

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

3 Abstract

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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
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19 highlights the value of competing multiple modeling approaches to identify methods that best predict rare
20 bycatch events.

21
22 **Keywords:** gillnet, bycatch, marine mammals, seabirds, area-based management

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

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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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21 38 To guide effective bycatch reduction policies, it is important to understand the magnitude of
22 39 historical and recent bycatch as well as the drivers of bycatch in a fishery. Estimates of total bycatch are
23 40 needed to determine whether bycatch exceeds management targets or is on pace to exceed targets in the
24 41 near future (Bjørge et al., 2013; Geijer and Read, 2013; Read et al., 2006). Historical bycatch estimates
25 42 offer insights into the effectiveness of past management interventions, which provide useful benchmarks
26 43 for adapting management in response to recent bycatch levels and trends. Understanding the drivers of
27 44 bycatch risk is critical to guiding effective and efficient management adaptations. For example,
28 45 determining whether bycatch is concentrated within specific areas or seasons can support the design of
29 46 time-area closures that prevent fishing when and where risk is high while maintaining fishing
30 47 opportunities elsewhere (Lewison et al., 2014; O’Keefe et al., 2023; Soykan et al., 2008). Similarly, gear,
31 48 soak time, or time of day restrictions can be used to curb bycatch if there are strong relationships between
32 49 bycatch risk and gear or other characteristics of fishing (O’Keefe et al., 2023). Without this information,
33 50 bycatch management must be precautionary to guarantee compliance with protected species legislation,
34 51 which could forego considerable fisheries yields.

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38 53 Because observer programs, which place trained scientists on fishing vessels to collect bycatch
39 54 data, are costly and rarely cover all fishing trips, various analytical approaches have been developed to
40 55 estimate unobserved bycatch and to evaluate drivers of bycatch risk. Ratio estimation, a design-based
41 56 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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28 71 The California set gillnet fishery would benefit from updated bycatch estimation due to concerns
29 72 about the fishery's impact on protected marine mammals, which have led some conservation
30 73 organizations to call for the fishery's closure (Birch et al., 2023; Birch and Shester, 2023). The fishery
31 74 occurs in southern California and targets California halibut (*Paralichthys californicus*), white seabass
32 75 (*Atractoscion nobilis*), and Pacific angel shark (*Squatina californica*), among other species. It is currently
33 76 listed as a Category II fishery under the U.S. Marine Mammal Protection Act (MMPA), indicating that it
34 77 presents a medium bycatch threat to protected marine species (NOAA, 2024). Bycatch of marine mammal
35 78 and seabird species, including harbor porpoise (*Phocoena phocoena*), southern sea otter (*Enhydra lutris*
36 79 *nereis*), and common murre (*Uria aalge*) was high during the 1980s and 1990s, prompting large-scale
37 80 management interventions (Forney et al., 2001; Julian and Beeson, 1998). The fishery has also impacted
38 81 pinniped species such as California sea lion (*Zalophus californianus*), harbor seal (*Phoca vitulina*), and
39 82 northern elephant seal (*Mirounga angustirostris*). Total bycatch in the fishery has not been estimated
40 83 since 2012 (Carretta et al., 2014) and some conservation groups are concerned that bycatch remains an
41 84 issue (Birch et al., 2023; Birch and Shester, 2023).

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53 86 A number of management actions have been taken to reduce bycatch in the California set gillnet
54 87 fishery. During the 1980s, high bycatch of southern sea otters and common murres in central California
55 88 (Barlow et al., 1994) led to a depth restriction that closed fishing inside of 40 fathoms (73 m) in 1987
56 89 (Forney et al., 2001). This restriction shut down the fishery in the San Francisco area, effectively pushing
57 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).
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10 95 This effectively closed the fishery in Monterey Bay and Morro Bay (**Fig. 1A-4**). Currently, the fishery
11 only operates in southern California (south of Point Arguello) outside 3 nautical miles from the mainland
12 and outside 1 nautical mile or shallower than 70 fathoms (whichever is less) from the Channel Islands.
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19 100 Although these regulations are believed to have reduced bycatch in the set gillnet fishery
20 (Carretta et al., 2014), they have also greatly reduced fishery participation and revenues. The
21 implementation of the 40 fathom depth restriction in 1987 triggered a precipitous decline in participation
22 from ~400 vessels in 1987 to ~100 vessels in 1994. Since then, participation has continued to decline,
23 with ~40 vessels active in 2022, and the vast majority (>90%) of landings coming from just 13 vessels
24 (CDFW, 2023) (**Fig. 1B**). Fishing effort has similarly decreased from an estimated ~15,000 fishing trips
25 in 1987 to 1,000 trips in 2022 (**Fig. 1C**). This reduction in effort has significantly reduced bycatch levels
26 (Carretta et al., 2014) but at large costs to fishery revenues. Fleetwide revenues decreased from US\$15
27 million in 1987 to US\$1 million in 2022 (**Fig. 1C**; both values in 2022 dollars). Despite declining fishing
28 effort and bycatch, conservation groups are lobbying for additional restrictions, including permanent
29 closure, to further avoid bycatch (Birch et al., 2023; Birch and Shester, 2023). There is thus great need for
30 scientific guidance on management regulations that are likely to provide conservation benefits while also
31 avoiding unnecessary burdens on the fishing industry.
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114 In this study, we reconstruct the bycatch of select marine mammals and seabirds in the California
115 set gillnet fishery from 1981-2022 and identify drivers of bycatch that could be used to refine
116 management to more efficiently reduce bycatch, where efficient management achieves conservation
117 objectives while minimizing impacts on fishing opportunities. We focus on six protected species
118 encountered in the set gillnet fishery: California sea lion, harbor seal, northern elephant seal, harbor
119 porpoise, common murre, and Brandt's cormorant (*Phalacrocorax penicillatus*). Southern sea otter,
120 despite being one of the original species of management concern, is not included in this analysis due to
121 data limitations (**Fig. 2A**). We use ratio estimation methods to reconstruct historical bycatch levels and
122 compare recent bycatch levels to management targets. These methods, which have been used to estimate
123 bycatch in the fishery at various points in the past (**Table S1**), provide a complete time series of bycatch
124 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
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59 183 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
9 percentage of annual fishing trips with onboard observers has varied over time, ranging from 0.3% of
10 trips in 2006 to 16.7% of trips in 1993 (**Fig. 2B; Table S2**). Observers collected information on the
11 188 amount and fate of catch (kept, discarded, or damaged), the length composition of the catch, the location
12 189 and time of the catch, and characteristics of the gear used to target the catch (**Fig. S2**). We developed a
13 190 series of simple assumptions to impute missing values for a few key variables (GPS coordinates, fishing
14 191 depth, soak hour, mesh size) used to describe gillnet sets documented in the observer data (**Fig. S3**; see
15 192 supplemental methods for details).
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19 196 Although historical reports document low levels of observer coverage in Morro Bay, Monterey
20 Bay, and San Francisco in the 1980s (**Table S1**), the data that we received from CDFW excluded most of
21 these observations. We recovered a small portion of the missing raw data – observations from Monterey
22 Bay from 1987-1989 (**Fig. 2**) – from original CDFW data sheets that were given to a colleague at the
23 Southwest Fisheries Science Center during the late 1990s for a reanalysis of historical bycatch rates in
24 that region (Forney et al., 2001). We extracted summaries of set-level bycatch rates from historical reports
25 (**Table S1**) for years and regions missing raw data to support the ratio estimation analysis (**Table S3**). We
26 converted set-level bycatch rates to trip-level bycatch rates assuming an average of 3 sets per trip (**Table**
27 **S3**), as indicated by the observer data (**Fig. S4**). **Fig. 2C** illustrates the coverage of the available,
28 recovered, and lost observer data.
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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).

26 27 2.4.3 Sea surface temperature 28

29 230 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 41 reported GPS location on the reported day of fishing. For trips reported in logbooks, we calculated the
42 43 average SST in the reported block on the reported day of fishing.

44 45 240 2.5 Ratio estimation 46

47 241 We estimated annual bycatch for each study species using ratio estimators. Ratio estimators
48 assume that the rate of bycatch in observed fishing trips is proportional to the rate of bycatch within all
49 fishing trips within a given stratum (Cochran, 1977). This assumption requires that the characteristics of
50 51 observed trips do not systematically differ from the characteristics of all trips, which was confirmed by a
52 53 two-sided Kolmogorov-Smirnov test for six key traits (i.e., day of year, depth, latitude, mesh size,
54 55 distance from shore, soak time) (**Fig. S7**). We used trips rather than sets as the sampling unit given the
56 57 inability to identify unique sets in the logbook data (**Fig. S4B**). This is valid because the number of gillnet
58 59 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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$$m_{s,i} = D_i * r_{s,i}$$

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26 261 Where the total number of trips (D_i) is derived from the logbook data.
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30 263 We calculated annual bycatch estimates using a seven-region stratification scheme (**Figs. 1A, S8**).
31 This stratification scheme combines the scheme by Diamond and Hanan (1986) and Julian (1993) for
32 areas north and south of Point Conception, respectively. Although early efforts to estimate bycatch in the
33 California set gillnet fishery often stratified estimates by region and season (**Table S1**), later efforts found
34 that observer coverage was often too limited to employ complex temporal stratification and that estimates
35 between temporally stratified and unstratified approaches were generally similar (**Table S1**). Stratum-
36 specific bycatch rates for years without observer coverage in the stratum are borrowed from the closest
37 year (forwards or backwards) with observer coverage in the stratum (**Fig. 2C & S9**), as has been the
38 practice in previous studies. We collated annual bycatch estimates from past studies (**Table S1**) for
39 comparison with our updated estimates (**Figs. S10-S12**).
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44 274 Although there are methods for estimating the uncertainty of bycatch estimates generated through
45 ratio estimation (Julian and Beeson, 1998), we were unable to implement these methods because they rely
46 on bootstrap procedures that sample from the bycatch rates of observed trips. Because these procedures
47 require raw observer data, we cannot use them for (1) years where summary values from historical reports
48 are used because the raw data have been lost or (2) years without observer data from within one of the
49 fished strata. As a result, only 6 of the 42 evaluated years had the data required to estimate uncertainty:
50 2006, 2007, 2010, 2011, 2012, and 2017 (**Fig. 1AC; Fig. 2BC**). An exploration of the uncertainty
51 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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295 2.6 Random forest modeling

296 2.6.1 Model training

297 We used random forest classification models trained on the observer data to identify drivers of
298 bycatch risk for each of the six study species. We used a random forest approach, a machine learning
299 method that ensembles predictions from hundreds of decision trees, rather than a classical regression
300 method (e.g., generalized linear or additive models) because of their comparatively high predictive skill
301 for rare events, ability to model non-linear relationships, and insensitivity to collinear or unimportant
302 predictor variables (Cutler et al., 2007; Prasad et al., 2006). We considered nine attributes of fishing as
303 potential drivers of bycatch risk: haul depth (fathoms), mesh size (inches), soak time (hours), latitude
304 ($^{\circ}$ N), longitude ($^{\circ}$ W), distance from shore (km), Julian day, sea surface temperature ($^{\circ}$ C), and whether the
305 fishing occurs near an island (i.e., within 10 km of island coast). These attributes were selected based on
306 their demonstrated relationship to bycatch risk in other papers (e.g., (Bettoli and Scholten, 2006; Bjørge et
307 al., 2013; Kroetz et al., 2020)) and their availability in the observer data or their ability to be derived
308 through remote sensing (i.e., distance from shore, temperature, island area). They represent a range of
309 spatial (latitude, longitude, distance from shore, depth, island area), temporal (Julian day), environmental
310 (temperature), and fishing-related (soak time, mesh size) attributes. For each species, we classified an
311 observed set as having (1) or not having (0) bycatch, and trained a classification model assuming a
312 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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26 326 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.
43 Specifically, we assigned majority observations (sets without bycatch) a weight of 1 and assigned
44 minority observations (sets with bycatch) weights of 25 to 200 in increments of 25. Thus, a total of eight
45 candidate weighted random forest models were evaluated as described below. We fit the weighted
46 random forest model using the *ranger* package in R (Wright and Ziegler, 2017).
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54 344 We trained each of the eleven candidate models (three balanced random forest models, eight
55 weighted random forest models) on 80% of the observer data, withholding the remaining 20% for model
56 testing. In training the models, we performed a grid search to identify the “mtry” hyperparameter – the
57 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

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4 410 the Morro Bay stock, where the bycatch took place), California sea lion (194 individuals per year = 1.4%
5 of a PBR of 14,011 individuals), and northern elephant seal (7 individuals per year = 0.1% of a PBR of
6 5,122 individuals) (**Fig. 7**). The assessment that bycatch during the last 10 years poses the greatest risk to
7 harbor seals is supported by the fact that the harbor seal stock size has been stable or declining in recent
8 years while all of the other marine mammal populations have been undergoing sustained population
9 growth (**Fig. 7**). The sustainability of estimated seabird bycatch is more difficult to evaluate given more
10 limited population monitoring data (**Fig. 7**) and the lack of legally binding reference points for defining
11 allowed incidental take. However, increasing Brandt's cormorant nests from 1980 to 2020 (**Fig. 7**) and
12 steeply reduced bycatch of common murre (**Fig. 3**) suggests low risks posed to these species.
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21 419 3.2 Random forest modeling
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24 420 3.2.1 Model performance
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26 421 The best-fitting model performed well for all species, except for Brandt's cormorant and harbor
27 porpoise, which exhibited poor performance (Cohen's kappa less than 0.2) and were therefore excluded
28 from further consideration. Weighted random forest models performed best for California sea lion,
29 common murre, northern elephant seal, and harbor seal with case weights of 25, 25, 25, and 75,
30 respectively (**Table 1; Fig. S13**). For common murre, Cohen's kappa was 0.71, indicating "good"
31 performance, while for harbor seal, California sea lion, and northern elephant seal, Cohen's kappa was
32 0.25, 0.24, and 0.23, respectively, indicating "fair" performance. Model performance was positively
33 correlated with the frequency of bycatch observations, i.e., species with more observed bycatch events
34 produced models with greater skill (**Table 1**; $r^2 = 0.64$ for Cohen's kappa for training data). Cohen's
35 kappa was positively correlated with the area under the receiver operator curve (AUC) ($r^2 = 0.68$ for the
36 training dataset), indicating minimal tradeoffs in using this metric for model selection (**Table 1**).
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46 432 3.2.2 Drivers of bycatch risk
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48 433 The importance of the evaluated explanatory variables in determining bycatch risk varied by
49 species but some general patterns emerged (**Fig. 4**). In general, spatial (latitude, longitude, depth, and
50 distance from shore) and temporal (Julian day) variables were more influential than variables associated
51 with the environment (sea surface temperature) or the fishing methodology (soak time, mesh size).
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53 436 Whether fishing occurred close to an island (a spatial variable) was the exception, as it was consistently
54 the least important variable. Sea surface temperature, which is closely related to space and time, was
55 generally more important than soak time and always more important than mesh size.
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435 distance from shore) and temporal (Julian day) variables were more influential than variables associated
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438 the least important variable. Sea surface temperature, which is closely related to space and time, was
439 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
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10 444 latitude, including some deeper offshore areas (**Fig. 6**). They also exhibited a pronounced increase in risk
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12 445 during the spring, lasting approximately from Apr 1 (90th day of the year) to June 15 (166th day of the
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14 446 year). They are infrequently caught in nets with mesh sizes smaller than 8.5 inches, though the use of
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16 447 such nets is rare (**Fig. S5E**). Variability in bycatch risk for common murre and northern elephant seal is
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18 448 most strongly determined by latitude and longitude (**Fig. 5**), with the only area of elevated risk in
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20 449 contemporary fishing grounds occurring just north of Point Conception (**Fig. 6**). For all four species,
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22 450 bycatch risk exhibits an asymptotic relationship with soak time, though the shapes of these relationships
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24 451 differ by species. The species exhibit complex and variable relationships to temperature.
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27 452 3.2.3 Maps of bycatch risk
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30 453 The species exhibited different patterns of spatial bycatch risk. California sea lion bycatch risk is
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32 454 predicted to be highest in four areas: (1) on the northern coasts of the northern Channel Islands, especially
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34 455 on the northern coast of Santa Rosa; (2) a small nearshore area west of Santa Barbara; (3) the eastern
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36 456 coast of Santa Cruz Island; and (4) the northwestern shores of Santa Catalina and San Clemente Islands;
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38 457 (**Fig. 6**). Harbor seal bycatch risk is predicted to be highest in four areas: (1) the sliver of nearshore area
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40 458 stretching from Santa Barbara to Point Sal; (2) the eastern coasts of Santa Cruz Island; (3) a broad coastal
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42 459 area near Point Mugu; and (4) the sliver of nearshore area stretching from Point Mugu to Point Vicente
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44 460 (**Fig. 6**). Common murre bycatch risk is predicted to be negligible throughout most of southern California
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46 461 (**Fig. 6**). It is only predicted to be high in a small patch near Point Sal and even there, the maximum risk
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48 462 index is much lower than for the other evaluated species. Like common murre, northern elephant seal
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50 463 bycatch risk is also predicted to be negligible throughout most of southern California except in the region
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52 464 near Point Sal (**Fig.6**).
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465 4. Discussion

466 Our study provides the first update to total estimates of protected species bycatch in the
467 California set gillnet fishery since 2012. We find that bycatch, once high and unsustainable for some
468 species (Forney et al., 2021, 2001), is now well below the “zero mortality rate goal” (ZMRG) of 10% of
469 the potential biological removal (50 C.F.R. § 229.2). Recent marine mammal bycatch estimates range
470 from 0.1-4.0% of their potential biological removals and common murre bycatch has been effectively
471 eliminated. All of the evaluated populations, including the once declining and heavily depleted Morro

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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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48 498 Hotspots of bycatch risk are aligned with the location of known haulouts, breeding colonies, and
49 foraging grounds. High California seal lion bycatch risk around the northern Channel Islands is likely
50 related to the large haulouts of sea lions in that area (**Fig. S15**). Similarly, hotspots of harbor seal bycatch
51 risk correspond to the locations of large harbor seal haulouts on Santa Cruz Island and near Point Mugu
52 (**Fig. S15**). The absence of common murre bycatch risk in southern California is consistent with the
53 distribution of the species, which has no breeding colonies or permanent foraging grounds in southern
54 California (**Fig. S15**). Similarly, the negligible risk for northern elephant seals is consistent with the
55 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 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 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 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 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 perhaps by prioritizing characteristics known to impact bycatch risk in other gillnet fisheries (Northridge
27 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 and improved documentation of the characteristics of logged sets would enhance future bycatch estimates
30 by allowing sets to be the sampling unit and by avoiding assumptions about missing data, respectively.
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32 523 This could be achieved by redesigning logbooks, training fishers on completing logbooks, expanding
33 electronic monitoring, and/or demonstrating that better data can actually lead to fewer restrictions.
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39 531 Our results highlight the importance of considering multiple modeling approaches when
40 estimating and evaluating rare bycatch events. Although model-based methods (e.g., random forests) for
41 estimating bycatch are often preferred to sample-based methods (e.g., ratio estimators) (Stock et al.,
42 2019), we find complementary value in using both approaches. While ratio estimation generated more
43 reliable bycatch estimates due to its ability to leverage both raw and summarized data, the random forest
44 model provided the empirical basis for assessing drivers of bycatch risk. Furthermore, our results
45 highlight the value of considering multiple sample balancing approaches when evaluating bycatch using
46 model-based methods, as the specification of the best performing model varied by species. Recent efforts
47 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
5 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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561 There will always be tradeoffs between maximizing fishing opportunities and minimizing bycatch
562 of protected species (Samhouri et al., 2021). As a result, managers often seek to implement regulations
563 that maximize fishing outcomes while keeping bycatch below legally defined sustainable reference points
564 (Kirby and Ward, 2014). The identification of such strategies is seldom straightforward and depends on
565 substantial investments in data and scientific enterprises. Notably, they depend on monitoring populations
566 of protected species to support the assessment of their status and levels of allowable incidental take and
567 monitoring bycatch in key fisheries to support assessments of total bycatch, drivers of bycatch, and the
568 effectiveness of past management interventions (Kirby and Ward, 2014; Punt et al., 2021). In the absence
569 of such data, management must often be precautionary to ensure compliance with protected species
570 legislation (Punt et al., 2021). We illustrate the potential return on investment of supporting such
571 scientific enterprises as our results show that past management interventions have been successful at
572 reducing bycatch in the California set gillnet fishery well below target levels, opening the door for more
573 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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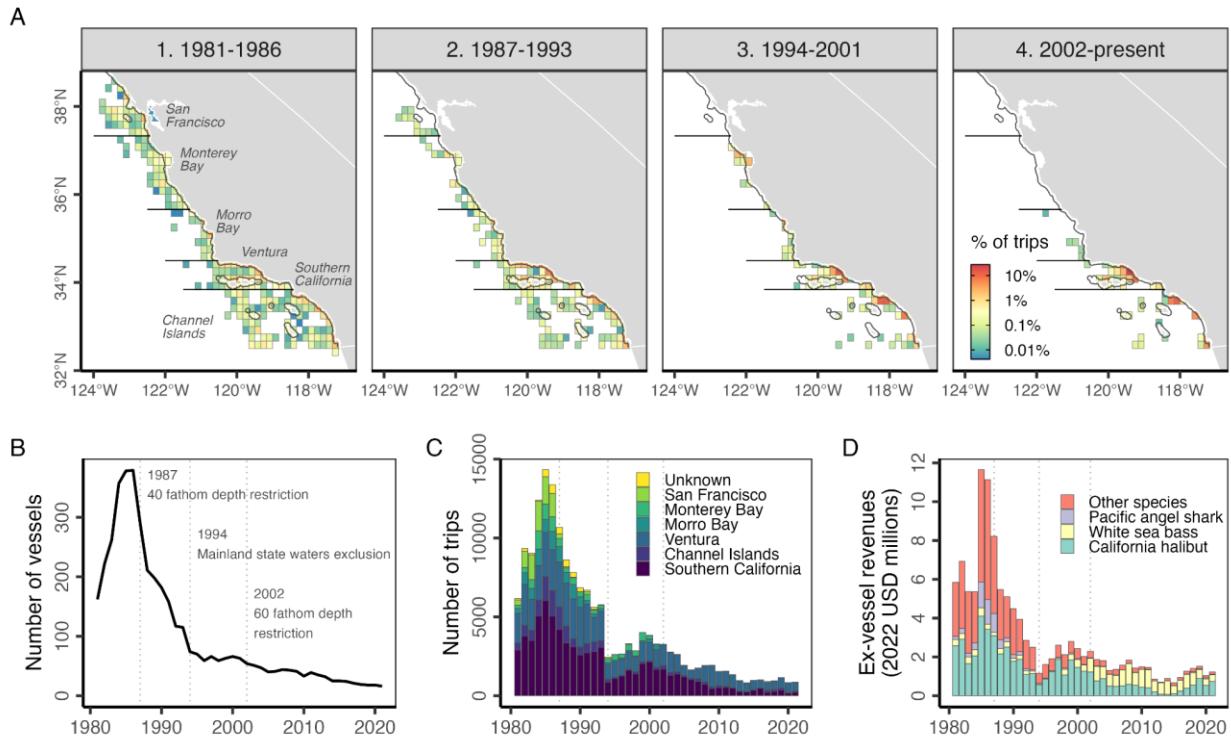
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837 Tables & Figures



838
839 **Fig. 1.** History of the California $\geq 3.5"$ set gillnet fishery. Panel A shows the spatial history of fishing
840 effort during four regulatory periods. Trips are reported by the 10×10 minute ($\sim 18 \times 18$ km) statistical
841 blocks used for fisheries catch and effort reporting. The horizontal lines delineate geographical strata used
842 in the ratio estimation analysis; strata are labeled in the first plot. The thin coastal line marks state waters
843 (less than 3 nautical miles from the coast). Blocks visited by fewer than three vessels during each
844 regulatory period are hidden to maintain confidentiality and comply with the “rule-of-three.” The other
845 panels show time series of fisheries (B) participation, (C) effort, and (D) revenues. Vertical lines mark
846 years in which major regulations, labeled in Panel B, were implemented; these define the regulatory
847 periods used in Panel A. These regulations became operative on April 15, 1987; January 1, 1994; and
848 April 26, 2002. See Fig. S1 and the supplemental methods for details on estimating ex-vessel revenues
849 from the fishery.

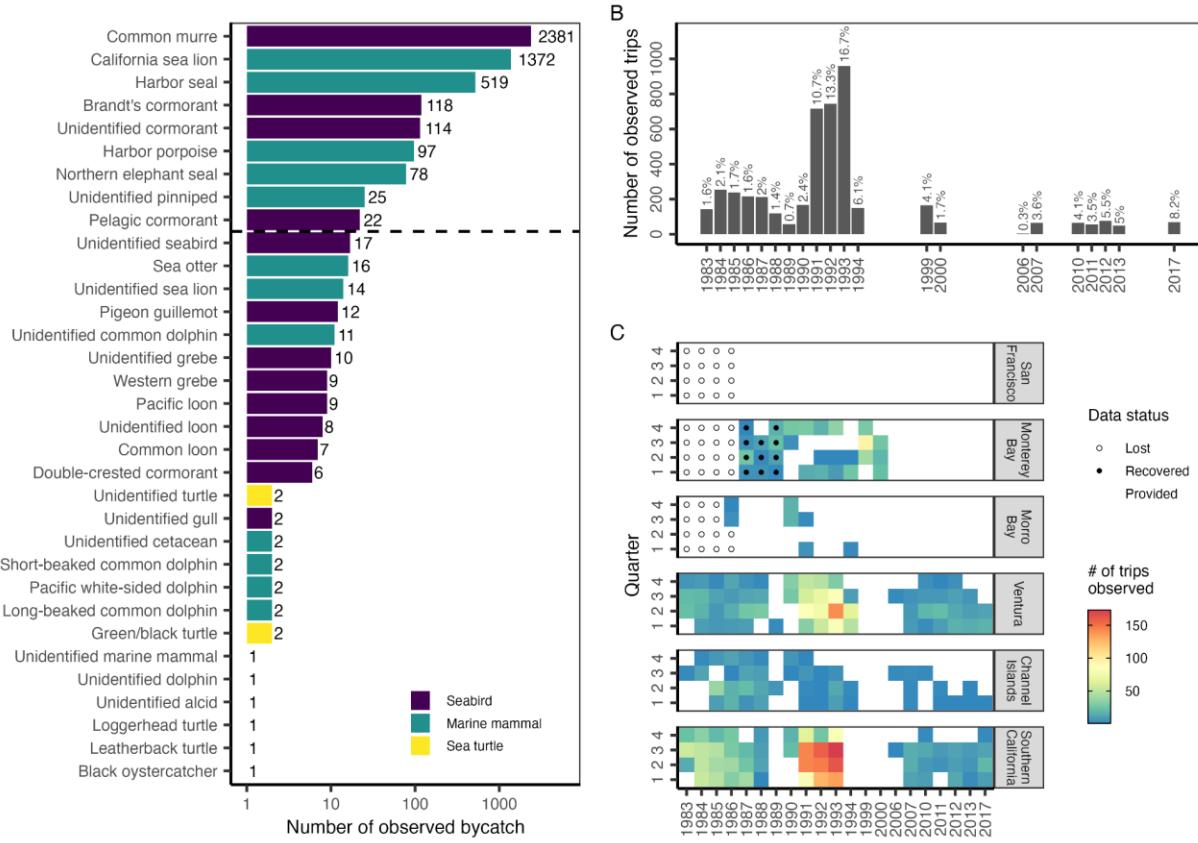


Fig. 2. History of observer coverage in the California $\geq 3.5"$ set gillnet fishery. Panel A shows the bycatch of marine mammals, seabirds, and sea turtles recorded by observers from 1983–2017. We focus on species with ≥ 50 observations, which are delineated by the horizontal dashed line. Note log-scale 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).

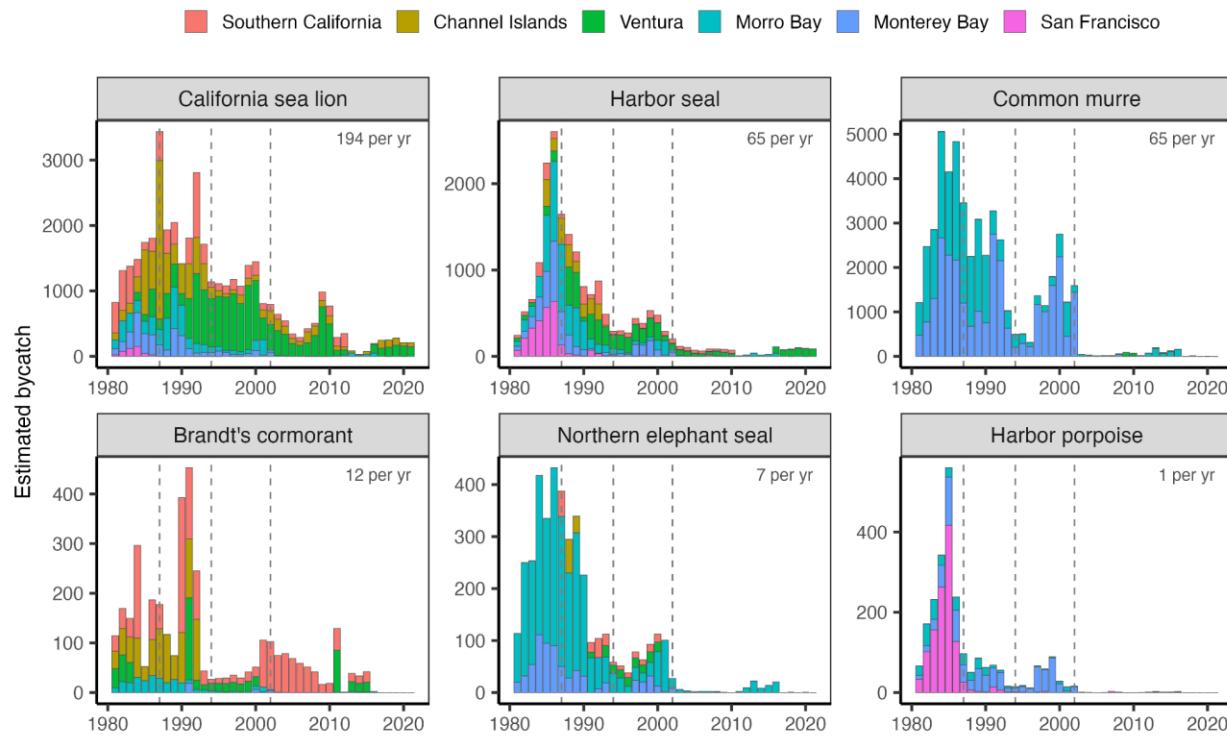


Fig. 3. Estimated bycatch in the California $\geq 3.5"$ set gillnet fishery from 1981-2021 predicted using the ratio estimation approach. Average estimated annual bycatch rates for the last 10 years (2012-2021) from the ratio estimator are marked in the top-right corner. Species are listed in order of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

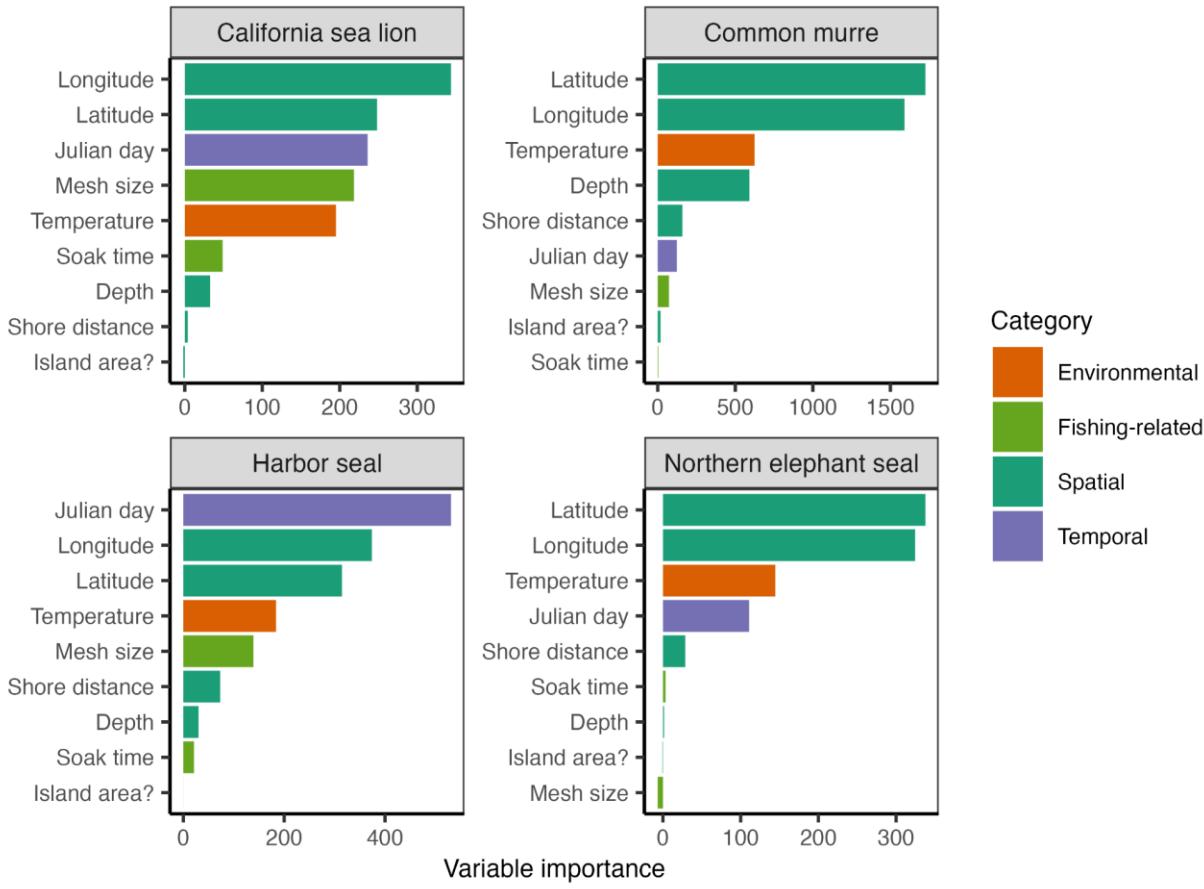
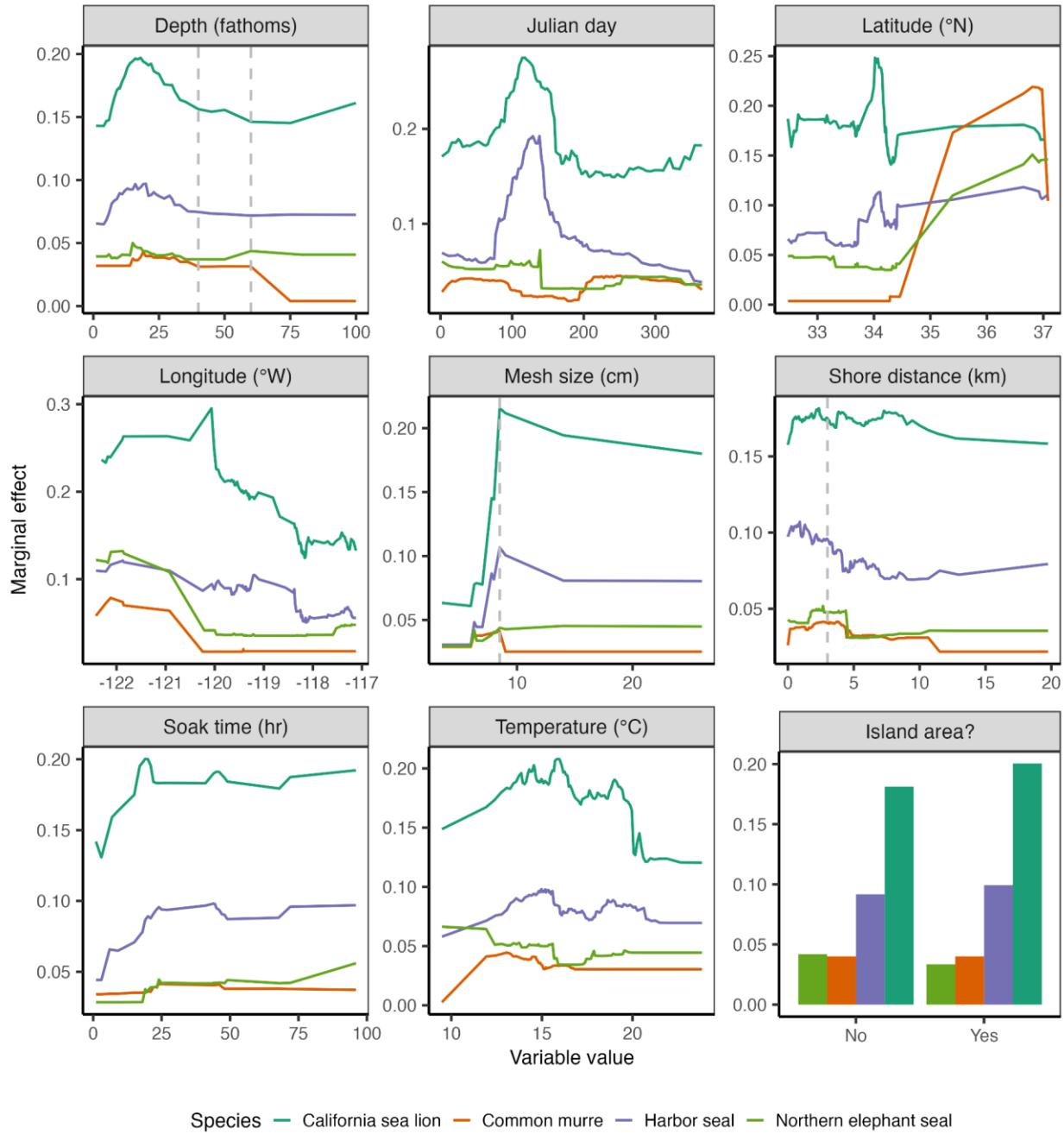


Fig. 4. Importance of the evaluated explanatory variables in the final random forest model for the study species with acceptable model performance. With the bias towards continuous variable corrected, variable importance is measured as the total decrease in node impurities from splitting on the variable 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).

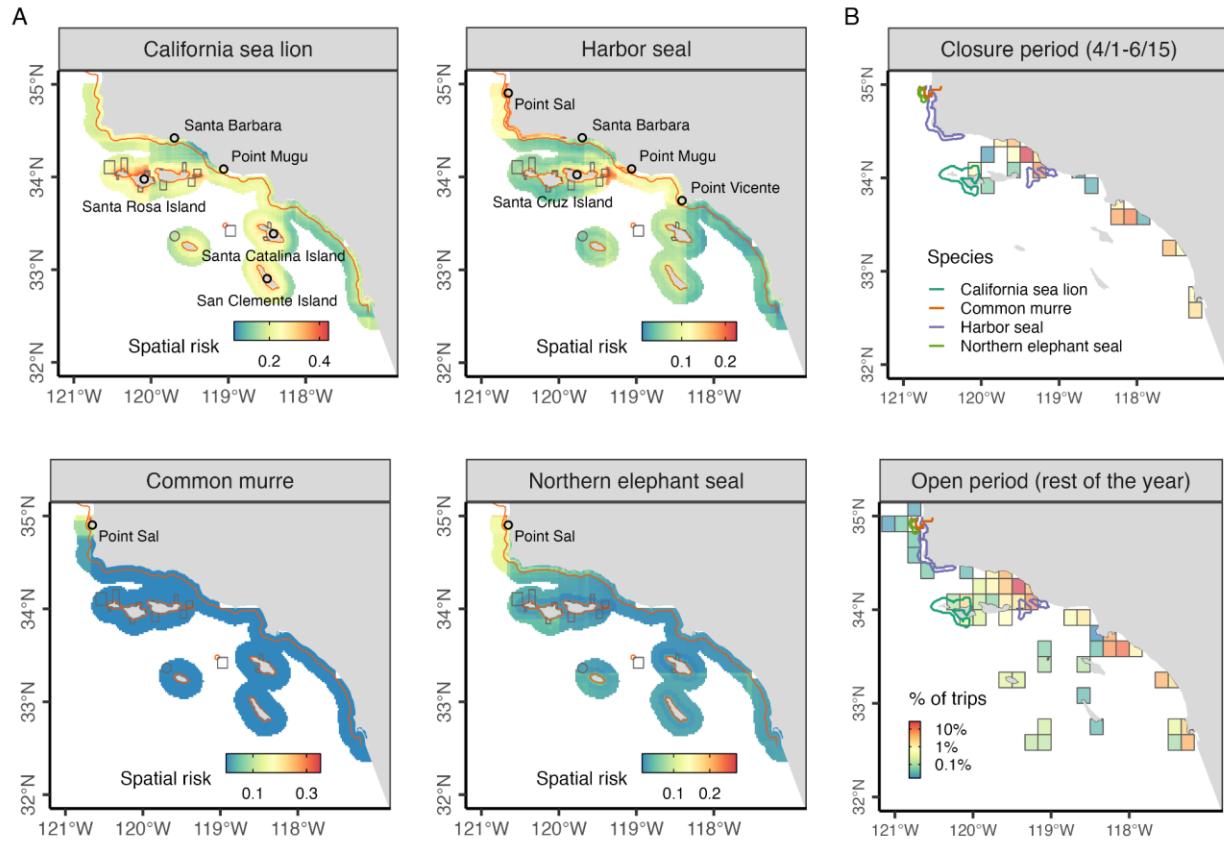


Fig. 6. The (A) spatial bycatch risk relative to current management areas and (B) spatial bycatch hotspots relative to recent fishing effort. Panel A shows the average spatial bycatch risk as estimated by the best fitting random forest model for the four study species with acceptable model performance. 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.

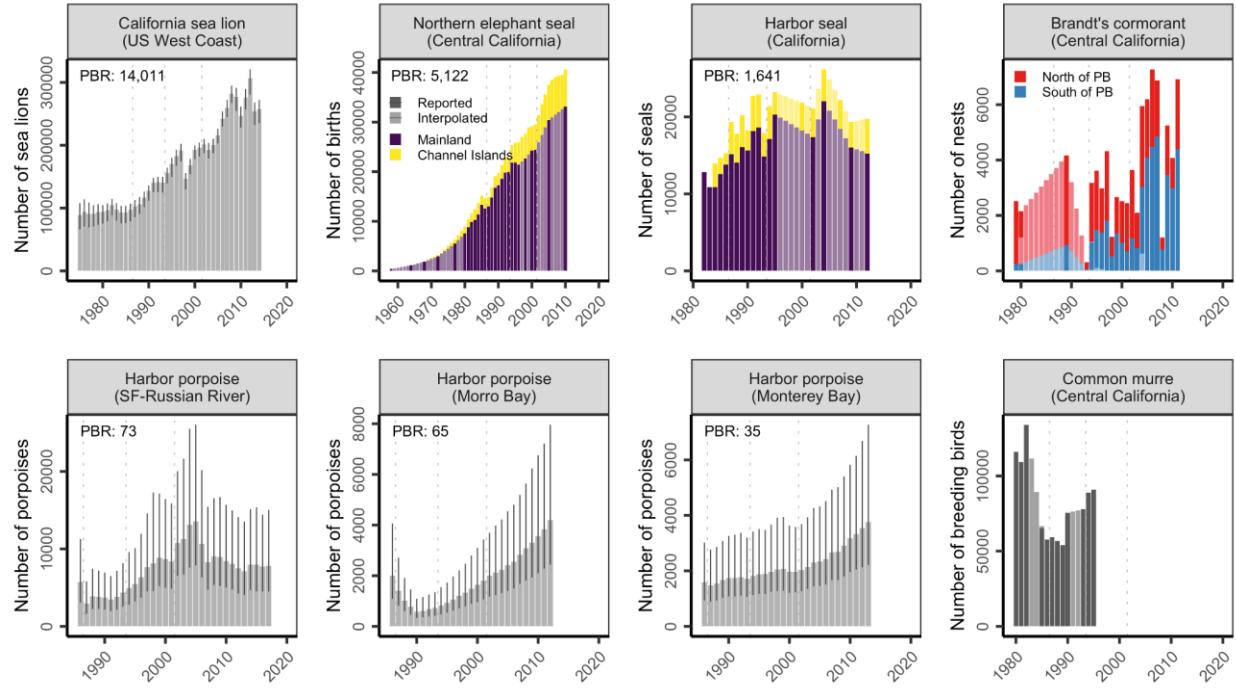


Fig. 7. Estimated abundance of populations of the six study species. The potential biological removal (PBR) indicates the maximum number of animals, not including natural mortalities, that may be removed from a marine mammal stock while allowing the stock to reach or maintain its optimum sustainable 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
16 939 receiver operator curve (AUC) because (1) the models were tuned and selected based on Cohen’s kappa
17 940 and (2) simulation work shows that deriving thresholds based on AUC tends to overestimates the
18 941 prevalence of rare events while it underestimates the prevalence of common events (Freeman and Moisen,
19 942 2008; Manel et al., 2001). We summed the number of pseudo-sets predicted to have bycatch each year,
20 943 converted this sum to “true sets” assuming three sets per pseudo-set (**Fig. S4AB**), and multiplied this sum
21 944 by the median number of captures when a capture occurs to generate estimates of the total number of
22 945 captured animals (**Fig. S4C**). We opted not to employ a more complex two-stage or hurdle model
23 946 approach, where a second model estimates the number of captured individuals when bycatch occurs,
24 947 given the rarity of bycatch events larger than one for all species but common murre (**Fig. S4C**).
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26 949 The random forest models estimate trends in bycatch that are similar to the estimates from the
27 950 ratio estimator from 2000-2022 (**Fig. 3**). However, the estimates produced by the two approaches diverge
28 951 from 1981-2000 by various extents. While they generally agree for California sea lion back to 2000, the
29 952 random forest model underpredicts bycatch relative to the ratio estimator in the late 1990s and
30 953 overpredicts in the 1980s and early 1990s (**Fig. 3**). While the approaches generally agree for harbor seal
31 954 back to 1995, the random forest model underpredicts bycatch relative to the ratio estimator before 1995,
32 955 especially in the Channel Islands and Ventura strata (**Fig. S14**). For common murre, the random forest
33 956 model overpredicts bycatch relative to the ratio estimator in the mid- to late-1990s and underpredicts
34 957 relative to the ratio estimator in earlier years, especially in Morro and Monterey Bays. These
35 958 underpredictions likely occur because of the unequal impacts of lost data from the northern strata in the
36 959 1980s (**Fig. 2**). Unlike the random forest models, the ratio estimators are able to use summarized observer
37 960 data for this region and time period from old reports. As a result, the ratio estimators can learn from
38 961 observations from this region and time period while the random forest models are blind to data from this
39 962 region and period. Thus, the random forest models are likely to underpredict risk in early years in
40 963 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 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 photographic surveys. Sea lion haulouts occur along the California coast but were not mapped to single
47 1019 sites in this study and therefore not plotted in our range maps. We mapped seabird colonies using the
48 1020 2010 CDFW Seabird Colonies Database (CDFW, 2010). These data were collected as part of the Marine
49 1021 Life Protection Act (MLPA) planning process and report the maximum number of seabirds of 26 species
50 1022 at all known colonies.
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49 1021 Life Protection Act (MLPA) planning process and report the maximum number of seabirds of 26 species
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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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4 **Table S2.** Number of fishing trips in the California 3.5 inch mesh set gillnet fishery by year.
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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	Raw data available?
				observed	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	*	

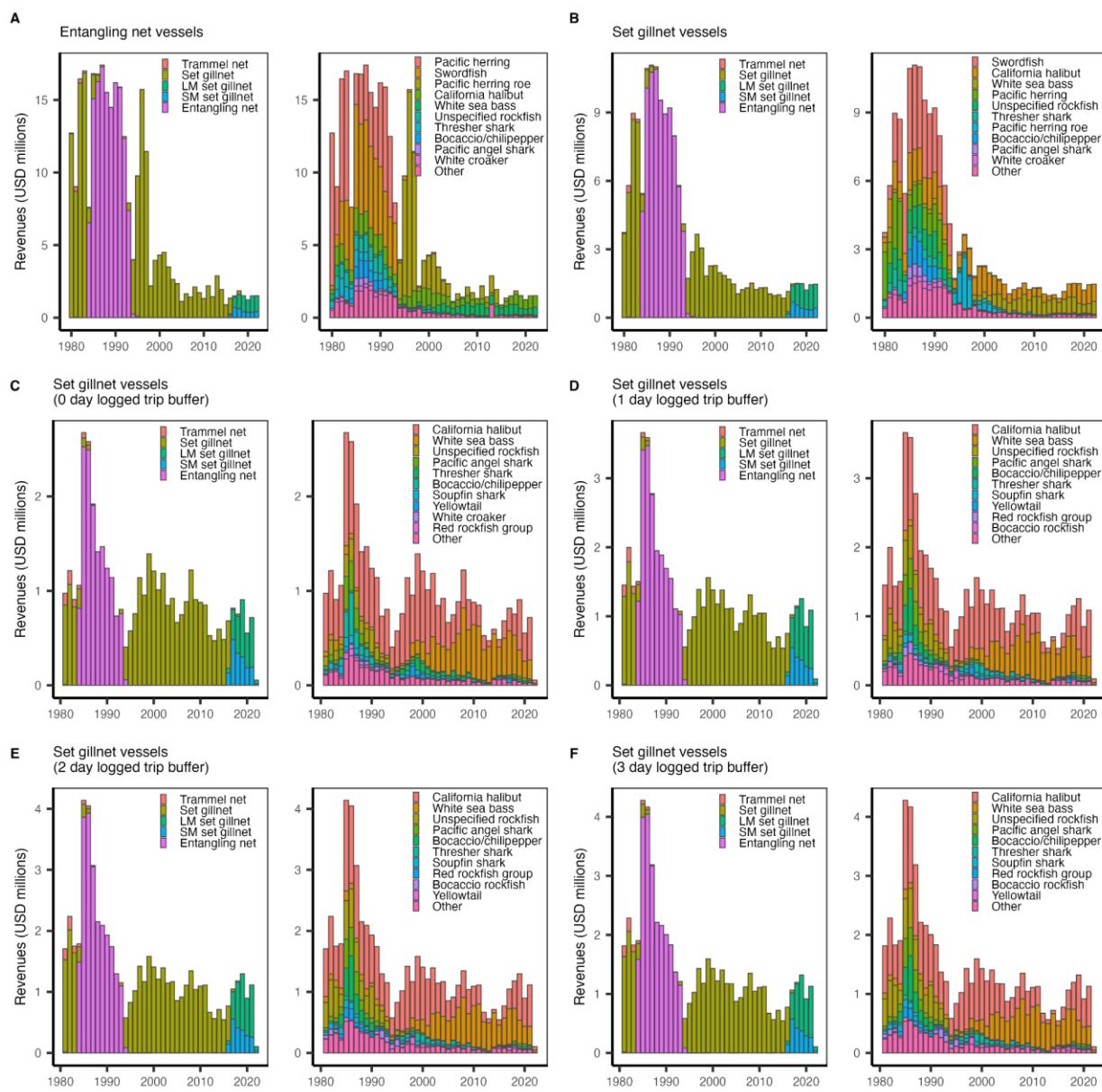
⁴1037 **Table S4.** Assumptions made throughout the analysis, their likelihood, and their potential impact on the
⁵1038 results (*assumptions that are expected to be valid or true on average are not likely to impact the results).
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Assumption	Likelihood 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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6 1044 Fig. S1. Estimated ex-vessel revenues generated by the California $\geq 3.5"$ set gillnet fishery as estimated
7 through six different filtration procedures. The filtration procedures examine the sum annual ex-vessel
8 revenues reported on landing receipts from (A) vessels using various reported entangling net gears; (B)
9 vessels using various entangling net gears that are known to use set gillnets based on logbooks; and (C-F)
10 vessels known to use set gillnets based on logbooks that are dated within various buffers of a logged set
11 gillnet trip. We adopted the final filter, which sums landing receipts date within 3 days of logged set
12 gillnet trip, as the best estimate of ex-vessel revenues for the fishery. Revenues have not been adjusted for
13 inflation (see Fig. 1D for the inflation adjusted ex-vessel revenues).
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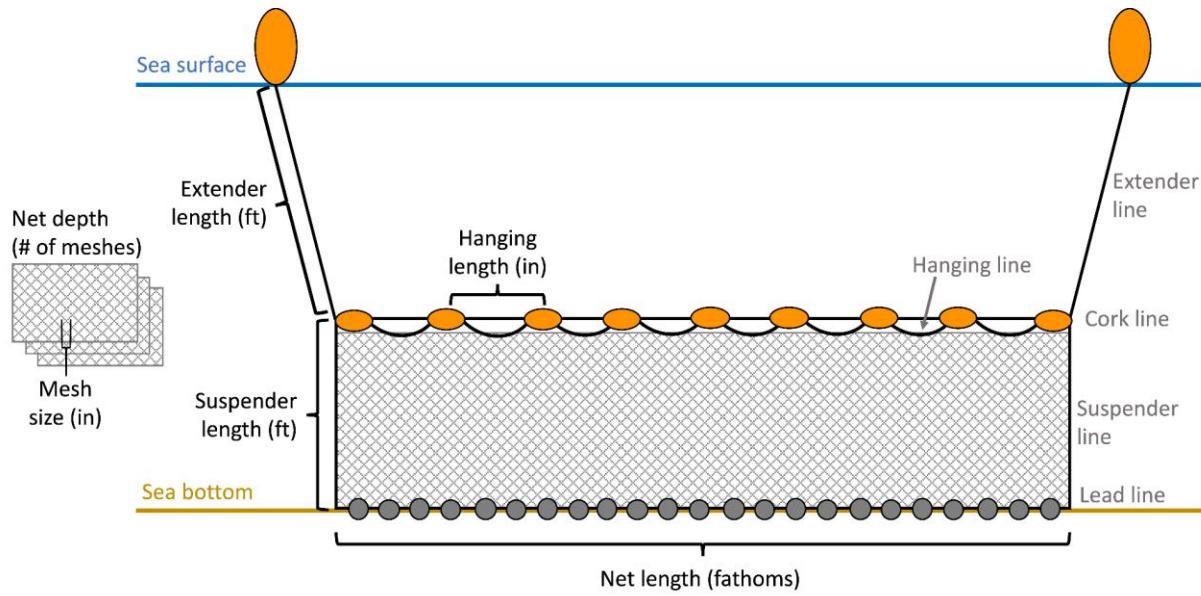
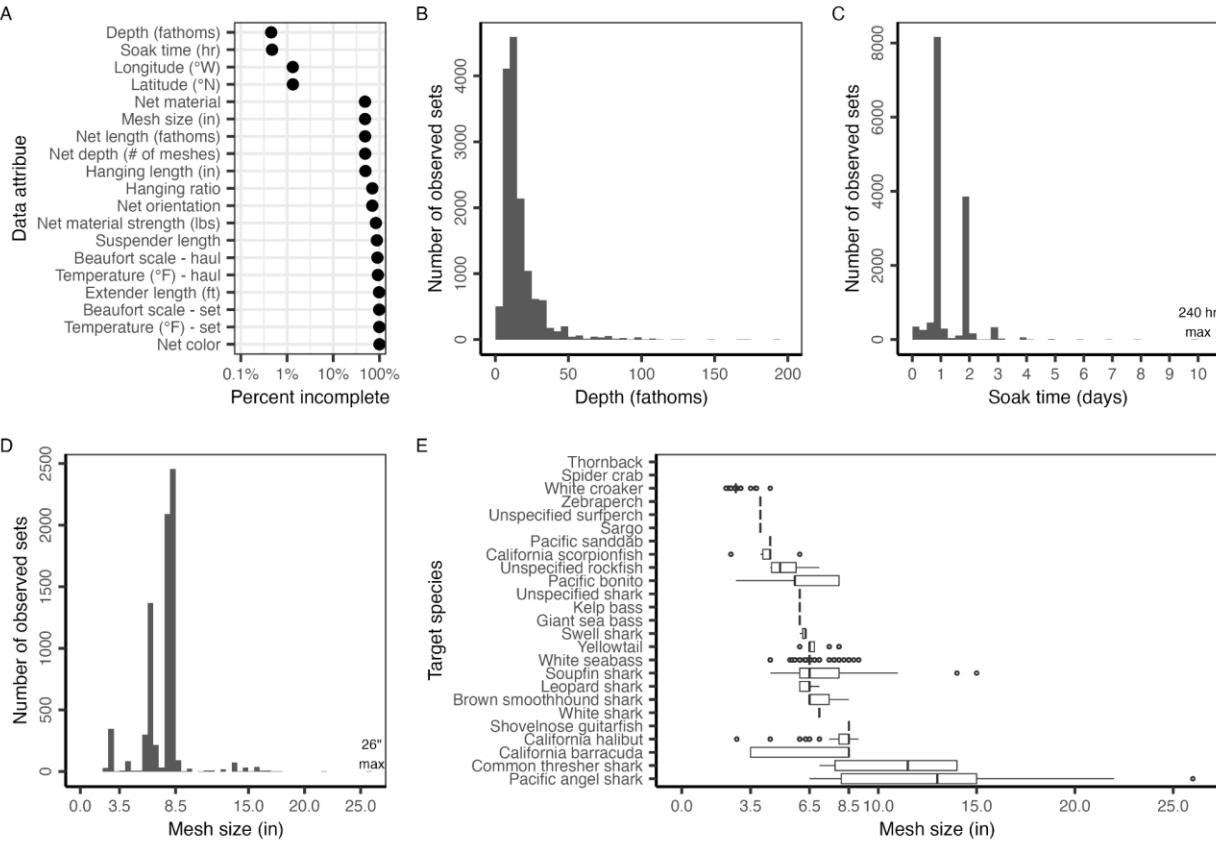
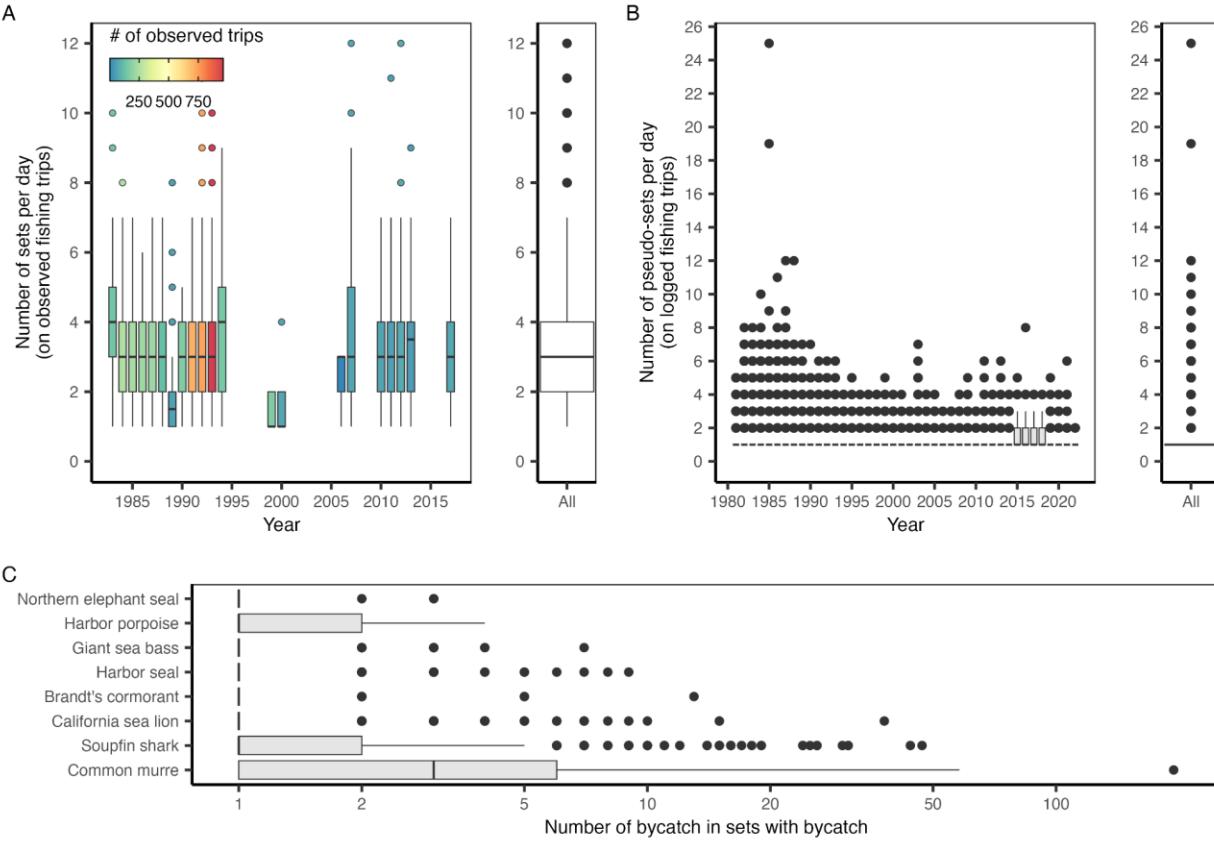


Fig. S2. Diagram of a typical California set gillnet illustrating the measurements reported in the logbook and observer data. Grey text indicates parts of the gillnet and black text indicates the reported measurements. The material, strength, and color of the net is also reported. Finally, the hanging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



32 1058
33 1059 Fig. S3. Traits of the observed set gillnet metadata. Panel A shows the level of completeness of gillnet
34 metadata. Panel B shows the distribution of reported depths. Panel C shows the distribution of reported
35 1060 soak times; the maximum reported soak time is 240 hours (10 days). Panel D shows the distribution of
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37 1061 reported mesh sizes; the maximum reported mesh size is 26 inches. Panel E shows the distribution of
38 1062 reported mesh sizes by reported target species. In the boxplots, the solid line indicates the median, the box
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40 1063 indicates the interquartile range (IQR; 25th to 75th percentiles), the whiskers indicate 1.5 times the IQR,
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42 1064 and points indicate outliers.
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32 1066
33 1067 **Fig. S4.** The (A) number of sets per observed fishing trip by year and overall; (B) number of pseudo-sets
34 35 1068 per logged fishing trip by year and overall; and (C) number of bycatch in sets with bycatch of each of the
36 37 1069 species of interest. In the boxplots, the solid line indicates the median, the box indicates the interquartile
38 39 1070 range (IQR; 25th to 75th percentiles), the whiskers indicate 1.5 times the IQR, and points indicate
40 1071 41 1072 outliers. In (A), the fill color indicates the number of observed fishing trips contributing to the annual
42 distribution.

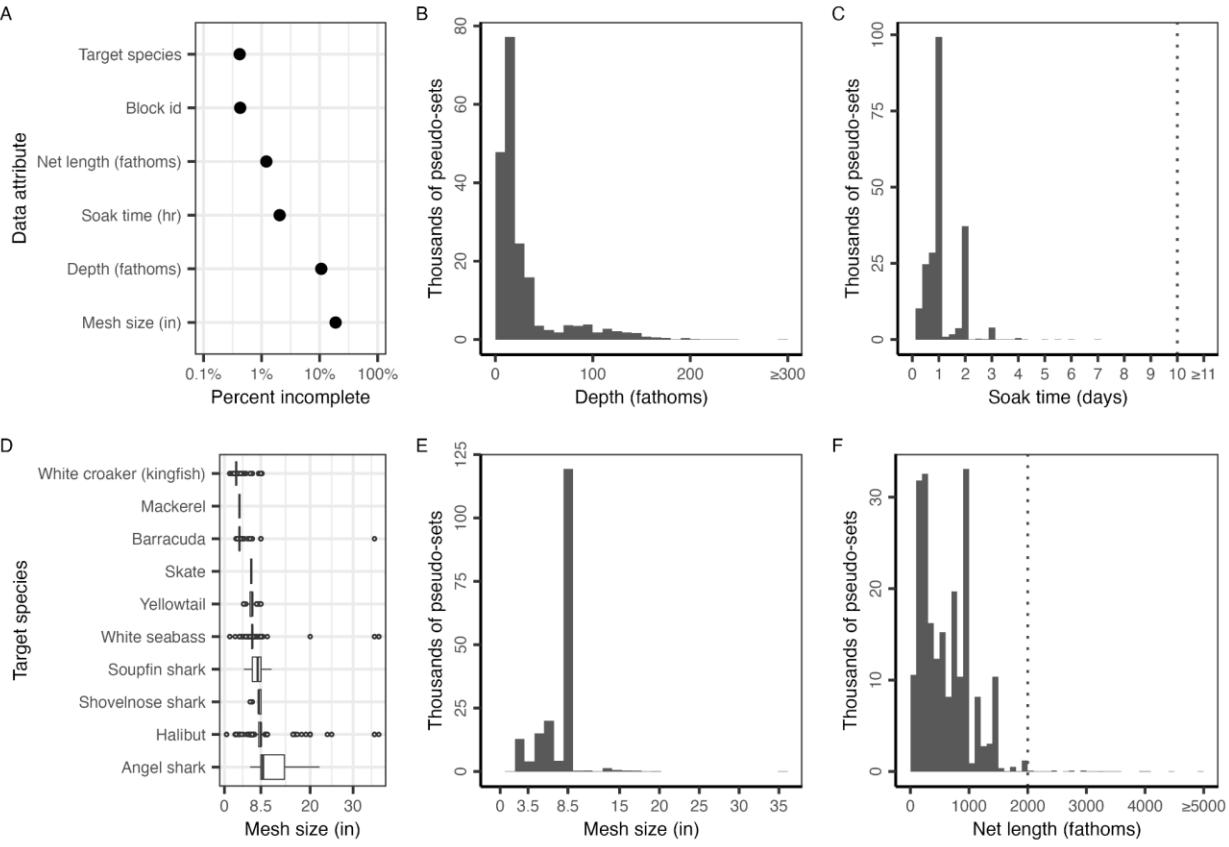
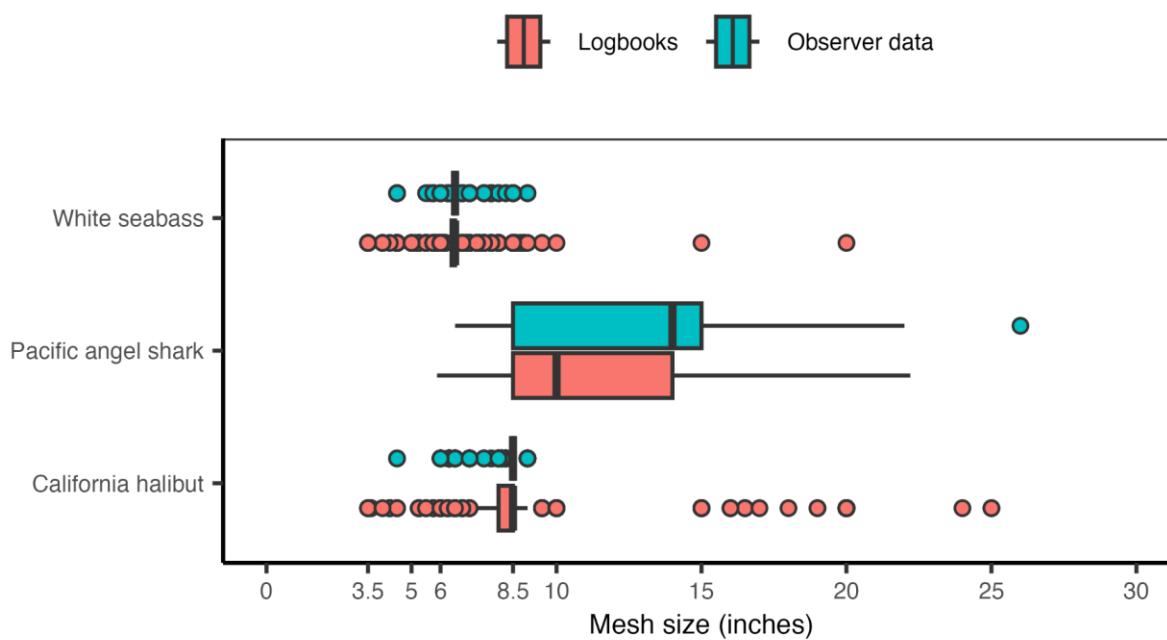
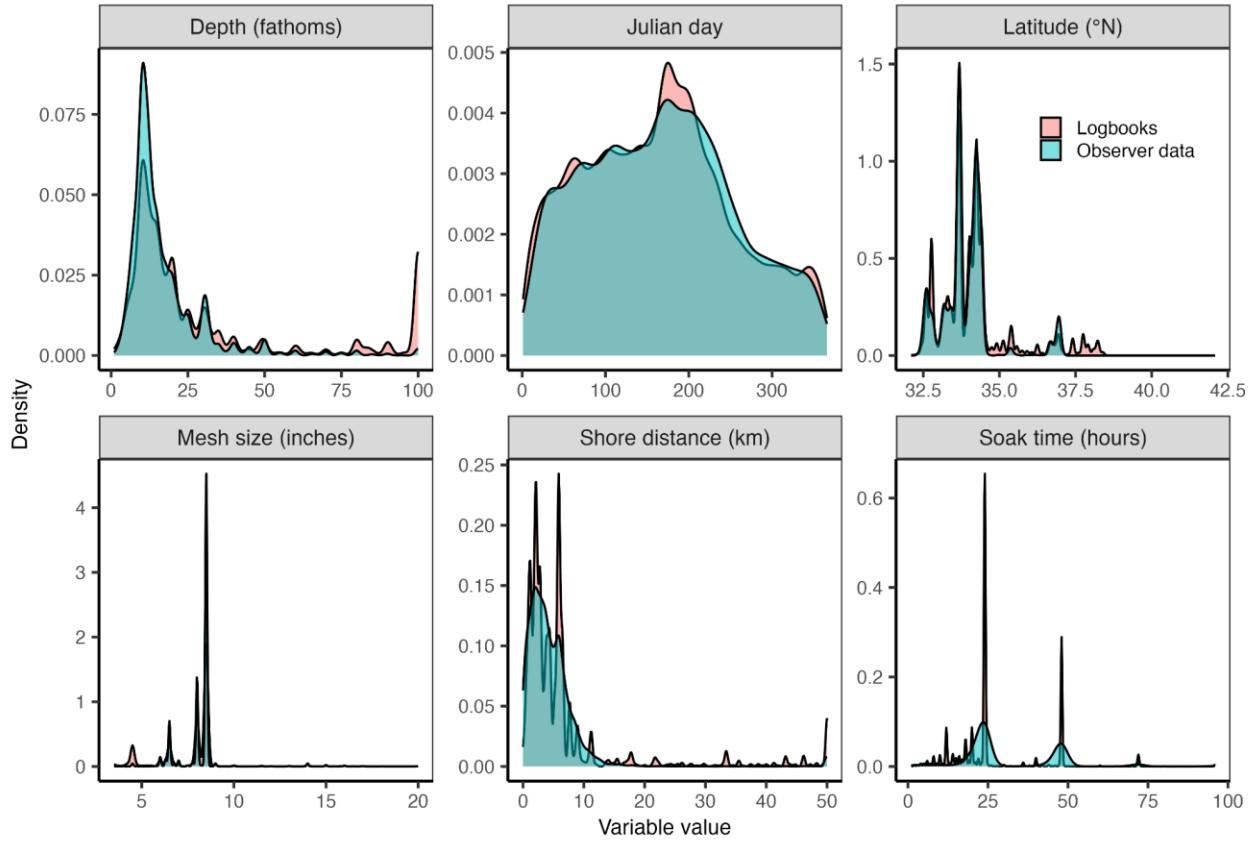


Fig. S5. Traits of the set gillnet sets documented in the logbook data. Panel **A** shows the level of completeness of gillnet logbook metadata. Panel **B** shows the distribution of reported depths. Panel **C** shows the distribution of reported soak times. The maximum reported soak time in the observer 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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1083 **Fig. S6.** Mesh size (inches) by target species in the logbook (red) and observer (blue) data. Since 1989,
1084 California halibut and Pacific angel shark can only be targeted using mesh sizes larger than 8.5 inches.
1085 White seabass are typically targeted using a minimum mesh size of 6.0 inches; however, a small amount
1086 of incidental take (<20% of catch and ≤ 10 individuals) in mesh sizes between 3.5 to 6.0 inches is allowed
1087 from June 16 to March 14 (14, § 155.10).



1088
 1089 **Fig. S7.** Comparison of key set traits in the logbook (red) and observer (blue) data. We used a two-sided
 1090 Kolmogorov-Smirnov test to confirm that the traits of the logbook and observer data could have come
 1091 from the same probability distribution (all p-values < 0.001) could have come from the same probability
 1092 distribution (all p-values < 0.001).

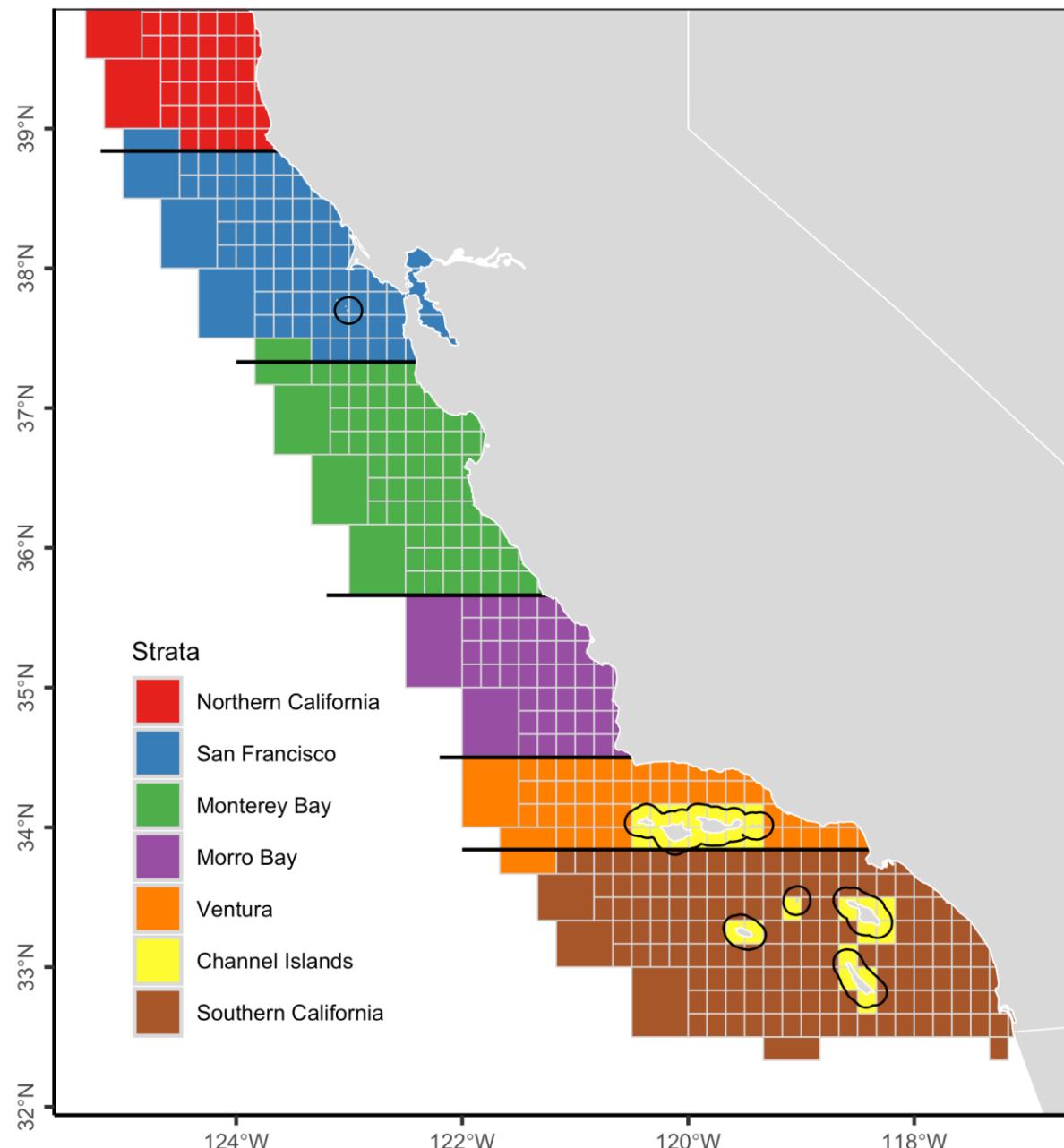


Fig. S8. The regional stratification scheme used throughout the analysis. The stratification scheme north of Point Conception was originally proposed by Diamond and Hanan (1986). The stratification scheme south of Point Conception was originally proposed by Julian (1993). The dark black lines around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

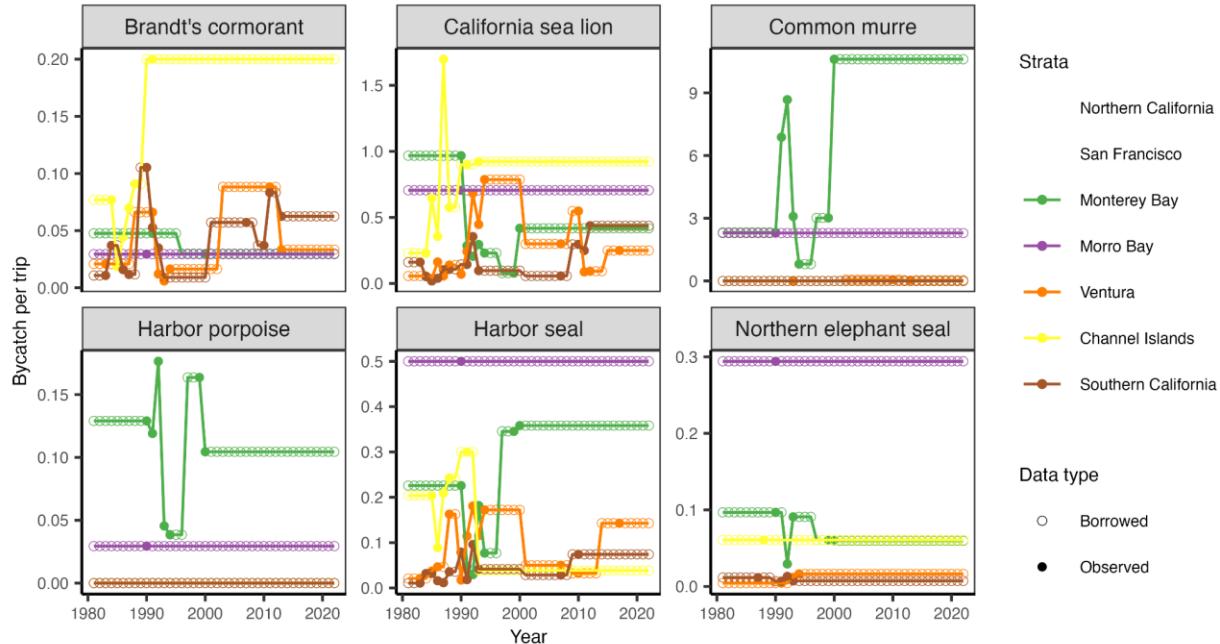
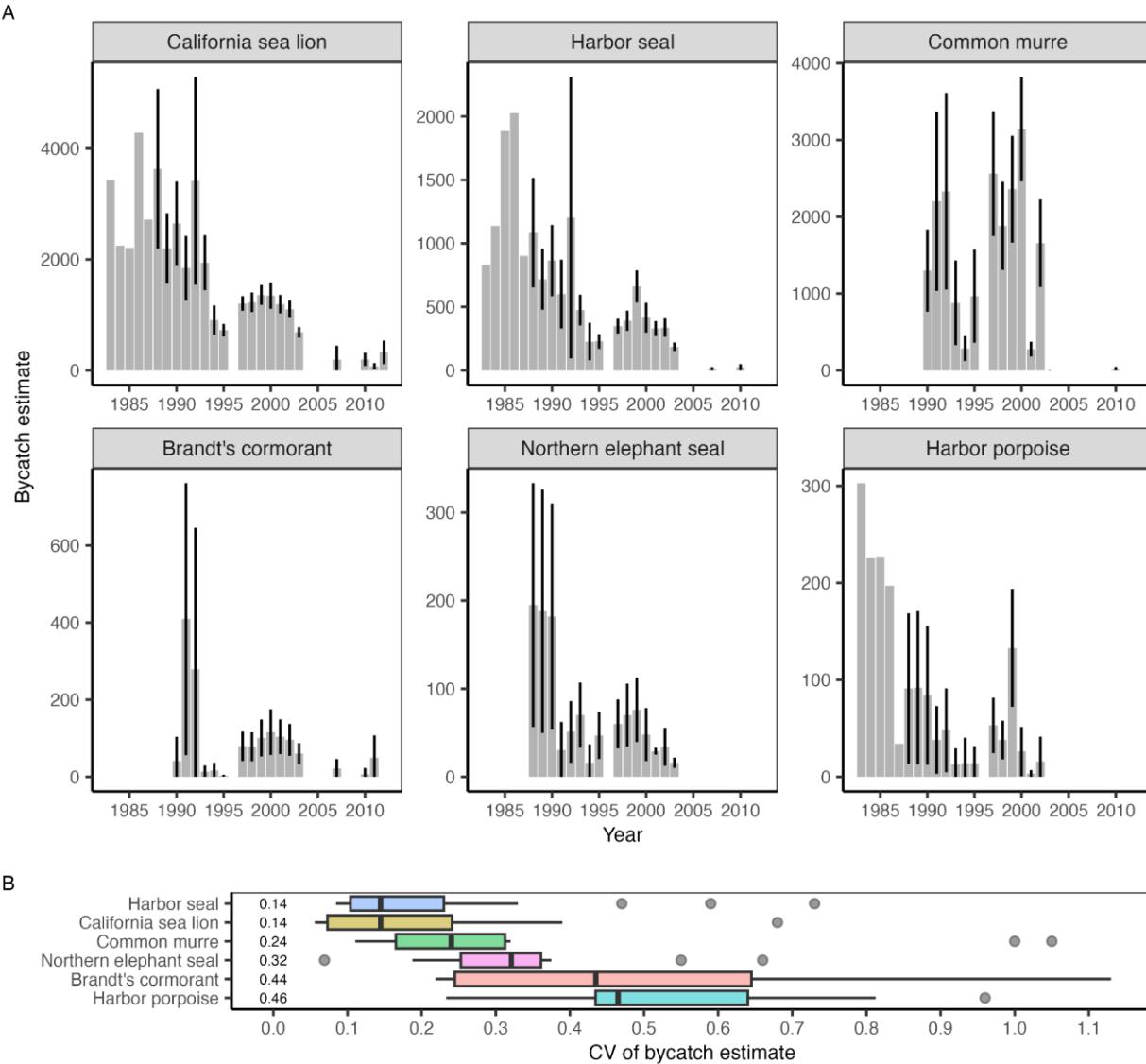


Fig. S9. Annual bycatch rates by species and regional strata as assumed in the ratio estimation analysis. Solid circles indicate years with observer data and open circles indicate years whose bycatch rates are borrowed from the closest year with observer data. See **Figs. 1A and S8** for maps of the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



1102
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
45 intervals. See **Table S1** for the sources of these estimates. In (B), the median CV of the bycatch estimates
46 for each species is printed on the far left. In the boxplots, the solid line indicates the median, the box
47 indicates the interquartile range (IQR; 25th to 75th percentiles), the whiskers indicate 1.5 times the IQR,
48 and points indicate outliers.
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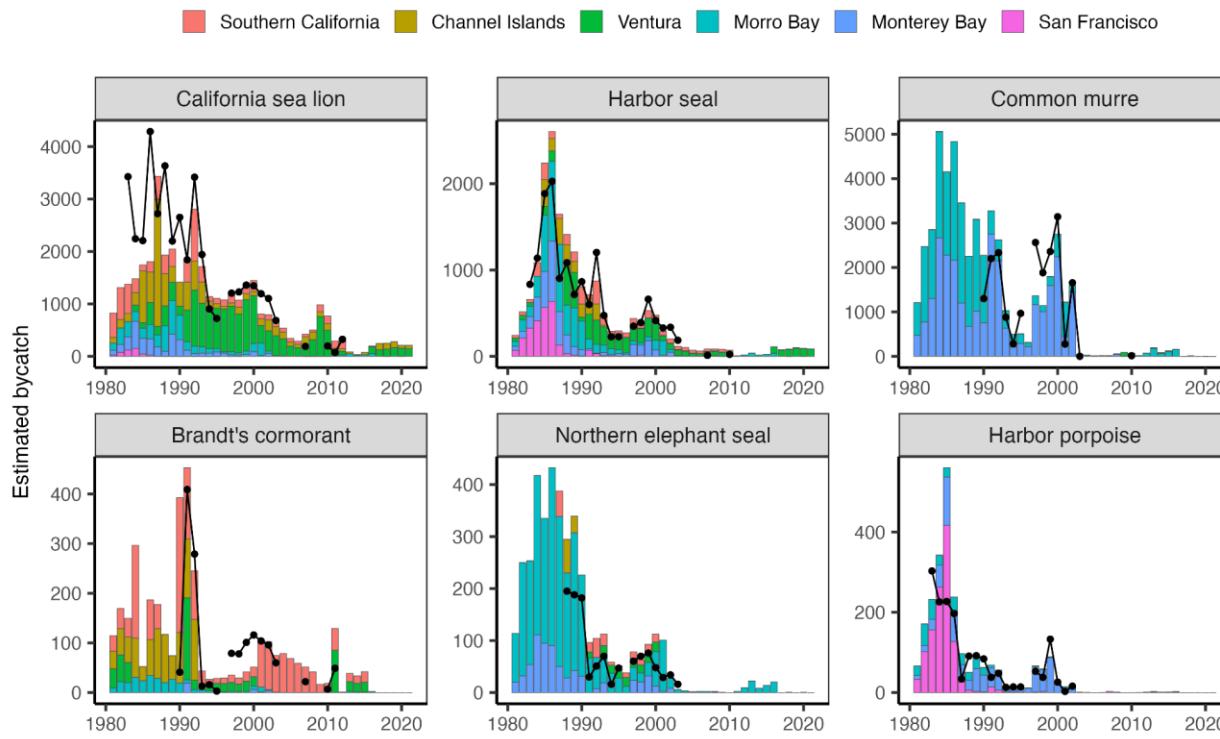


Fig. S11. A comparison of estimates of annual bycatch from our study (bars) and historical studies (points and lines). Potential reasons for these differences are explored in **Fig. S12**.

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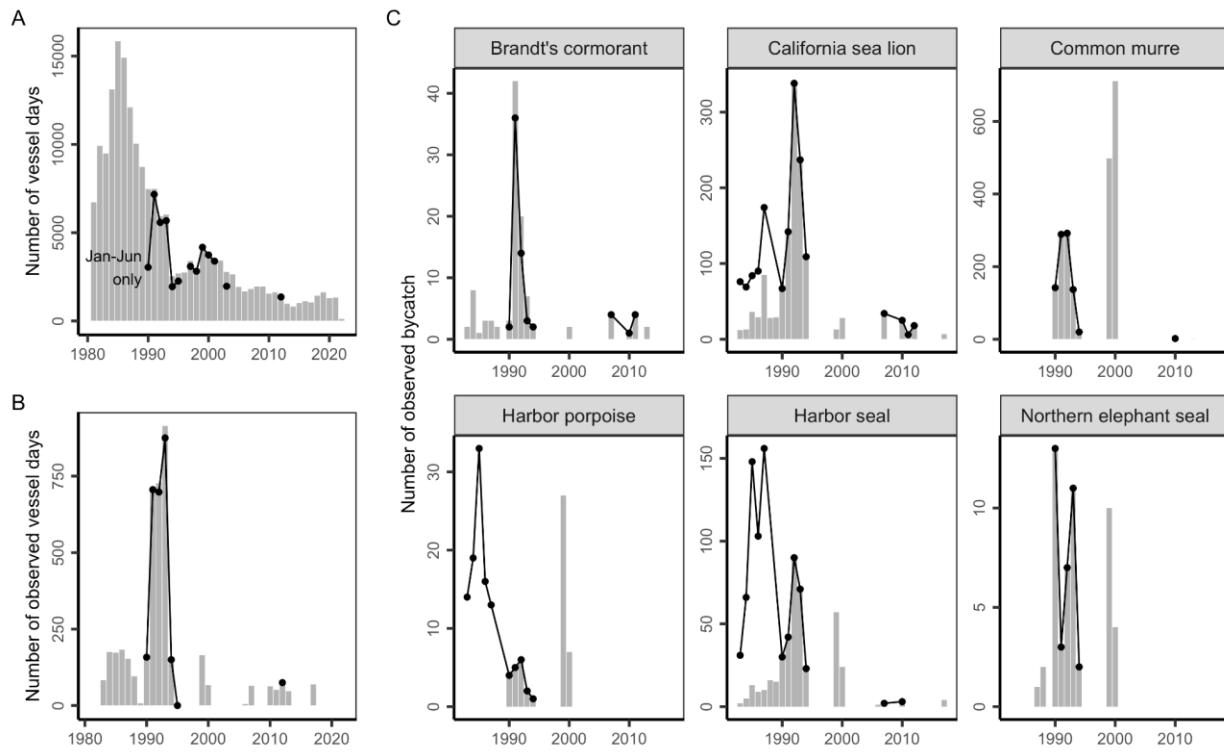
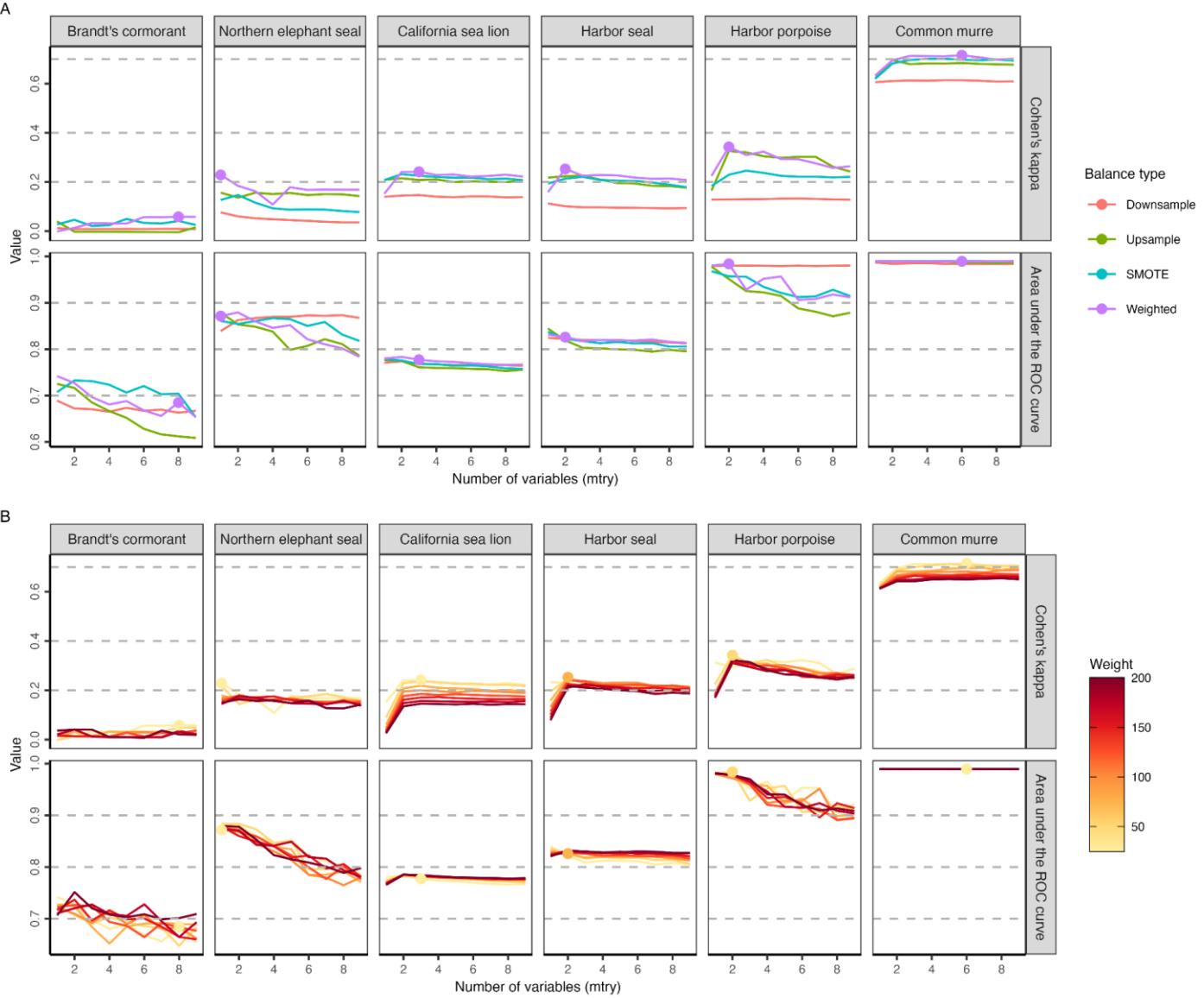
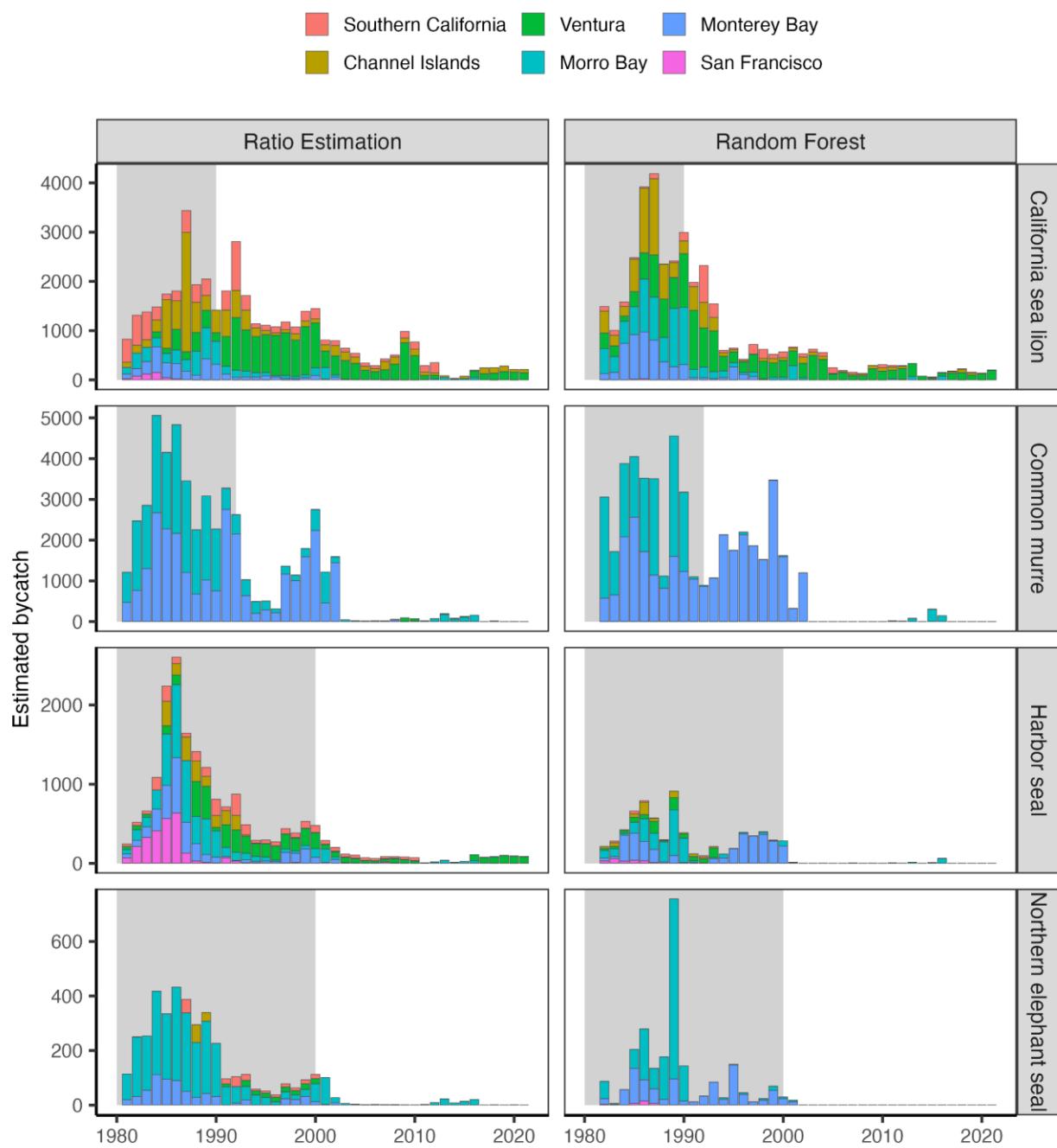


Fig. S12. A comparison of historical (A) fishing effort (number of vessel days); (B) observer coverage (number of observed vessel days); and (C) observer records (number of observed bycatch) derived in our analysis (bars) and reported in historical studies (points and lines). These time series represent 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 S1 for additional details on historical studies.



1120
1121 **Fig. S13.** A comparison of the model performance as measured through cross-validation on the training
1122 dataset between (A) balanced random forest modeling approaches and (B) weighted random forest
1123 modeling approaches for all candidate bycatch species (i.e., observer records > 50). We evaluated the
1124 model performance using Cohen's kappa and area under the receiver operator curve (ROC) and selected
1125 the model with the highest Cohen's kappa as the best performing model (labeled in the plot). We
1126 excluded Brandt's cormorant in the model prediction as their Cohen's kappa (<0.1) is too low to produce
1127 reliable predictions.



1128
 1129 **Fig. S14.** A comparison of estimated bycatch numbers between ratio estimation and random forest
 1130 stratified by regions (Fig. S8). The major differences between years are highlighted in gray.
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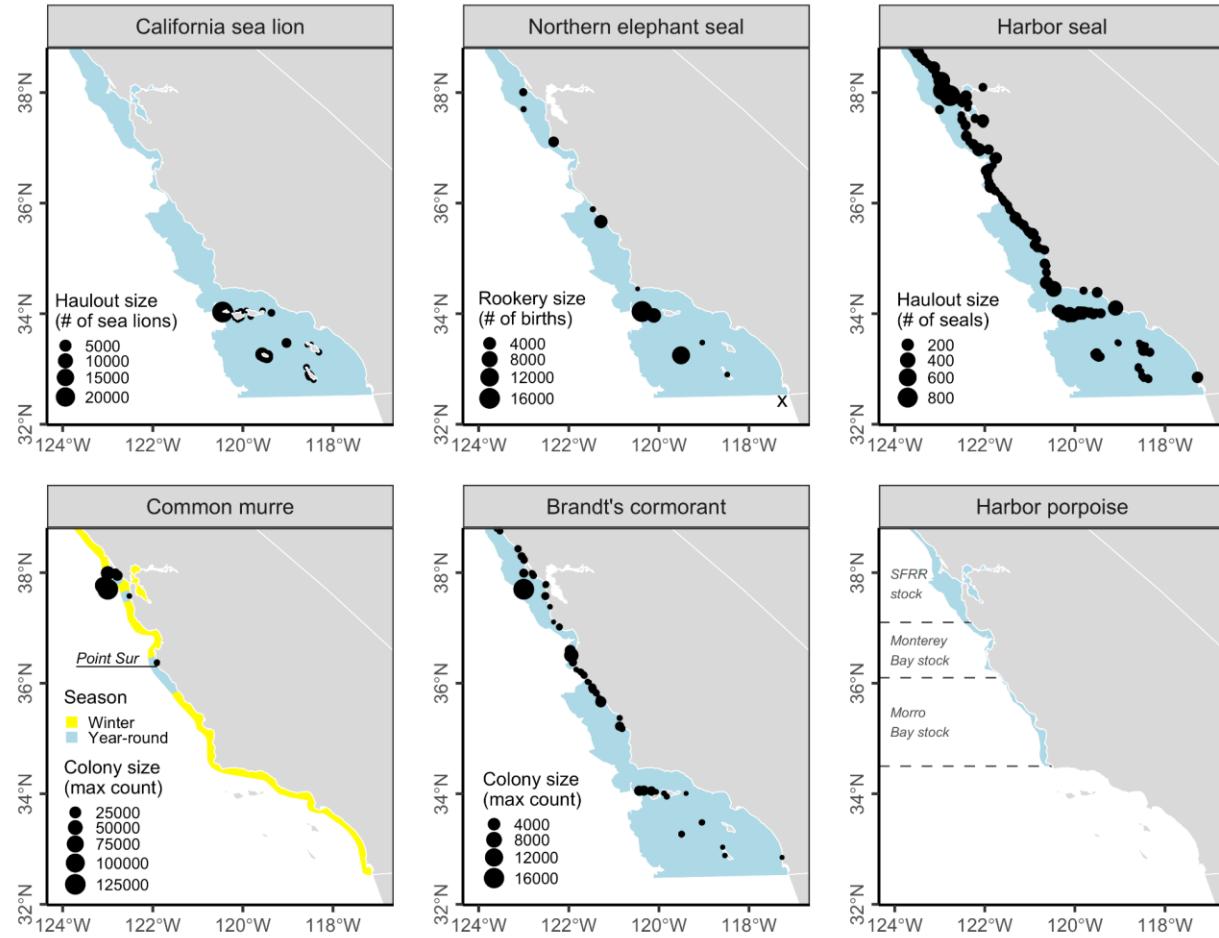


Fig. S15. Ranges of the six study species and information on haulout and colony size for selected species.

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).

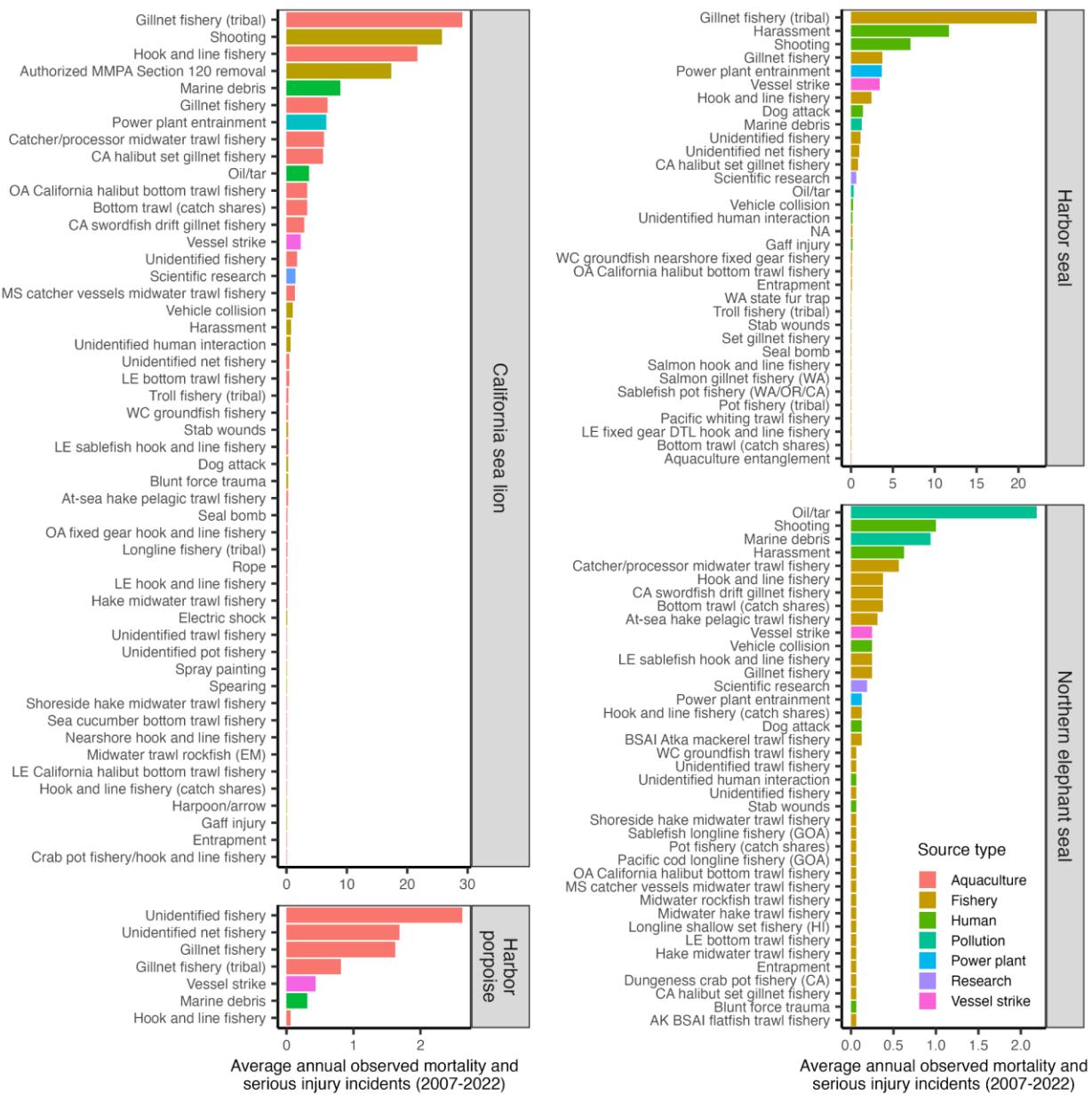


Fig. S16. Average annual observed mortality and serious injury incidents by source on the entire U.S.

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

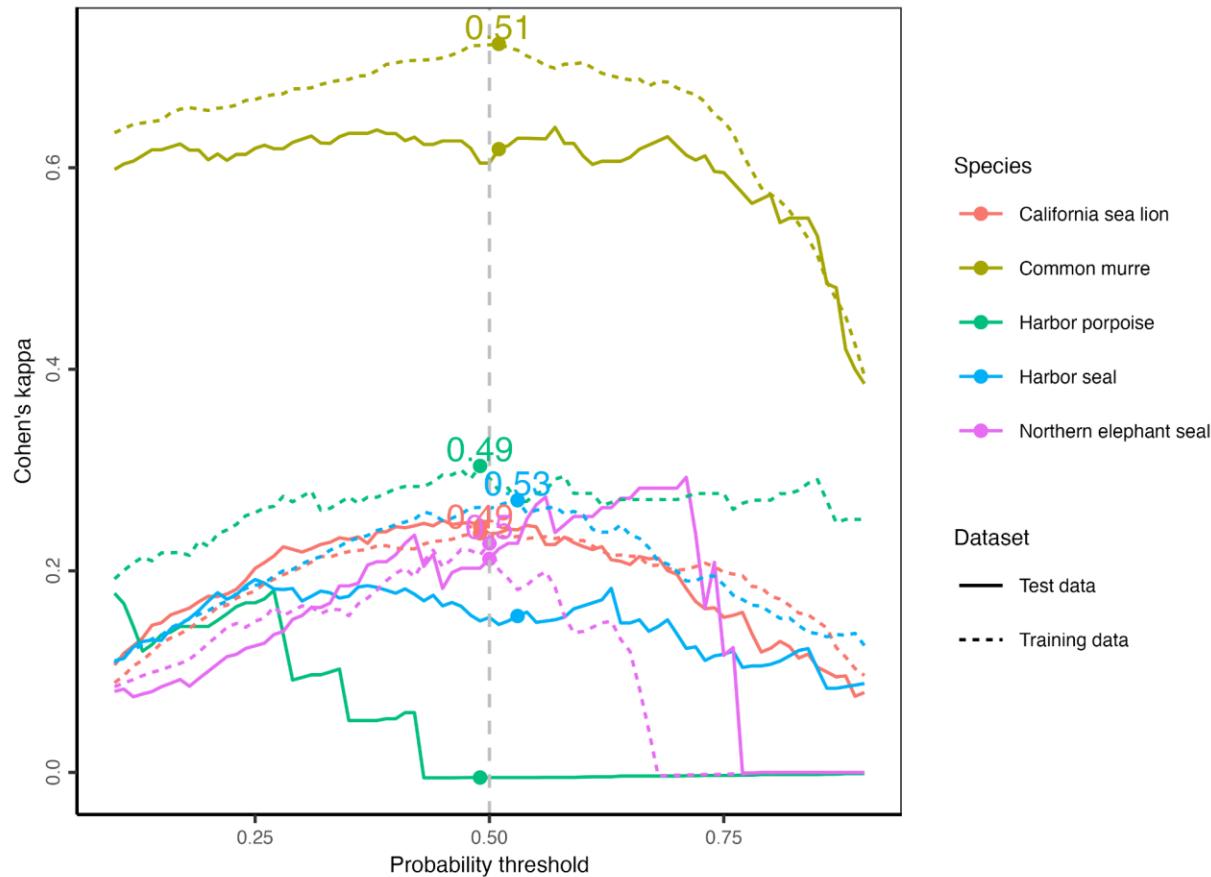


Fig. S17. Illustration of the methods used to select a probability threshold for classifying a logged set as having or not having bycatch. We selected the probability threshold that maximizes Cohen's kappa when applied to the training data as the optimal threshold (labeled in plot). We highlight the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