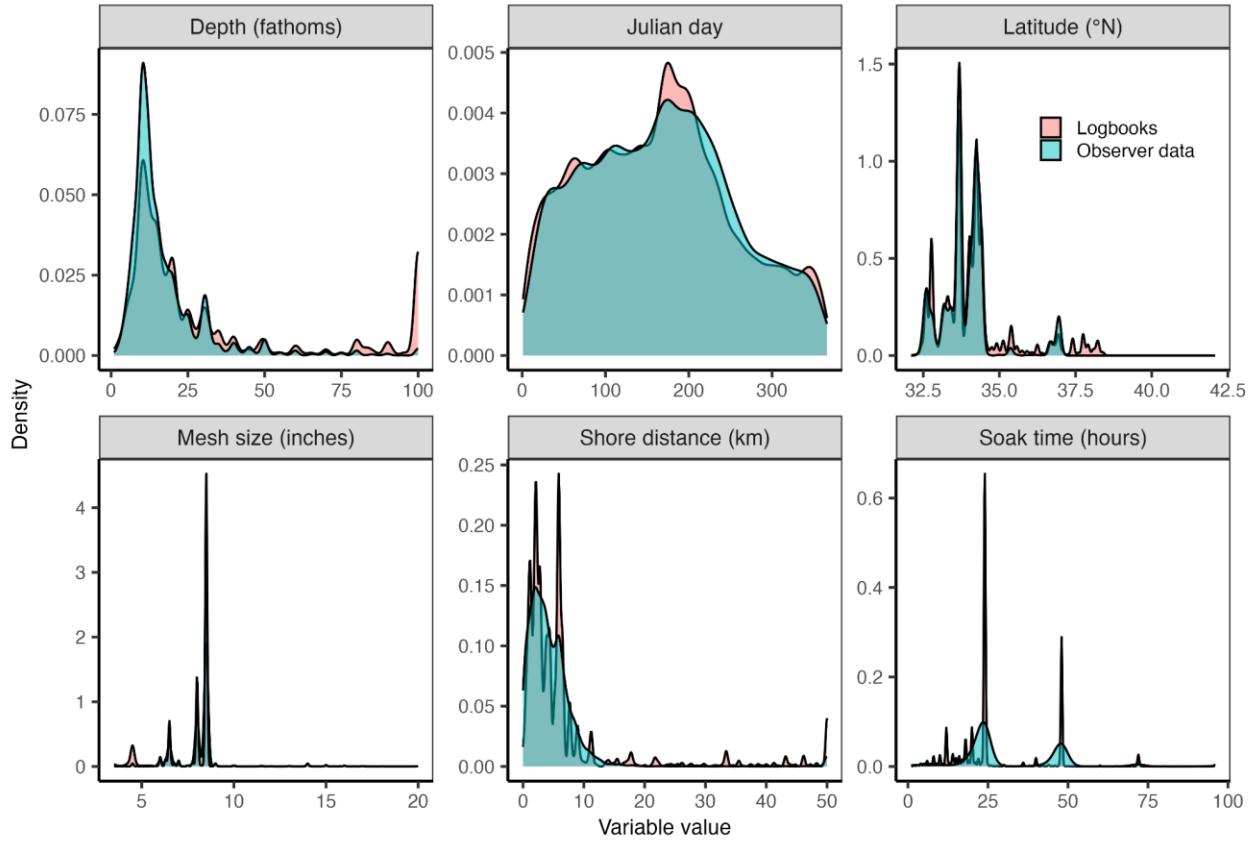


Fig. S6. Mesh size (inches) by target species in the logbook (red) and observer (blue) data. Since 1989, California halibut and Pacific angel shark can only be targeted using mesh sizes larger than 8.5 inches. White seabass are typically targeted using a minimum mesh size of 6.0 inches; however, a small amount of incidental take (<20% of catch and ≤ 10 individuals) in mesh sizes between 3.5 to 6.0 inches is allowed from June 16 to March 14 (14, § 155.10).



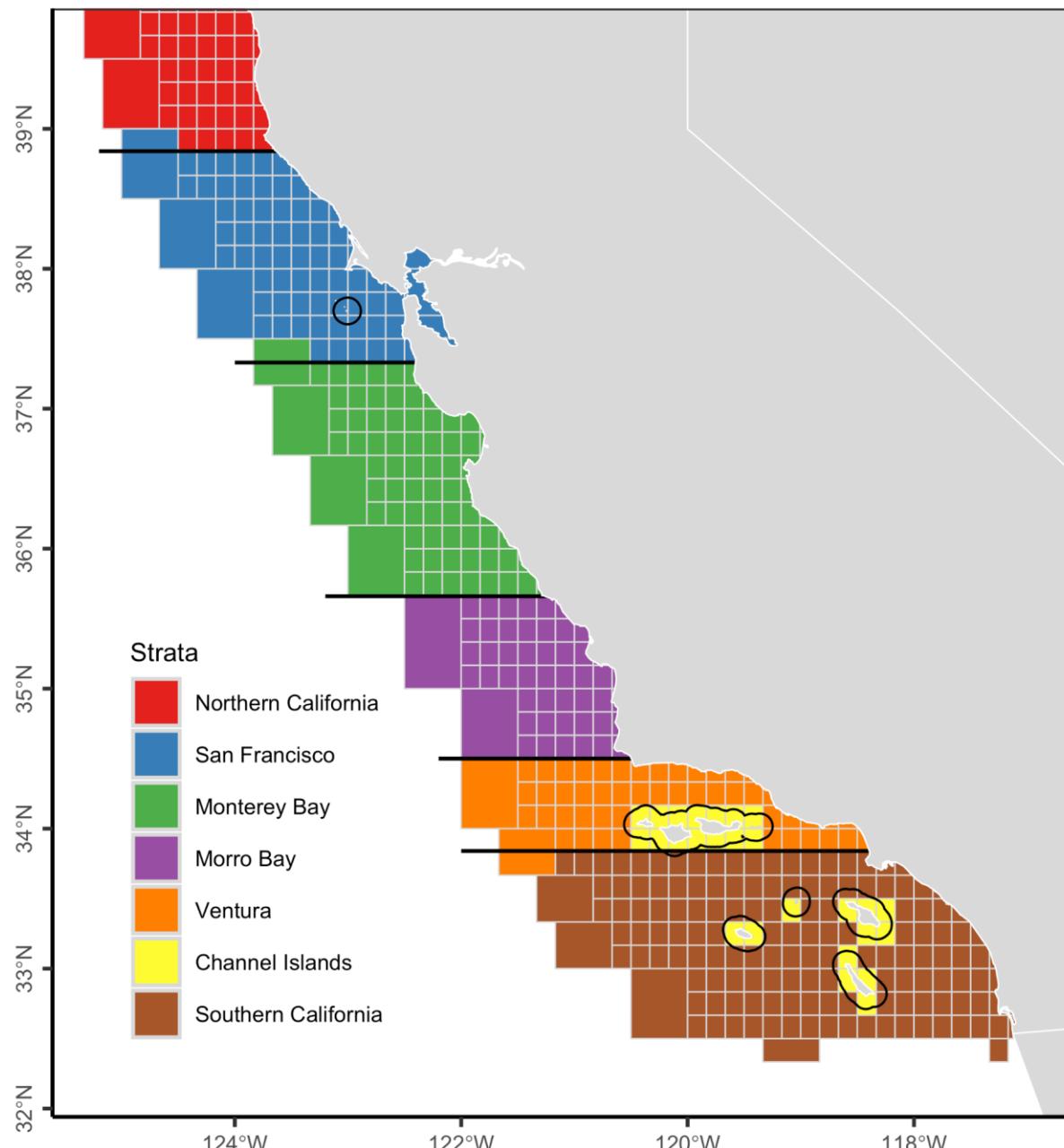


Fig. S8. The regional stratification scheme used throughout the analysis. The stratification scheme north of Point Conception was originally proposed by Diamond and Hanan (1986). The stratification scheme south of Point Conception was originally proposed by Julian (1993). The dark black lines around the Channel Islands show the 10 km buffer used to identify island-associated fishing trips.

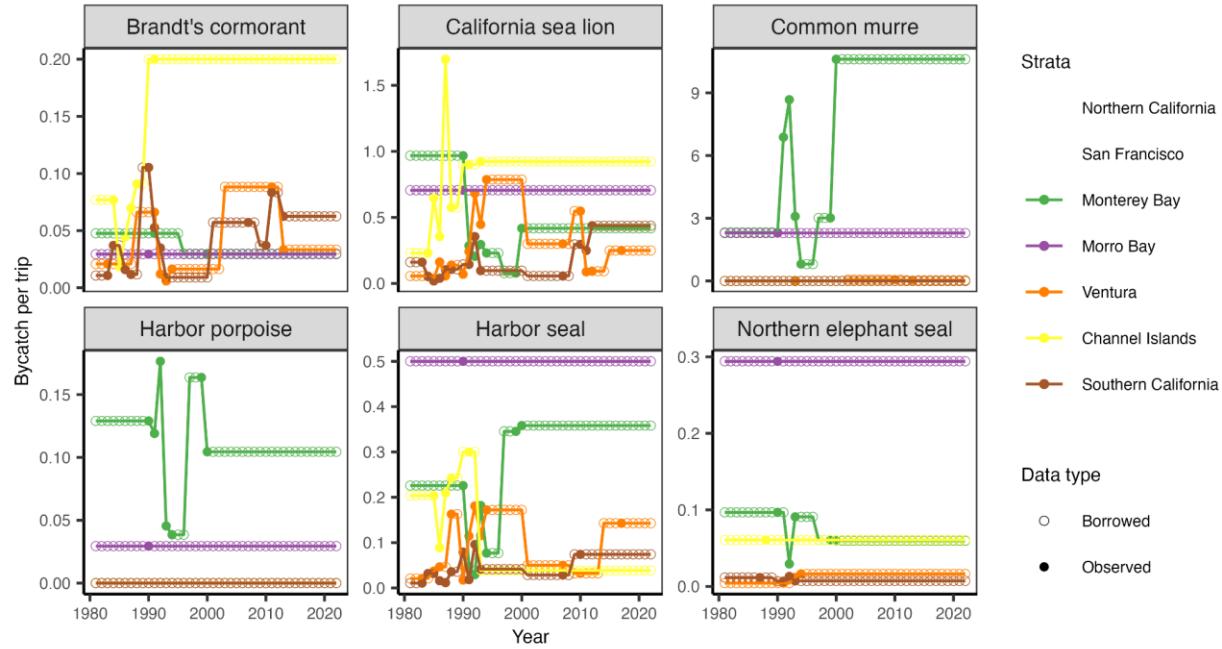
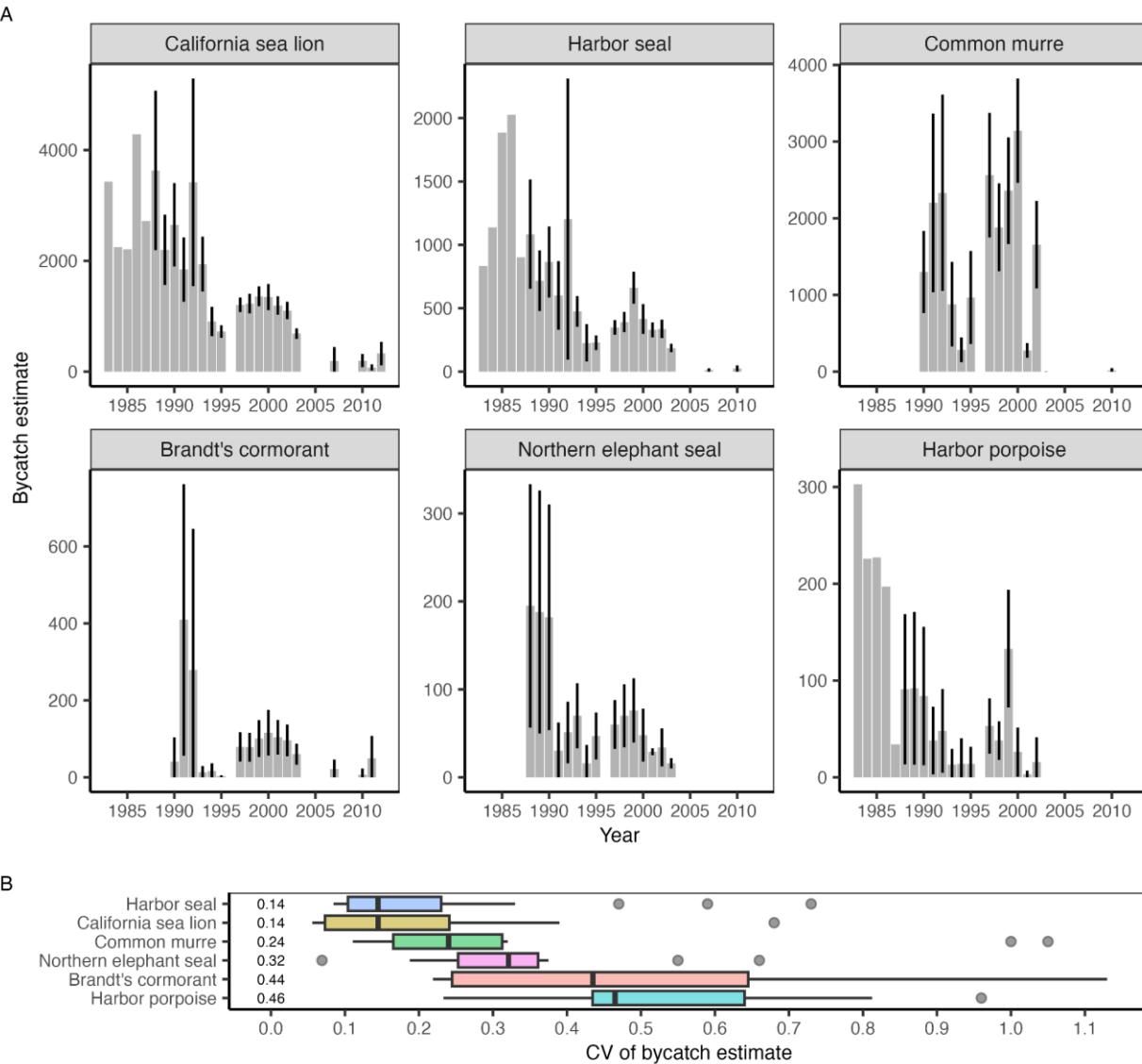


Fig. S9. Annual bycatch rates by species and regional strata as assumed in the ratio estimation analysis. Solid circles indicate years with observer data and open circles indicate years whose bycatch rates are borrowed from the closest year with observer data. See **Figs. 1A and S8** for maps of the regional strata.



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1103 **Fig. S10.** Estimates of (A) annual bycatch from historical studies and (B) the uncertainty of these
1104 estimates expressed as the coefficient of variation (CV). In (A), error bars indicate 95% confidence
1105 intervals. See **Table S1** for the sources of these estimates. In (B), the median CV of the bycatch estimates
1106 for each species is printed on the far left. In the boxplots, the solid line indicates the median, the box
1107 indicates the interquartile range (IQR; 25th to 75th percentiles), the whiskers indicate 1.5 times the IQR,
1108 and points indicate outliers.

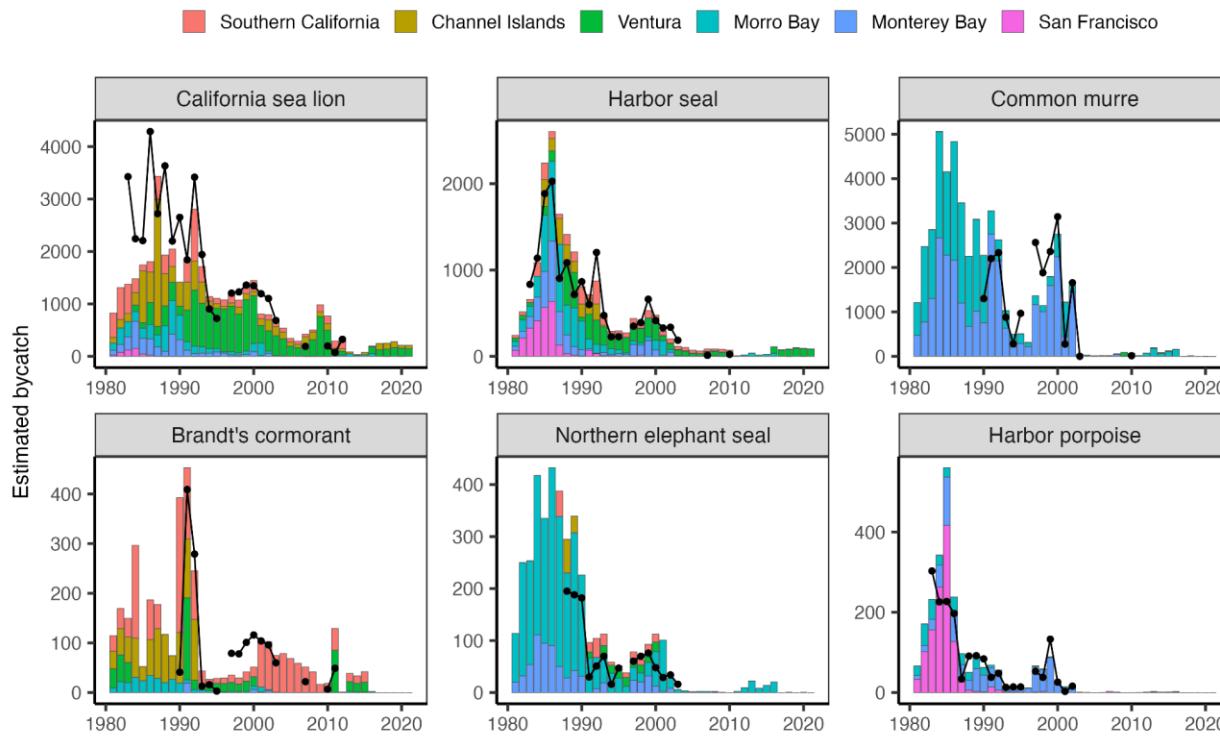


Fig. S11. A comparison of estimates of annual bycatch from our study (bars) and historical studies (points and lines). Potential reasons for these differences are explored in Fig. S12.

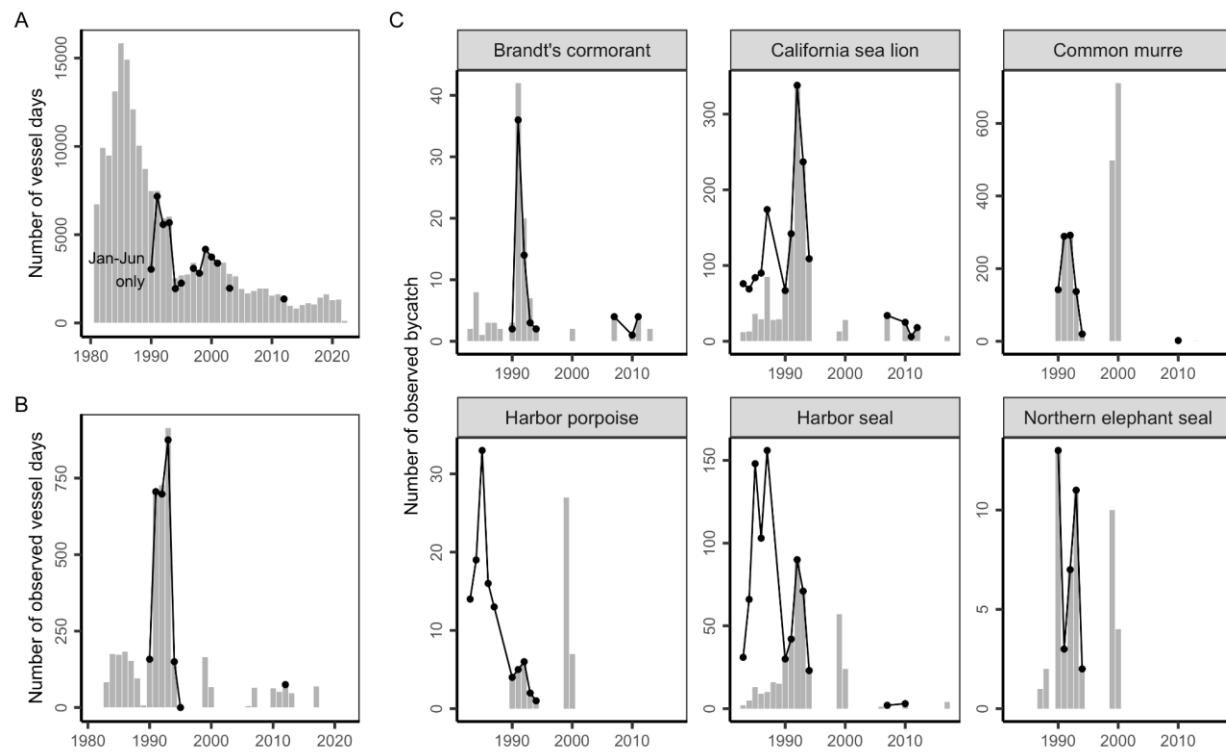


Fig. S12. A comparison of historical (A) fishing effort (number of vessel days); (B) observer coverage (number of observed vessel days); and (C) observer records (number of observed bycatch) derived in our analysis (bars) and reported in historical studies (points and lines). These time series represent the key inputs into the ratio estimation analysis and help to explain the differences between the estimates of annual bycatch derived in our study as compared to historical studies (see **Fig. S11**). See **Table S1** for additional details on historical studies.

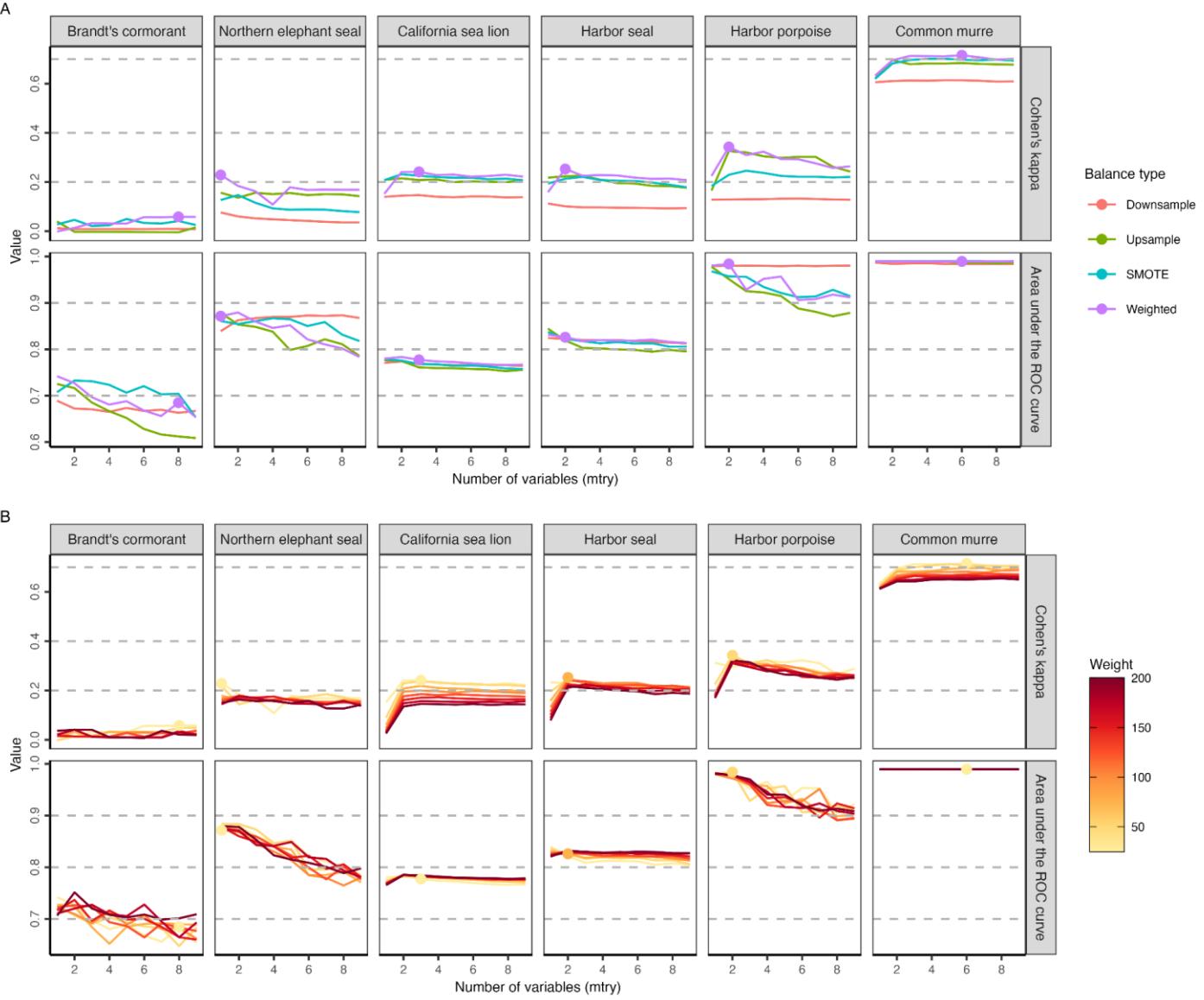


Fig. S13. A comparison of the model performance as measured through cross-validation on the training dataset between (A) balanced random forest modeling approaches and (B) weighted random forest modeling approaches for all candidate bycatch species (i.e., observer records > 50). We evaluated the model performance using Cohen's kappa and area under the receiver operator curve (ROC) and selected the model with the highest Cohen's kappa as the best performing model (labeled in the plot). We excluded Brandt's cormorant in the model prediction as their Cohen's kappa (<0.1) is too low to produce reliable predictions.

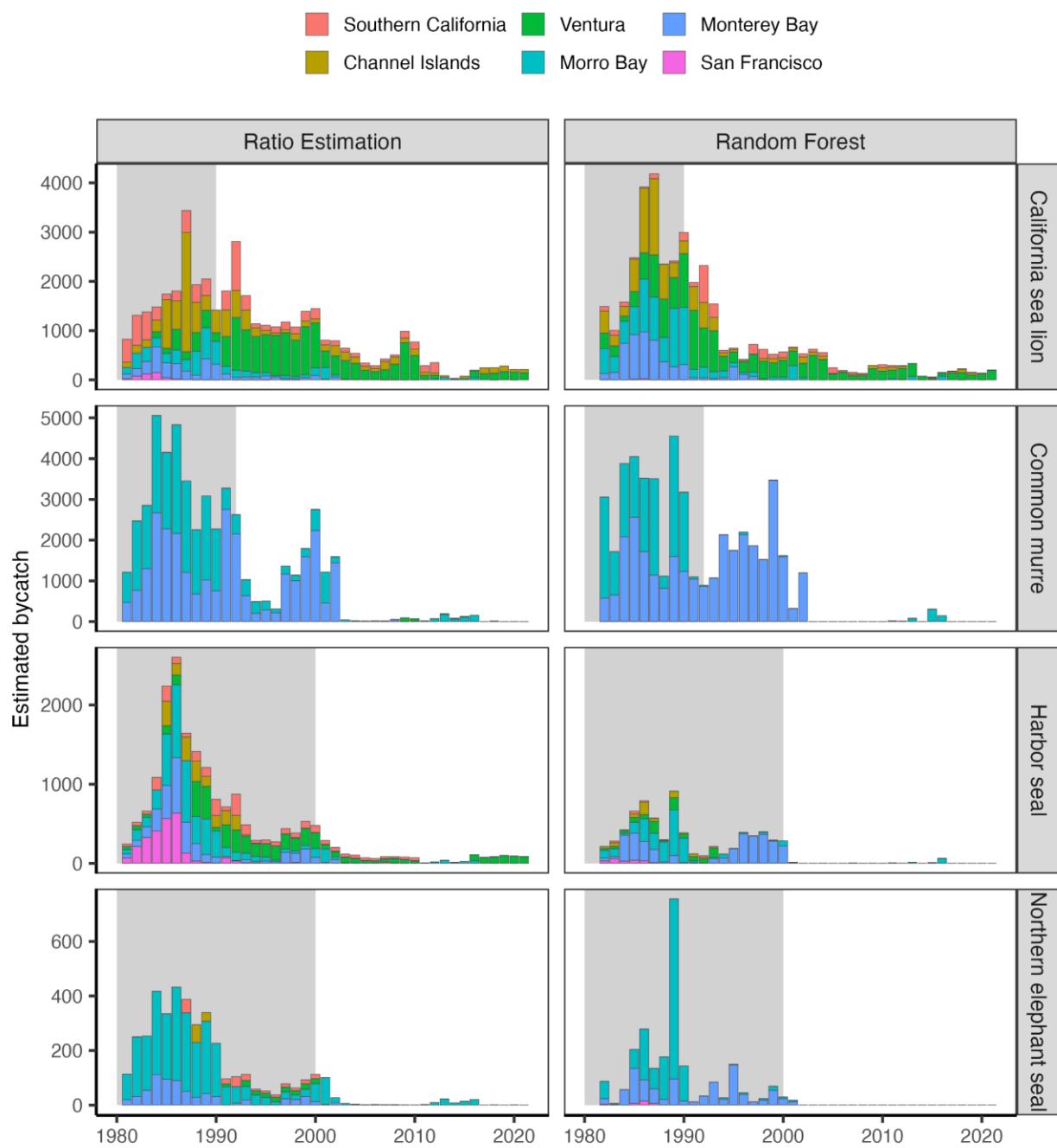


Fig. S14. A comparison of estimated bycatch numbers between ratio estimation and random forest stratified by regions (Fig. S8). The major differences between years are highlighted in gray.

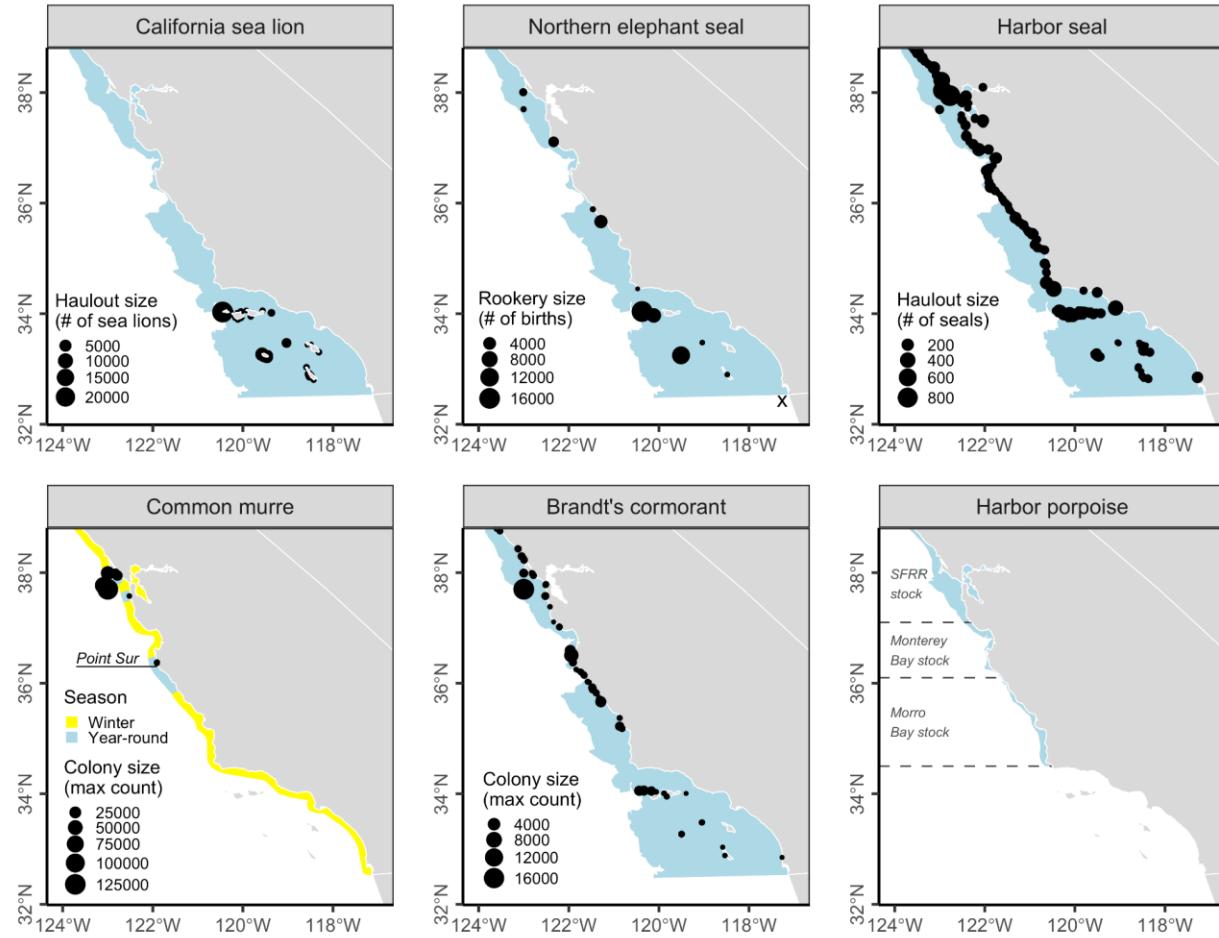


Fig. S15. Ranges of the six study species and information on haulout and colony size for selected species.

Blue colors indicate year-round ranges and yellow colors (common murre only) indicate winter ranges.

The range maps for all species except harbor porpoise are from the California Wildlife Habitat Relationship System (CDFW, 2021). Harbor porpoises occur in waters less than 50 fathoms (92 meters) deep north of Point Conception (Forney et al., 2014). SFRR indicates the San Francisco-Russian River harbor porpoise stock. Common murre and Brandt's cormorant colony counts are from the California Seabird Colony Database (CDFW, 2010). There are no historical breeding records for common murre south of Point Sur except for at the Prince Island colony in Cuyler Harbor of San Miguel Island. Harbor seal haulout counts are from the CDFW Harbor Seal GIS dataset (CDFW, 2014).

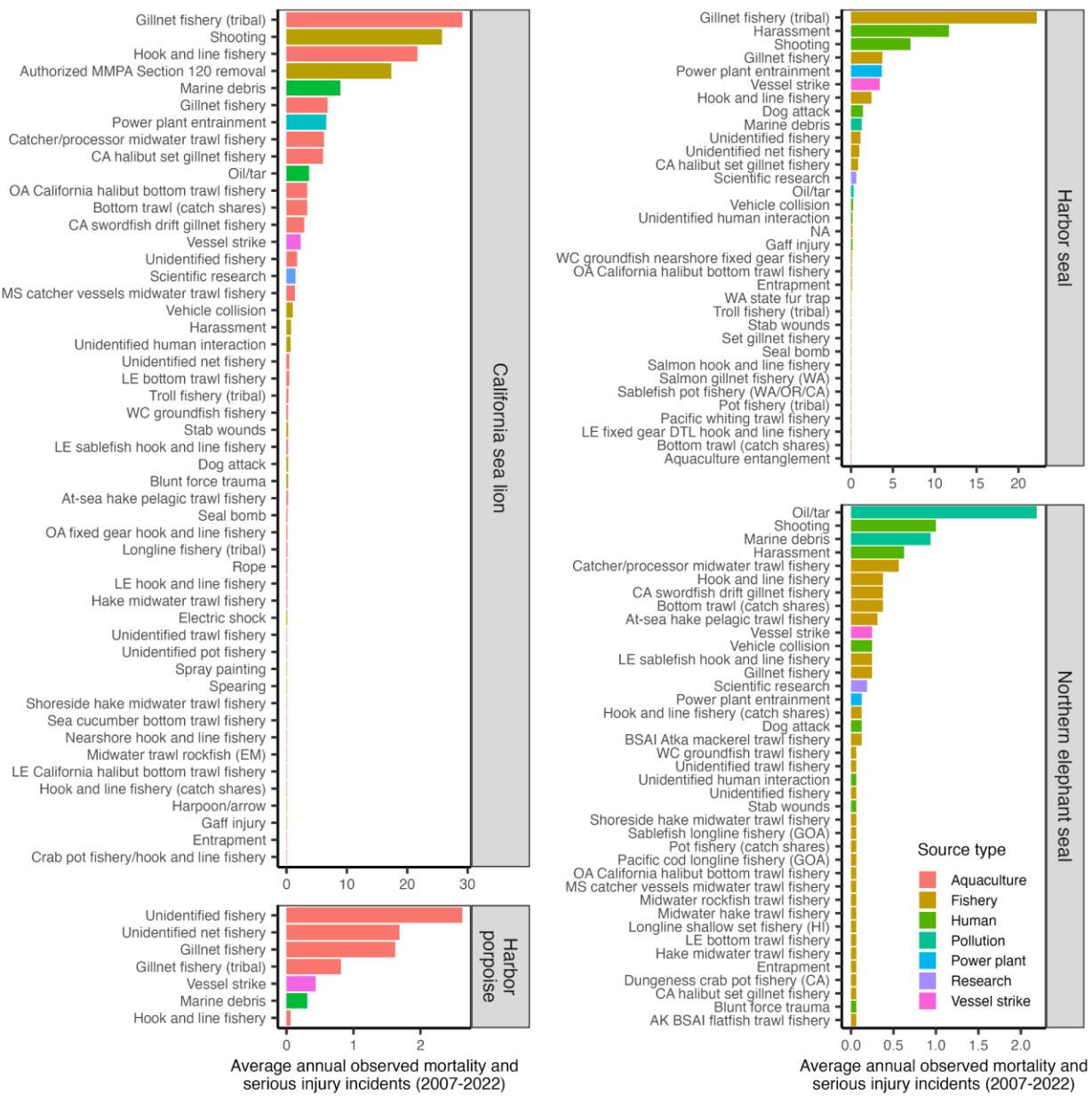
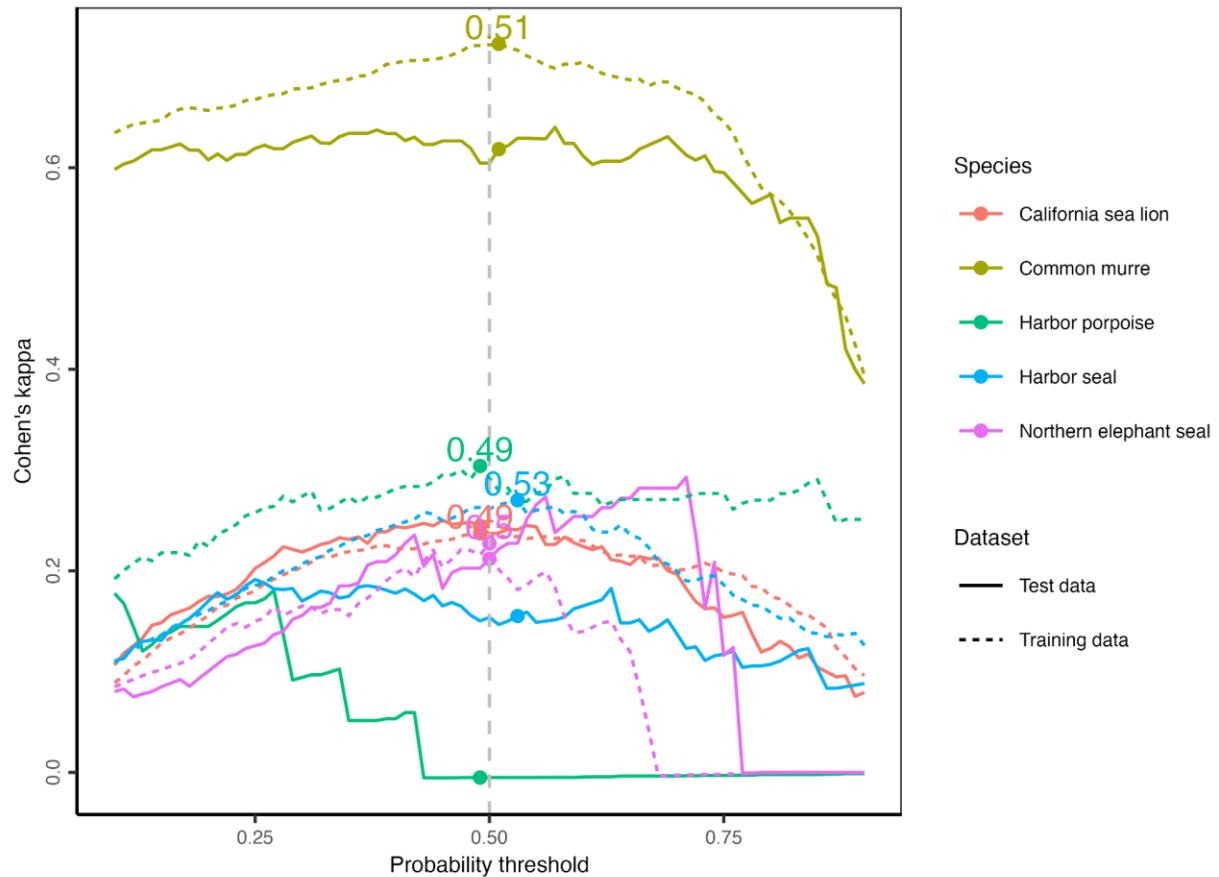


Fig. S16. Average annual observed mortality and serious injury incidents by source on the entire U.S.

West Coast between 2007-2022 (Carretta, 2023).



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1146 **Fig. S17.** Illustration of the methods used to select a probability threshold for classifying a logged set as
1147 having or not having bycatch. We selected the probability threshold that maximizes Cohen's kappa when
1148 applied to the training data as the optimal threshold (labeled in plot). We highlight the performance of this
1149 threshold when used on the independent test data to illustrate performance on out-of-sample data.
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Highlights

- We evaluate protected species bycatch in the California ≥ 3.5 inch set gillnet fishery
- Bycatch has declined precipitously due to management and reduced fishing effort
- Recent marine mammal bycatch ranges from 0.1-4.0% of the potential biological removal
- Targeted time-area closures reduce bycatch better than mesh size or soak time limits
- Competing multiple modeling approaches improves predictions of rare bycatch events

Estimates and drivers of protected species bycatch in the California set gillnet fishery

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Declaration of Interest Statement

The authors declare no conflicts of interest related to this work.

Declaration of interests

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests:

Yutian Fang reports financial support was provided by Arnhold UC Santa Barbara-Conservation International Climate Solutions. Christopher M. Free reports financial support was provided by The Nature Conservancy California. If there are other authors, they declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

CRediT author statement

Yutian Fang: Conceptualization; Methodology; Formal analysis; Investigation; Data Curation; Writing - Original Draft; Writing - Review & Editing; Visualization. **James V. Carretta:** Methodology; Data Curation; Writing - Review & Editing. **Christopher M. Free:** Conceptualization; Methodology; Formal analysis; Investigation; Data Curation; Writing - Original Draft; Writing - Review & Editing; Visualization; Supervision; Project administration; Funding acquisition.