MULTIDISCIPLINARY APPROACHES TO MITIGATING FISHERIES BYCATCH

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MULTIDISCIPLINARY APPROACHES TO MITIGATING FISHERIES BYCATCH

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EDITORIAL
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Editorial: Multidisciplinary Approaches to Mitigating Fisheries Bycatch

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Editorial on the Research Topic

Multidisciplinary Approaches to Mitigating Fisheries Bycatch

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INTRODUCTION

This Research Topic of Frontiers, *Multidisciplinary Approaches to Mitigating Fisheries Bycatch*, was inspired by a workshop which gathered an international and multidisciplinary group of researchers from a wide range of institutions including government, academia, and industry. Featured speakers described relevant tools from environmental economics as well as the more traditional scientific and technical approaches to bycatch. Some participants shared their experiences with novel approaches that benefitted from engagement of industry (fishing, seafood and retailers) as well as other stakeholders such as NGOs and intergovernmental organizations (FAO, RFMOs, other intergovernmental organizations - IGOs).

Bycatch continues to be a key threat to many species, particularly protected species. In the case of marine mammals and sea birds, bycatch is the primary threat and, in some cases, can lead to extinction. Classic command-and-control, top-down fishery management has resulted in few success stories for addressing bycatch. The case studies and overall policy frameworks in the papers contained in this special edition seek to demonstrate that by expanding the toolbox and working throughout the process with the entire range of stakeholders and disciplines, there are practical solutions to addressing bycatch while allowing a viable fishing industry. Where appropriate, lessons can be learned from other environmental policy, including energy, forestry, water conservation and pollution.

The fundamentals of the multidisciplinary approach featured in the workshop are presented in a biodiversity mitigation hierarchy in a key overview paper in this Research Topic, notably Squires et al. This provides a framework for the other articles in this Special Research Topic, establishing four basic approaches to bycatch mitigation:

1. Private solutions
2. Direct or 'command-and-control' regulation
3. Incentive or market-based approaches and
4. Hybrid of direct and incentive-based regulation.

The articles feature some of the workshop case studies and provide additional, field-tested reports on the strengths and weaknesses of various innovations in multidisciplinary bycatch monitoring and mitigation.

The papers included in this Research Topic cover a broad range of:

Species: elasmobranchs, cetaceans, sea turtles, finfish and others, taking in some cases a multi-taxa approach.

Gear types: trap, trawl, gillnets, longlines, purse seine.

Fisheries and community structures: from small scale to industrial, and in some cases a multi-sized fleet.

Regions: South and North America Pacific and Atlantic, Indian Ocean, Western Pacific.

Tools: Bycatch quotas and credits, risk pools, gear modifications, gear deployment modifications, time/area closures, 'move on' measures, use of real-time information sharing. These can be voluntary or mandatory, regulatory or agreed within a fleet.

CASE STUDIES: LESSONS LEARNED

Among the most significant and novel insights offered by the case studies from the workshop and this special edition is a synthesis of "lessons learned". Specifically, successes and failures, and the situation-specific characteristics of each case study offer important insights into potential bycatch mitigation solutions, as summarized below. References to some of the articles in this Research Topic are noted in italics to provide specific examples. It should be noted however that several characteristics listed below appear in the case studies contained in most of these references.

Large-Scale, Industrial Fleet

- Individual Bycatch Quotas, transferable or non-transferable (Ballance et al.)
- Credit systems (cap-and-trade on bycatch, or penalty/reward either bycatch or effort) (Squires et al.)
- Performance standards for market access
- High-tech fishing for avoidance
- Incentives with penalties and rewards
- Rationalization to provide tools to address bycatch
- Risk pools and insurance (Holland and Martin)
- Hard cap on level of bycatch for whole fishery or vessel-level bycatch quotas (transferable or non-transferable)
- Addressing waste from regulatory discards (Watson et al.).

Small-Scale Coastal Artisanal Fishery

- Critical importance of a multidisciplinary approach, including social science
- Economic/social incentives focused on net income and livelihoods
- Community-based measures, working with local fleet to understand the context and ensure buy-in (Arlidge et al.)
- Investments in technology change and/or gear substitutions (Berninsone et al.)
- Offsets from industrial fleet or other activities with impacts on the same species (Gupta et al.)
- Rationalization *via* cooperatives to provide tools/incentives to address bycatch and create long-term value/stewardship
- Engaging the entire distribution chain to promote implementation of bycatch reduction measures/best practices
- Certification/access to markets for "eco-friendly" fishers
- Alternative livelihoods.

Lack of National Policy Measures

- Import restrictions (such as the US MMPA rule), if target catch is exported to the U.S. market
- Collective action/sectors; make use of existing cooperatives between local fishers and processors.

Characteristics of Bycatch Species

- Localized Ranges: Education, Adopt a species, regional collaboration
- Global/Transboundary Ranges: Collaborate (RFMO, FAO, IWC, CBD, CMS, CITES, other regional or global collaboration
- Time/area closures; dynamic or static (Smith et al., 2021)
- K-Selected: Risk pool with low quota, Limit/quotas, Ecosystem impacts on removals, Protected areas
- R-Selected: Constant rate of bycatch limit.

Community Structure

- Homogeneous: use social pressure/incentives, identify measures through bottom-up approach, political ecology (Bisack and Magnusson)
- Heterogeneous: develop common goals, use cooperation, social science and opportunities, political ecology.

Situation: Target Bycatch Species Are Highly Migratory (Shared, High Seas)

- Work with RFMO, FAO, etc. to develop guidelines, best practices, shared conservation and management measures to monitor and mitigate bycatch.
- Use measures that restrict market access to "good actors"
- Buyer/Processor-imposed requirements/incentives (private vs. government incentives)
- Ecolabels/certification (particularly where markets are globally integrated, e.g., tuna)
- Multilateral implementation of technical measures for gear and gear deployment; recognize that unilateral actions may be counter-productive.

Lack of Alternative Livelihoods

- Seek aid (national government, foundations, FAO) (Pakiding et al.)
- Relocation and retraining programmes with incentives (e.g., engage fishers in enforcement, observer or research jobs)
- Government subsidize/other support and Payments for Ecosystem Services

- Basic income support
- Buyouts of vessels, gear (Sanjuro-Rivera et al.).

Poor Compliance, Corruption

- Create incentives for self-enforcement within groups
- Rights-based measures
- Co-management
- Incentives, financial or others
- In-kind retraining/gear swaps vs buyouts
- Surveys or focus groups to understand economic/normative factors that influence non-compliance
- Affordable monitoring equipment, e.g., VMS (pings with GPS coordinates) to enforce spatial/temporal closures and camera surveillance.

Situation: Open-Access, Over-Capitalized Fishery

- Subsidies for bycatch-friendly gear development/deployment
- Community fishing cooperatives with performance standards, community coordination
- Rights-based management with transferability to reduce capacity; recognize role of RBM in effectively addressing bycatch.

Bycatch has not always been given attention in national and regional fisheries management bodies due to their focus on target species. As noted above, this is regrettable given that in many cases, bycatch injury and mortality is substantial and even leading to extinction for some species. This has been known since the early 2000s and has led to the concept of ecosystem-based fisheries management (Pikitch et al., 2004). Unfortunately, we are still far from ecosystem-based fisheries. Integrated cross-cutting approaches, adapted to each fishery, species and ecosystem context, have shown results, which it is important to disseminate. The research results presented in this Research Topic provide a wide and expanding array of potential solutions for monitoring and mitigating bycatch, even in the most challenging of circumstances. These options should reassure policy makers and stakeholders that it is possible to ensure a viable fishing industry whilst addressing the ecosystem impacts of fishing.

AUTHOR CONTRIBUTIONS

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

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ORIGINAL RESEARCH
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Bycatch Quotas, Risk Pools, and Cooperation in the Pacific Whiting Fishery

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The United States Pacific whiting fishery uses mid-water trawl gear to target Pacific whiting off the United States West Coast. The fishery is subject to sector-specific bycatch caps for Chinook salmon (*Oncorhynchus tshawytscha*) and several rockfish species (widow rockfish–*Sebastes entomelas*, canary rockfish-*Sebastes pinniger*, darkblotched rockfish–*Sebastes crameri*, Pacific Ocean Perch (POP)-*Sebastes alutus*, and yelloweye rockfish-*Sebastes ruberrimus*). Chinook bycatch can include fish from endangered populations and rockfish stocks were recovering from severe depletion though most are now rebuilt. Catch of these species is rare and uncertain, making it difficult for vessels to meet strict individual performance standards. Consequently the industry has developed risk pools in which bycatch quota for a group of vessels is pooled, but vessels are required to follow practices that minimize bycatch risk including temporal and spatial fishing restrictions. The risk pools also require vessels to share information about bycatch hotspots enabling a cooperative approach to avoid bycatch based on real-time information. In this article we discuss the formation and structure of these risk pools, the bycatch reduction strategies they apply, and outcomes in the fishery in terms of observed bycatch avoidance behavior and utilization of target species. The analysis demonstrates the ability of these fishers to keep bycatch within aggregate limits and keep individual vessels from being tied up due to quota overages.

Keywords: bycatch, fisheries, trawl, risk pools, Pacific whiting, individual bycatch quotas

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INTRODUCTION

Fishing gear, particularly trawl gear, often has limited selectivity and captures fish or marine fauna that are not the target of the fishery. United States law mandates that “conservation and management measures shall, to the extent practicable, (a) minimize bycatch and (b) to the extent bycatch cannot be avoided, minimize the mortality of such bycatch.” This, along with strict legal requirements to eliminate overfishing and rebuild overfished stocks, has put increasing pressure on fishery managers and the industry to reduce bycatch. Fishery managers and industry also face pressure from interest groups. NGOs have successfully harnessed consumer pressure to incentivize bycatch reduction, most famously with the development of the dolphin-safe label for tuna, which led to global changes in tuna fishing methods to reduce dolphin bycatch (Teisl et al., 2002; Ward, 2008). A discard ban in Europe was adopted after more than 650,000 signed a petition calling for “discards” to be banned following a series of programs by television chef Hugh Fearnley-Whittingstall (De Vos et al., 2016).

Traditional approaches to bycatch management rely either on command-and-control measures such as gear restrictions or closed areas, or bycatch caps imposed at the fishery level

(Hall and Mainprize, 2005). These approaches, while sometimes effective at reducing bycatch, are often costly and inefficient (Abbott and Holland, 2013). Bycatch caps at the fishery level can spur a race for fish (before bycatch caps are reached) that can actually increase bycatch rates and reduce the amount of target species harvested (Holland and Ginter, 2001; Abbott and Wilen, 2010).

The use of near real-time data to guide the spatial distribution of commercial activities, sometimes referred to as dynamic ocean management (Lewison et al., 2015), is increasingly used in fisheries around the world (Little et al., 2015). It can be particularly effective when it can rely on data from third party observers on vessels who can rapidly transmit reliable information. However, when compliance is voluntary or hard to enforce it is important that vessels are incentivized to use this information to reduce bycatch. Otherwise cooperation can break down, particularly if an overall bycatch quota can close down a fishery or area (Abbott and Wilen, 2010).

Alternative approaches that allocate bycatch quotas to individuals or cooperatives can reduce costs and increase effectiveness of bycatch avoidance by effectively harnessing the knowledge and skills of fishers (Abbott and Holland, 2013; Abbott et al., 2015; Holland, 2018). These approaches can also spur investment in technology and information sharing systems (Pascoe et al., 2010). However, individual bycatch quotas (IBQs) can create substantial financial risk for fishers if bycatch is highly uncertain and variable (Holland, 2010). Quota markets in these cases may fail to redistribute quota effectively resulting in underutilization of both bycatch and target quota (Holland, 2016).

Holland (2010) showed that this risk can be reduced though the formation of voluntary risk pools where groups of fishers pool bycatch quota. Cooperative approaches have other advantages. They can motivate, or require, fishers to share information about bycatch hotspots and ways to avoid bycatch which can help reduce the cost and increase effectiveness of bycatch avoidance (Holland, 2018). Holland and Jannot (2012) notes, however, that risk pools, like other insurance products, can create problems of moral hazard and adverse selection. Enabling pool members to draw freely from pooled bycatch quota reduces incentives for vessels to exert sufficient care to avoid bycatch if doing so is costly. Risk pools may also be subject to adverse selection attracting vessels with higher bycatch risk or lower bycatch quotas relative to their risk. Thus risk pool developers may need to mandate, monitor, and enforce best practices for bycatch avoidance and may want to charge premiums related to bycatch risk and perhaps some form of co-pay or deductible that maintains sufficient incentives for vessels to avoid bycatch.

Although there are many examples of incentive-based and cooperative approaches to managing bycatch (see Holland, 2018), there are few empirical analyses that describe the mechanics of how bycatch reduction is reduced in these systems and how effective it is. This paper contributes to the literature by detailing and evaluating bycatch reduction measures for a particular case study that provides broader insight about how the characteristics of the industry group and management affect the choice and effectiveness of bycatch reduction measures implemented by industry cooperatives. In this paper we describe and analyze strategies used by industry groups to manage bycatch of rockfish and Chinook salmon in the United States Pacific whiting fishery off the coasts of Washington, Oregon, and California. Prior to 2011 individual vessels had weak incentives to avoid bycatch since the bycatch cap was a common pool. After 2011 vessels either had individual incentives created by individual quotas or operated under risk pool agreements as part of a formal cooperative that dictated best management practices for avoiding bycatch. We describe the structure of the Pacific whiting cooperatives and risk pools and the practices they used to reduce bycatch. We then evaluate whether actual fleet behavior is consistent with these practices and discuss reasons for differences in bycatch avoidance practices across sectors. We find differences in the bycatch reductions measures used across the different sectors that are attributable to differences in the way the vessels operate and the control and monitoring capabilities that cooperative entities have over individual vessels.

STRUCTURE OF THE PACIFIC WHITING FISHERY COOPERATIVES AND RISK POOLS

The Pacific whiting fishery uses midwater trawl gear and catches are comprised almost solely of Pacific whiting, but the fishery does have a small incidental catch of rockfish and salmon. Since 2005 the fishery has been subject to sector-specific bycatch caps for Chinook salmon (*Oncorhynchus tshawytscha*) and several rockfish species (widow rockfish–*Sebastes entomelas*, canary rockfish-*Sebastes pinniger*, darkblotched rockfish–*Sebastes crameri*, Pacific Ocean Perch (POP)-*Sebastes alutus*, and yelloweye rockfish-*Sebastes ruberrimus*). Chinook bycatch can include fish from endangered populations, and rockfish stocks were recovering from severe depletion though most are now rebuilt. Before 2011 these caps were applied at the sector level and would potentially shut down the entire fishery sector when reached. There was a separate cap for the shore-based and at-sea sectors, but the at-sea cap was a combined cap for the mothership and catcher-process sectors.

In 2011, a catch share system was implemented in the fishery. Catch shares provide exclusive catch rights for a share of the total catch to individuals or groups. For the shore-based component of the fishery an individual fishery quota (IFQ) system was implemented with 42% of the total Pacific whiting quota allocated to individuals and firms based on catch history (**Table 1**). The IFQs for the shore-based Pacific whiting are integrated into a larger IFQ system for the groundfish trawl fishery which includes quotas for the rockfish species taken as bycatch in the Pacific whiting fishery as well as other species targeted by other gears. There are not individual bycatch quotas for Chinook salmon bycatch, however, only a sector cap. The at-sea processing sector of the fishery is managed with cooperatives - one for the at-sea processors which are allocated 34% of the Pacific whiting allowable catch, and one for vessels delivering to floating processors called motherships which are allocated 24% of the Pacific whiting allowable catch collectively.

TABLE 1 | Whiting industry and cooperative structure and bycatch avoidance practices.

Sector	Shore-based	Mothership	Catcher-processor
Type of Harvest Privilege	Individual quotas	Group quota	Group quota
Risk Pool Membership	Voluntary – Not all vessels participate in risk pool	All vessels required to participate in risk pool	Vessel/company bycatch allocations
Information Sharing	Yes	Yes	Yes
Night Fishing Restriction	Yes	Yes	Yes
Year-round Closed Areas	Yes	Yes	Yes
Hotspot Closures	Yes	Yes	No
Move-on rules	No	Yes	No
Test tows	No	Yes	No
Monetary Penalties for Non-compliance	Yes	Yes	Yes

Each cooperative receives allocations (or caps) of rockfish and Chinook salmon to cover incidental catch. The catcher-processor sectors has operated contractually under a cooperative since 1997, and the 2011 regulatory action simply formalized the management approach in regulation. The mothership sector, however, had been fishing their whiting allocation competitively but created a single cooperative in 2011. Within the mothership cooperative, shares of the whiting allocation were assigned to individual vessel owners, however, the rockfish allocations were pooled, and a cooperative "risk pool" approach has been applied to manage bycatch.

Both the shore-based and at-sea sectors of the Pacific whiting fishery share the same problem: how to ensure that bycatch limits are not reached and shut down the fishery before the quota of Pacific whiting has been harvested. However the catcher-processor and mothership sectors are allocated collective bycatch quotas while the shore-based sector vessels have individual quotas. The at-sea sectors must create a bycatch management approach that includes all active vessels and can impose rules on all those vessels contractually. In contrast, the shore-based sector can limit access to its risk pool but cannot impose rules on those who don't join it. In the end, both groups implemented cooperative approaches that limit individual risk by pooling bycatch quota but still incentivizes bycatch avoidance by individuals. While all active vessels in the mothership sector are part of the mothership cooperative this is not the case for the shore-based fleet. The shore-based group must balance incentives to get more vessels to join with rules that ensure that riskier members are either excluded, controlled once in the risk pool, or compensate the rest of the pool for the additional risk they add. As the manager of the shore-based risk pool was quoted:

> The mothership co-op's task is to "individualize accountability while managing a common quota." the shore-based co-op's task is to "collectivize risk while maintaining individual accountability" (Blikshteyn, 2016).

In 2011 the mothership sector of the Pacific Whiting fishery formed a single cooperative that included the owners of 37 catcher vessels endorsed for operation in the mothership sector. The cooperative receives an allocation of Pacific whiting each year as well allocations of several rockfish species (POP, darkblotched rockfish, canary rockfish, and widow rockfish). The cooperative's internal contract (Fraser, 2011) allocates shares of Pacific whiting to each of the catcher vessels in proportion to the contribution to the cooperative's allocation made by NMFS (which is on the basis of the whiting catch history assigned to the Cooperative by its members). Individual allocations are transferable within the cooperative allowing for consolidation in the harvest operations, and many of the vessels do not fish in a given year. The number of vessels actually fishing ranged from 14 to 19 between 2011 and 2015. In recognition of the uncertainty and lack of control over bycatch, the cooperative pools the bycatch quota. The cooperative divides the whiting allocation into as many as four sub-annual pools with various start dates. Members must decide in advance how much of their whiting quota to allocate to each pool. Each pool then receives a share of the bycatch allocations in proportion to the proportion of whiting quota allocated to it. The individual vessels maintain their rights to the whiting quota submitted to the sub-annual pool, but the bycatch pool is a common pool. The co-op Agreement specifies that if a pool reaches its share of the bycatch prior to harvesting its whiting allocation, the members of the pool must cease fishing. Unused bycatch from each pool, other than the last pool of the year, is carried over to the next pool.

To ensure that vessels are avoiding bycatch, the mothership risk pool agreement implements a number of operational rules. These include: precautionary closures of past bycatch hotspots and in-season hotspot closures; restrictions on fishing at night (when the bycatch species tend to move up off the bottom increasing potential bycatch); and mandatory relocation of the fleets delivering to each mothership if a fleet's bycatch rate exceeds specified rates. Relocations are triggered either by 3 days rolling averages exceeding 125% of a base rate for a species (e.g., the ratio of total bycatch allocation to Pacific whiting allocation) or if the fleet bycatch rate in a single day is twice the base rate. Perhaps most importantly, the mothership cooperative requires members to share spatially explicit information about both whiting catches and bycatch. This is done through a company called Sea State, Inc., which receives this information directly from the fishery observer program (which places observers on all vessels) and processes it and relays it on to the fleet daily. Sea State identifies bycatch hotspots and designates time-area closures which vessels are obligated to avoid. The cooperative's manager can use observer data to monitor compliance with closures and other risk pool rules.

The bycatch avoidance practices of the catcher-processor sector are less clear. The catcher-processor cooperative produces an annual report that indicates that the cooperative contracts with Sea State, Inc., to monitor bycatch rates and authorizes it to impose in-season closures of bycatch hotspots, but there are no indications in these reports that this authority has been used. Reports indicate that individual vessels have kept bycatch with individual vessel allowances, and both bycatch and Pacific whiting catch are reported for each individual catcher-processor.

The shore-based sector risk pool also requires members to follow specified bycatch avoidance rules including prohibitions on night fishing and adherence to pre-season and in-season time and area closures (personal communication Dave Frazer, Risk Pool Manager). In addition it implemented a system of premiums, deductibles, co-payments and penalties that attempt to limit adverse selection and moral hazard. Risk pool members are required to make a minimum contribution of bycatch quota to the risk pool proportionate to the Pacific whiting quota they intend to fish (which they must declare in advance). The minimum contribution is the members' pro-rata share of the aggregate quota of constraining canary rockfish, widow rockfish, darkblotched rockfish and POP allocated to the member vessels relative to their aggregate whiting quota. The minimum pool commitment for yelloweye is the average amount of yelloweye quota allocated to a permit. For each of the bycatch species, other than yelloweye, 50% of the individual's commitment remains in a restricted account for their access and 50% goes into a reserve account for the risk pool. For yelloweye, which has a very small total quota, 100% goes into the risk pool reserve account. Members must first cover their own bycatch out of their own restricted reserve account but when that is exhausted can draw from the risk pool reserve account provided their average bycatch rate has not exceeded 120% of the base rate (the ratio of all members' bycatch quota for that species to whiting quota held). If that rate is exceeded, the vessel is required to stand down for 7 days or make an additional contribution of bycatch quota to the risk pool to bring their rate (of bycatch covered by the pool) down below 120% of the base rate. There are stricter criteria and longer stand-downs for yelloweye bycatch. Members who use bycatch in excess of what they contribute to the risk pool reserve account (and their own restricted account) can be compelled to pay an amount per pound of bycatch determined by the risk pool board. A funding mechanism enables the board to purchase additional bycatch quota to supplement the risk pool reserve account if necessary. Spatially explicit information on catch and bycatch for shore-based pool members is also collated and distributed by Sea State, Inc., (the same company doing this for the at-sea sector), but there is a delay in distributing this information because it relies on information not available until after vessels land. Captains are asked to enter preliminary data before starting a new trip. In addition to the individual incentives to reduce bycatch rates created by the cooperative's rules, individual vessel are also subject to limits on how much an individual vessel can catch of any species during the year regardless of whether they can acquire quota to cover it. Exceeding these vessel catch limits can result in the vessel being shut down for the remainder of the year or even longer.

ANALYSIS OF FLEET AND VESSEL BEHAVIOR AND OUTCOMES

Data

We use observer data collected by the West Coast Groundfish Observer Program and the At-Sea Hake Observer Program and fish tickets (landings records) compiled by PacFIN to evaluate outcomes and changes in behavior for the different fleets in the Pacific whiting fishery. This data includes tow-level information on catch, the location and time of the tow, the vessel, etc. For the mothership sector there is accurate information on the bycatch for each individual tow since observers sample from it when it is offloaded to the mothership. For the shore-based sector, vessels often make only 2–3 tows on a trip and the catch is mixed in the refrigerated seawater hold. Estimates of the catch of whiting per tow are available, but the bycatch is typically only known at the trip level and is documented in fish tickets.

Aggregate Bycatch and Bycatch Rates Over Time

Figure 1 shows bycatch rates per 100,000 pounds of whiting from 1994 to 2016. These rates are erratic and do not show any clear trends or changes following implementation of catch shares with exception of the shore-based sector for which bycatch rates for rockfish appear to have increased in recent years - though we do not undertake a formal times-series analysis to look for changes in trends or their causes here. Increases in catches of rockfish by the shore-based sector, particularly for widow rockfish, likely reflect increasing availability of quota pounds resulting from increases in the annual catch limits for those species. Widow rockfish was declared rebuilt in 2012, and the total annual quota allocation to the IFQ nearly tripled in 2013 and increased another 50% in 2015. Some vessels began targeting widow rockfish in 2015, and incentives to avoid it would have been low. Quotas for canary and darkblotched rockfish and POP remained small through 2016, but 40% or more of the quota pounds for these stocks in the shore-based sector went unused in all years except for Canary rockfish in 2015 when the quota was fully utilized. Vessels could lease quota pounds for these species to cover bycatch though quantities of quota available for lease were small with many fishers holding on to the small amount they had in case of unexpected bycatch. Quota pound prices exceeded ex-vessel value (price paid to vessels) for canary and darkblotched rockfish and POP (**Table 2**) suggesting they were seen as a constraint or at least potential constraint, despite the consistent surplus quota at the end of the year (Holland, 2016). Canary and darkblotched rockfish and POP have now all been rebuilt and their abundance appears to have increased substantially since 2011 which would also partially explain increases in bycatch rates.

In contrast to the shore-based sector, the allocations of rockfish species to the at-sea sectors did not increase over this period in proportion to total allowable catches and abundance, and they maintained very low bycatch rates for them. Industry representatives have also stated that they face a trade-off in avoiding Chinook salmon and some rockfish species which can cause one to rise if avoidance of the other is seen as a higher priority. These factors and the volatile and uncertain nature of bycatch makes identifying changes in bycatch rates associated with particular policies or behaviors problematic.

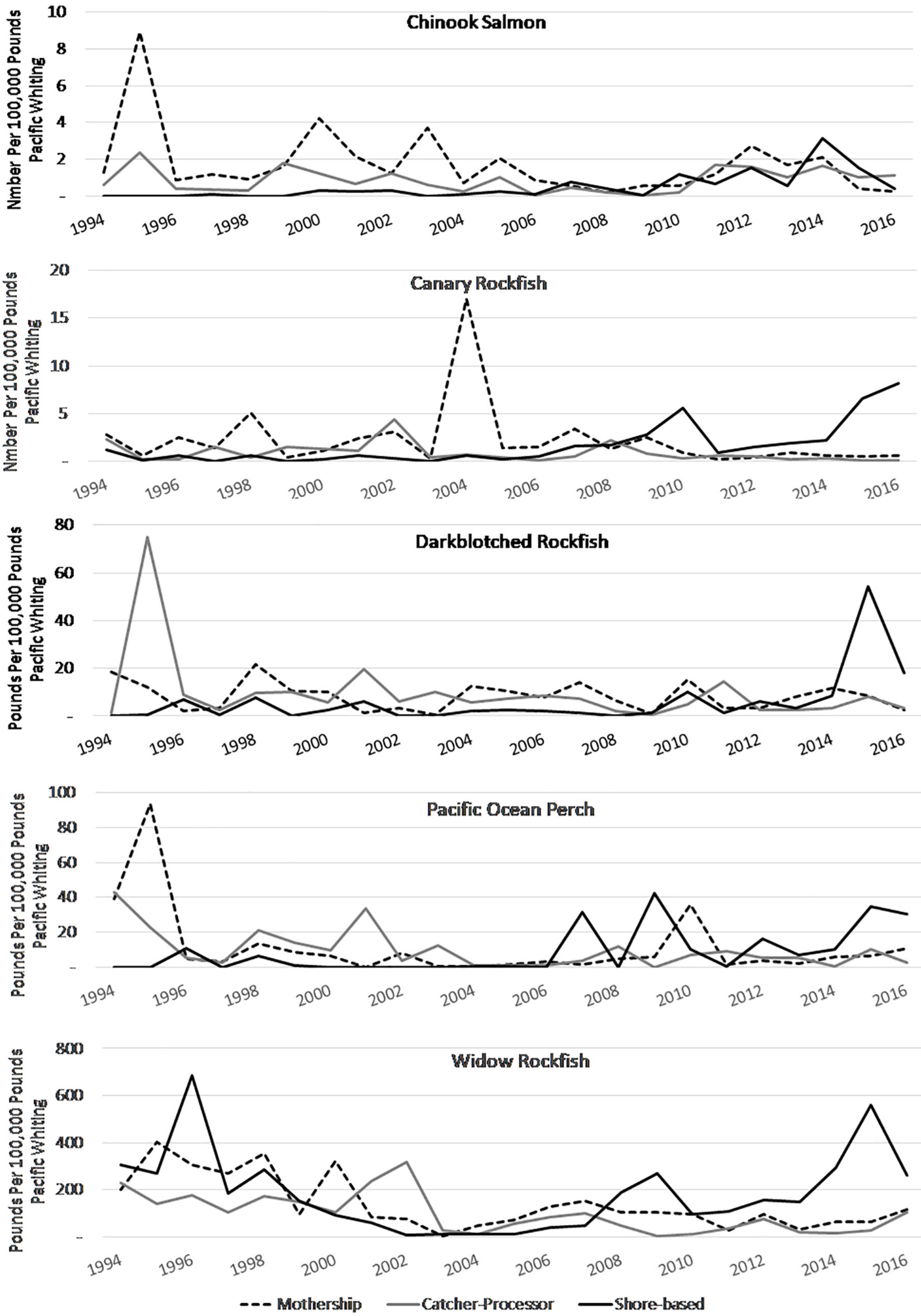

FIGURE 1 | Bycatch rates for different sectors of the US Pacific whiting fishery.

TABLE 2 | Ratio of quota pound price/ex-vessel price.

IFQ Species	2011	2012	2013	2014	2015	2016
Canary rockfish	2.24	2.91	6.18	3.88	2.05	2.75
Darkblotched rockfish	0.84	0.45	1.11	2.43	1.15	1.21
Pacific ocean perch	0.28	–	1.58	2.30	1.14	1.17
Widow rockfish	1.01	0.81	1.18	0.53	0.37	0.36

Changes in Behavior

Closed Areas

While it is difficult to determine whether or how much bycatch avoidance affected realized bycatch rates, we can look directly at whether and how fishing behavior was adapted to reduce bycatch. Little information is available about bycatch avoidance activities in the catcher-processor sector and it is not clear that behavior changed significantly after 2011. The catcher-processor sector had already been operating under a cooperative since 1997 and had been using Sea State to facilitate bycatch avoidance over that period.

In contrast, the mothership sector transitioned in 2011 from a race-for-fish to operation under a single cooperative with a highly structured agreement that included a number bycatch avoidance practices. The mothership sector of the fishery and the shore-based whiting risk pool all implemented self-imposed closed areas in locations where high bycatch rates had been experienced in the prior years. The mothership sector designated 9 precautionary area closures, totaling nearly 2000 km^2 which have been closed year round since the cooperative began operation in 2011 (Fraser, 2011). The mothership sector also authorized the risk pool managers to implement in-season closures in bycatch hotspots. Annual reports from the mothership cooperative indicate that these were used only a few times. The boundaries of the voluntary closures are not reported, thus it is not possible to verify whether the fleet fully complied with its own closures, but the cooperative's annual reports indicate that there have been no violations of the cooperative's agreements. The mothership cooperative manager could also shut down fishing once the seasonal pool of bycatch was exhausted. This occurred for at least one pool in most years.

The shore-based whiting risk pool also imposed year-round closures determined annually (Dave Frazer, personal communication November 2017), but the number and extent of closures is not available. There is no formal reporting on use of in-season closures by the shore-based risk pool but the risk pool manager indicates they have been used rarely if at all. However, vessels are required to share information about bycatch enabling others to voluntarily avoid these areas.

Night Fishing Restrictions

Both the mothership and shore-based cooperatives restrict night fishing during part of the year. We compared the proportion of tows taking place at night pre- and post-catch shares. For the mothership sector, between 1999 and 2010, 6.9% of tows took place between 10 PM and 5:30 AM while only 0.17% of tows took place in that time window between 2011 and 2014. For the shore-based vessels we did not have data on time-of-day prior to 2011, but between 2011 and 2014, only 0.58% took place during the night before September, 1.3% after September, and 0.89% overall. The catcher-processor sector apparently does not prohibit night-time fishing and there has been little change since 2011 with 25.3% of hauls taking place at night from 1999 to 2010 and 27.4% from 2011 to 2014.

Distance Moved

Move-on behavior is another important way in which vessels can reduce their exposure to bycatch. This method of spatial avoidance is closely related to area closures. Vessels experiencing a bycatch event can move to a new area for their next tow. When vessels learn of a bycatch event is an open question. In the case of the shore-based fleet, vessels may learn immediately on pulling their nets out of the water. For the at-sea fleets (both catcher-processors and motherships), there may be some delay before they learn the contents of their last haul, potentially until after the next haul has already begun.

We examined the contents of the previous haul on the distance moved between hauls. For motherships, we use the daily centroid of hauls for each motherships fleet of catcher vessels and model distance moved between days rather than between hauls. We utilize haul-level data from 1999 to 2014 and use vessel and year fixed effects in a generalized linear model with heteroscedasticity-robust standard errors. The log of distance moved in nautical miles since the last haul for haul h, by vessel i, in year y is the dependent variable, and explanatory variables include quantities of whiting and rockfish caught on the last haul (in metric tons) and an interaction of rockfish catch with the catch share categorical variable to determine if the distance moved in response to rockfish catch was greater once the catch share system began.

$$\begin{aligned} Dist\_move_{h,i,y} &= \alpha_{i,y} + \beta_1 whiting_{h-1,i,y} + \beta_2 rockfish_{h-1,i,y} \\ &\quad + \beta_3 (catchshare_y * rockfish_{h-1,i,y}) + \varepsilon \end{aligned}$$

For each of the results, we see the expected behavior with respect to the target species catch: additional tonnage of whiting leads to shorter distances moved. In other words, vessels who have found the fish continue fishing in the same area. Move-on distance decreases by 0.62% per ton of whiting caught in the shore based fleet ($p < 0.00$), 1.00% per ton in the mothership fleet, and 1.78% per ton in the catcher processor fleet (**Table 3**).

The central question as far as bycatch is concerned is whether move-on distances increase when bycatch is encountered. We are particularly interested in whether these distances changed under the catch-share or cooperative programs described above. In the shore-based fishery, there is clear evidence of move-on behavior after a bycatch event. An additional ton of bycatch caught corresponds with an 11.5% increase in distance moved between hauls (**Table 3**). After the implementation of the catch share program, the point estimate suggests that this reaction to bycatch diminishes, consistent with the decreased risk exposure afforded by the catch sharing program, but these results are marginally statistically insignificant ($p = 0.11$).

In the mothership fleet, the cooperative implementation leads to clear evidence of increased move-on behavior. Absent the cooperative, an additional ton of bycatch correlated with a 20.3% increase in distance moved (p = 0.00) (**Table 3**). After the cooperative implementation, distance moved increases even more, by 59.4%, per ton of bycatch (p = 0.03). This result is consistent with the aim of the cooperative in increasing vessel-level bycatch avoidance incentives.

For the catcher-processor fleet, each ton of bycatch increased distances between hauls by 9.3% ($p < 0.00$) (**Table 3**). Post-2011 catcher-processor cooperatives increased this response, with each ton of bycatch caught increasing distance moved by 61.9% (p = 0.02).

In short, all sectors exhibit move-on behavior after bycatch events, and there is evidence of increased move-on behavior (e.g., moving further on average) after the institutional change for the at-sea fleets (mothership and catcher-processor).

HAUL DURATION

The final margin of bycatch avoidance we examined in the fishery was haul duration. Vessels can survey a new area for both target species and bycatch species by performing test tows. The distribution of distance moved is relatively smooth and does not follow a bi-modal distribution, so it is difficult to ascertain when vessels would consider an area "new." We designated new areas as those occurring either as the first tow of a trip for the shore-based fleet, the first tow after a period of inactivity for the at-sea fleets, and after a move-on distance above the 60th percentile for either fleet. The results presented below are robust when using either higher or lower thresholds for move-on behavior.

Again we use a utilize haul-level data from 1999 to 2014 and use vessel and year fixed effects in a generalized linear model with heteroscedasticity-robust standard errors. The regression have haul duration in minutes for haul h, by vessel i, in year y as the dependent variable and explanatory variables include: a categorical variable taking a value of one if the vessel was fishing in a new areas since the last haul (as described in the prior paragraph); new area interacted with a categorical variable taking a value of one if the haul took place in the catch share period (later than 2011); variables for the quantities of whiting and rockfish caught on the last haul, and an interaction of rockfish catch with the catch share categorical variable to determine if the decrease in duration of the subsequent tow in response to rockfish catch was greater once the catch share system began. We also include a categorical variable for first haul of the day for catcher processes only.

$$\begin{aligned} &Duration_{h,i,y} \\ &\quad = \alpha_{i,y} + \beta_1 newarea_{h,i,y} + \beta_2 (newarea_{h,i,y}{}^{*} catchshare_y) \\ &\quad + \beta_3 whiting_{h-1,i,y} + \beta_4 rockfish_{h-1,i,y} \\ &\quad + \beta_5 (catchshare_y{}^{*}\ rockfish_{h-1,i,y}) + \beta_6 firsthaul_{h,i,y} + \varepsilon \end{aligned}$$

For the shore-based fleet, we find that new areas resulted in 17-minute reductions in typical tow duration before implementation of the catch share program ($p < 0.00$) with an average tow length of 268 min (**Table 4**). This effect disappeared after 2011 (p = 0.02). The amount of whiting catch in the last haul results in shorter tows, though this may reflect the vessel capacity and not test tow behavior (most trips consist of three or fewer tows). Tows after bycatch encounters are 7 min shorter per ton (p = 0.06) before catch shares, increasing to 20 min shorter per ton (p = 0.02) after 2011.

In the mothership fleet, we see a similar effect. Mean tow duration over the sample was 202 min. New areas are associated with 32-minute shorter haul duration ($p < 0.00$) before 2011 (**Table 4**). Before 2011, a ton of rockfish catch was associated with a 14 min increase in the subsequent tow duration ($p < 0.00$). The point estimate suggests that this effect was reversed and bycatch events led to shorter subsequent hauls post-2011, but these results are not statistically significant (p = 0.16).

For catcher-processors, the evidence of decreased tow duration after a bycatch event is strongest in both practical and statistical terms. Typical haul duration in this fleet was 121 min over the sample (**Table 4**). The first tows were typically 16 min shorter, increasing to 23 min shorter after catch shares were

TABLE 3 | Fixed effects models of distance moved by sector.

Explanatory variable	Shore-based	*p*-value	Mothership	p-value	Catcher-processor	p-value
Last Haul Whiting (metric tons)	−0.0062	<0.001	−0.0100	<0.001	−0.0178	<0.001
Last Haul Rockfish (metric tons)	0.1145	<0.001	0.2032	0.002	0.0928	0.002
Rockfish (metric tons) × Catch Share Program	−0.1090	0.109	0.3909	0.025	0.5270	0.025

Dependent variable is log(distance moved nautical miles). The constant is not shown as this is a fixed effects model and constants vary by vessel.

TABLE 4 | Fixed effect models of haul duration by sector.

Explanatory variable	Shore-based	*p*-value	Mothership	*p*-value	Catcher-processor	*p*-value
New Area	−16.73	<0.001	−31.85	<0.001	−16.36	<0.001
New Area × Catch Share Program	16.60	0.022	0.82	0.955	−6.69	0.071
Last Haul Whiting	−1.56	<0.001	−0.36	<0.001	0.42	<0.001
Last Haul Rockfish	−7.28	0.061	13.79	0.002	−1.70	0.084
Rockfish × Catch Share Program	−12.91	0.024	−36.79	0.162	−40.69	0.002
First Haul of the Day	*n.a.*		*n.a.*	–	14.84	<0.001

Dependent variable is haul duration in minutes.

implemented in 2011. A ton of rockfish was associated with a 2 min decrease in tow duration before 2011, increasing to a 42 min decrease after 2011. Tows were also shorter if there was rockfish caught on the previous tow, particularly after the catch share program was implemented ($p = 0.00$).

DISCUSSION

The three sectors of the Pacific whiting fishery all face a common problem of limiting bycatch of rockfish and Chinook salmon while targeting Pacific whiting. All three have implemented cooperative approaches to managing bycatch with many similar practices, but also some distinct differences that reflect differences in their operational characteristics, homogeneity of membership, and ability to exercise centralized control over vessel operations.

A primary advantage of risk pools (in addition to access to quota) is information sharing that enables individuals to make better decisions about when and where to fish. All three sectors use a private third party provider, Sea State, Inc., to share and collate information about bycatch hotspots to enable near real-time avoidance of these areas. All three sector have imposed rules upon their members that limit fishing at particular times or places, but there are some differences in the rules imposed that reflect differences in the flexibility of the vessels. For example, move on rules and test tows are not required by the shore-based risk pool which makes sense since vessels average only two tows per trip and vessels trip duration is strictly limited once the first fish is brought on board since quality deteriorates rapidly. In contrast the mothership sector which has fleets of vessels fishing cooperatively with a mobile floating processor can more easily implement test tows (i.e., by having one vessel make an initial tow before others begin fishing) and can take time to move locations without fear of spoilage since the fish is being processed at sea.

The sectors also differ in terms of how they incentivize individual vessels to avoid bycatch beyond simply following specified rules, and there are differences in what rules are imposed. The shore-based sector risk pool has looser control over individual vessels and consequently more issues with moral hazard and adverse selection than the mothership sector. Perhaps for this reason, it relies on premiums and co-payments and deductibles (all paid in-kind with quota), and also provides more limited coverage of bycatch risk. The mothership sector risk pool includes all vessels in the sector so does not face an adverse selection problem, and it exercises substantial control of vessels at the fleet level and can effectively impose bycatch avoidance behavior which essentially eliminates moral hazard. Consequently they do not rely on incentives at the individual vessel level, but do impose quarterly bycatch quotas that ensure vessels and motherships fishing in each quarter do not impinge on opportunities for those fishing later in the year.

The lack of discernable change in bycatch rates following implementation of catch shares and risk pools is likely due to the erratic and uncertain nature of bycatch and the trade-offs faced by vessels between different bycatch species (e.g., avoiding one increases risk of catching another). However it is clear that the fleets have made substantial efforts to develop institutions, technology and rules to avoid bycatch. Bycatch might have been substantially higher without this cooperation, but it is not possible to discern whether or how much it might have differed. Notably, the biomass of several of the rockfish stocks has increased substantially over the last few decades as these stocks have been rebuilt. Thus even constant bycatch rates would reflect increased avoidance.

Examining margins of adjustment, there is clear evidence of reduced night-fishing after cooperative institutions for the mothership whiting fleet. There is also evidence of increased bycatch avoidance for the at-sea fleets along other margins: move-on behavior after a bycatch event and test-towing in new areas or after a bycatch event. For the shore-based fleet, there is evidence that the fleet stopped test-towing in new areas but reduced tow duration after a bycatch event with the implementation of its bycatch risk pool.

Although risk pools have imposed what might be considered traditional command-and-control regulations on themselves (e.g., time and areas closures), many of approaches implemented by the risk pools probably could not have been implemented by regulators either because of the slowness of the rule making process (e.g., in-season closures) or because of regulators could not require vessels to share information. Risk pools are able to cooperatively decide on a carefully designed set of year-round closed areas that limit bycatch risk without closing down key harvest areas. They can also impose short-term area closures in bycatch hotspots quickly relying on shared information and without a formal rulemaking process.

A key characteristic of the bycatch problem in this fishery is the rarity, uncertainty and lumpiness of bycatch which makes it difficult for individual vessels to meet strict performance standards like maximum bycatch rates or individual quotas. In recognition of this, none of the sectors rely solely on individual incentives such as individual bycatch quotas, and two of the sectors pool bycatch quota. While the catcher-processor sector imposes vessel-level bycatch allowances, these vessel are extremely mobile and have greater ability to control bycatch over the course of a season. They can also trade bycatch allowances within the cooperative, and the cooperative has a long history of working cooperatively to avoid bycatch.

DATA AVAILABILITY STATEMENT

Restrictions apply to datasets. Analysis relies on observer data from fisheries which is confidential and requires negotiating a data access agreement.

AUTHOR CONTRIBUTIONS

Both authors participated in data analysis and writing of the manuscript.

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A Mitigation Hierarchy Approach for Managing Sea Turtle Captures in Small-Scale Fisheries

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The mitigation hierarchy has been proposed as an overarching framework for managing fisheries and reducing marine megafauna bycatch, but requires empirical application to show its practical utility. Focusing on a small-scale fishing community in Peru as a case study system, we test how the mitigation hierarchy can support efforts to reduce captures of sea turtles in gillnets and link these actions to broader goals for biodiversity. We evaluate three management scenarios by drawing on ecological risk assessment (ERA) and qualitative management strategy evaluation to assess trade-offs between biological, economic, and social considerations. The turtle species of management focus include leatherback turtle *Dermochelys coriacea*, green turtle *Chelonia mydas*, and olive ridley turtle *Lepidochelys olivacea*. Adopting a mixed-methods iterative approach to data collection, we undertook a literature review to collate secondary data on the fishery and the species of turtles captured. We then collected primary data to fill the knowledge gaps identified, including establishing the spatial extent of the fishery and calculating turtle capture rates for the fishery. We identified and evaluated the potential risk that the fishery poses to each turtle species within Pacific East regional management units using a qualitative ERA. Finally, we evaluated potential management strategies to reduce turtle captures, incorporating stakeholder preference from questionnaire-based surveys and considering preliminary estimates of trends across a range of performance indicators. We illustrate how the proposed framework can integrate existing knowledge on an issue of marine megafauna captures, and incorporate established decision-making processes to help identify data gaps. This supports a holistic assessment of management strategies toward biodiversity goals standardized across fisheries and scales.

Keywords: bycatch, fisheries management, management strategy evaluation, risk assessment, social-ecological system, structured decision-making, turtles

INTRODUCTION

Fisheries often seek to achieve "triple-bottom-line" outcomes that entail trade-offs between economic returns, social welfare, and biodiversity conservation (Halpern et al., 2013; Costello et al., 2016). Managing the recovery of depleted populations of marine megafauna species, which are defined as large-bodied, ocean dwellers like sea turtles, seabirds, marine mammals, and sharks, often sits in the middle of this nexus and persists as one of the major challenges in achieving ecologically and socioeconomically sustainable fisheries (Hall et al., 2000; Gray and Kennelly, 2018; Lewison et al., 2018). The complex and dynamic nature of attempting to target catch while minimizing the impact on non-target species means that fisheries management requires integrative processes to identify and mitigate the negative ecological impacts of fisheries while examining economic and social considerations on a fishery-by-fishery basis.

A variety of risk-based decision-making processes to assess the ecological impacts of fishing have been developed—also commonly known as ecological risk assessment (ERA; Lackey, 1994; Hobday et al., 2011). Management strategy evaluation (MSE) is a complementary simulation-based process for assessing trade-offs in potential management strategy performance (Smith, 1993, 1994; Fulton et al., 2014). While these and other structured decision-making processes are vital for fisheries management. There remains a need to further integrate fishery-specific management into national and international goals for biodiversity conservation. For example, those specified by multilateral agreements like the Convention on Biological Diversity (CBD). Since the 1992 adoption of the CBD (United Nations, 1992), the Food and Agriculture Organization (FAO) and regional fisheries management organizations (RFMOs) have made substantial progress in mainstreaming biodiversity conservation into fisheries management processes through frameworks, policies, and practices aimed at promoting more sustainable fishing practices (Friedman et al., 2018). But it is necessary to further support integrated partnerships between fisheries and the wider environmental sector, particularly in low- and middle-income countries, to ensure beneficial biodiversity conservation outcomes across fisheries at scale (Karr et al., 2017).

The mitigation hierarchy is a conceptual framework that can support integrating fisheries management with biodiversity conservation objectives (e.g., a scalable framework for linking actions to reduce sources of anthropogenic mortality over a species life cycle, migratory range, and habitat). In terrestrial and coastal ecosystems, the mitigation hierarchy is widely used as part of the decision-making process of environmental impact assessment (EIA; Council on Environmental Quality [CEQ], 2000) to identify and manage the negative impacts of human economic activities on biodiversity—most commonly applied to infrastructure development projects (e.g., roads, mining sites, wind farms; Bennett et al., 2017; Shumway et al., 2018). If implemented effectively, the framework can help to guide actions toward mitigating the negative impact on biodiversity following a traditionally damaging or extractive activity (Zu Ermgassen et al., 2019). Following widespread application in terrestrial and coastal development projects (Maron et al., 2016; Shumway et al., 2018), the mitigation hierarchy was proposed as an overarching framework for mitigating marine megafauna bycatch in fisheries (Milner-Gulland et al., 2018; Squires et al., 2018), and more broadly, for all human impacts on biodiversity (Arlidge et al., 2018).

A key benefit of the mitigation hierarchy is that it begins by setting a desired end-goal that can support the summation of multiple positive and negative impacts into a net, scalable, outcome (Bull et al., 2019). This goal is conventionally a no net loss or a net gain of biodiversity (Rainey et al., 2015). In a fishery setting, goals such as population recovery when managing protected species, or Maximum Sustainable Yield Biomass (B*msy*) when managing stocks of target catch, are equally feasible (Wolf et al., 2015; Squires and Garcia, 2018). The chosen goal is then measured using a quantitative target and metric(s) with reference to a baseline of biodiversity. Following goal-setting, the framework follows a step-wise decision-making process to identify a suite of measures for mitigating the negative impacts of human activity on biodiversity to achieve the specified goal. The mitigation hierarchy progresses in four sequential stages. The first three—avoid, minimize, and remediate—take place at the impact site (i.e., at sea where fishing is taking place). Then if any residual negative impacts remain, off-site compensatory measures such as biodiversity offsetting can be implemented (Council on Environmental Quality [CEQ], 2000; Bonneuil, 2015). All actions may not be applicable in all management scenarios (e.g., in-kind offsetting actions are not feasible for deep-sea trawl impacts on seamounts; Niner et al., 2018). Rather, the broad steps of the mitigation hierarchy act as a guide, with enough flexibility to achieve the integration of diverse fisheries management approaches toward a unified biodiversity goal that translates across scales (Milner-Gulland et al., 2018; Squires et al., 2018). Yet despite its theoretical attractiveness, there remains a need to empirically evaluate how the mitigation hierarchy can support fisheries management and bycatch mitigation in practice.

Peru's small-scale fisheries total more than 16,000 fishing vessels, with an estimated 44,161 fishers and 12,398 ship owners (Guevara-Carrasco and Bertrand, 2017). Of these vessels, approximately 4800 fish primarily with gillnets (Estrella and Swartzman, 2010). In Peru, the capture of sea turtles in coastal gillnets is a major conservation issue in the nation's northern fishing ports and landing sites (Alfaro-Shigueto et al., 2011, 2018). Gillnet fishing also plays an important role in food security, local employment, and social identity throughout Peru's coastal communities (Christensen et al., 2014). We explore the applicability of the mitigation hierarchy as an overarching framework for managing the population recovery of depleted sea turtle populations, by integrating multiple sources of data, highlighting uncertainties, and supporting management decisions that consider biological, social, and economic conditions in the coastal gillnet fishery. Throughout our investigation, we focus our attention on integrating the established decision-making processes of ERA and MSE with the mitigation hierarchy.

We draw on qualitative ERA (consequence × likelihood) theory to consider risks and associated impacts on multiple turtle species captured in our case study fishery (Fletcher, 2014). We then consider the integration of a qualitative MSE assessment with the mitigation hierarchy to measure the performance of management options aiming to reduce turtle captures and consider how they trade-off against economic and social considerations (Smith et al., 2004; Dichmont and Brown, 2010). The management objectives sought through both ERA, and MSE assessments, are typically fishery or management-region specific (Fulton et al., 2011b; Fletcher, 2014). Objectives typically focus on achieving economic efficiency and ensuring that exploitation is consistent with the principles of ecologically sustainable development, and the exercise of the precautionary principle. The mitigation hierarchy's goals, by contrast, are often specifically chosen to be translatable between global, management-region, national, and fishery scales. Goals and targets at the fishery level may vary depending on the biodiversity component assessed, as well as data availability and capacity while combining to achieve the overarching goal at higher levels (Milner-Gulland et al., 2018). The mitigation hierarchy framing, therefore, constitutes a shift in approach to objective-setting at an individual fishery level toward the summation of positive and negative impacts into a net, scalable, outcome (Bull et al., 2019). Economic and social management benchmarks could also be set using a mitigation hierarchy framework. An economically focused goal may seek to maximize net economic returns to the community within a fisheries management region, measured by summing fishery-related profits and losses against a predetermined baseline. A social goal may seek to ensure that the community within the fisheries management region is no worse off, or preferably better off, in terms of their well-being as a result of fishery management (Griffiths et al., 2018).

In this study, we explore the potential of the mitigation hierarchy for integrating fisheries management and biodiversity conservation processes, with a focus on achieving population recovery goals for captured sea turtles: a taxon primarily threatened by negative fisheries impacts (Wallace et al., 2010b). The mitigation hierarchy builds on ERA and MSE in two ways: First, it requires a clear definition of management benchmarks (a biodiversity goal and associated targets measured against a baseline of biodiversity) that are generalizable across fisheries and scales (Milner-Gulland et al., 2018). Second, its consideration of a broad suite of purely technical conservation actions and market-based mechanisms for environmental conservation (avoidance, minimization, remediation, offsetting)—rather than the common focus on at-sea minimization and remediation—encourages a more holistic recovery strategy for marine megafauna species that integrates measures to reduce anthropogenic mortality over a species life cycle, migratory range, and habitat into the management process (Dutton and Squires, 2008; Squires et al., 2018). The mitigation hierarchy can also help to identify key uncertainties and knowledge gaps, as well as difficult trade-offs between biodiversity conservation goals, different management strategies, and other socio-economic objectives of fishers and fisheries.

We implemented the mitigation hierarchy in an iterative process; collating existing data to characterize the fishery and the turtle capture problem, identified areas of uncertainty, and gathered primary data to address key knowledge gaps. We then integrated all existing and gathered data under the mitigation hierarchy framework to assess risk, using methods taken from ERA. We explored how potential management measures can be assessed by drawing on qualitative MSE methods. Finally, we discuss the potential for, and limitations of, the proposed mitigation hierarchy framework, with a focus on the need to better integrate diverse fisheries management approaches, impacts, and mitigation actions across scales.

MATERIALS AND METHODS

Study Site

San Jose, Lambayeque, Peru (6°46' S, 79°58' W) is a key site for coastal gillnet fishing (Guevara-Carrasco and Bertrand, 2017). Longline, purse seine, trawl, squid jigging, handline fishers, and divers also operate from the community. Among the diverse range of fishing gears, gillnets are the most prevalent. Two distinct gillnet fleets operate from San Jose. First, the "San Jose inshore gillnet fleet" (IG) comprises a class of open-welled vessels with a capacity range from 1–8 t. Second, the "San Jose inshore-midwater gillnet fleet" (IMG) comprises a larger vessel class with small closed bridges ranging in capacity from 5–32 t. We refer to the "San Jose gillnet fishery" when referencing both fleets together. Fishers operating in the San Jose gillnet fishery use both surface driftnet and fixed demersal nets configurations, with some fishers switching between the two on a single trip (Alfaro-Shigueto et al., 2010).

We focus on a single marine megafauna taxon for our assessment of bycatch impacts—sea turtles (superfamily Chelonioidea). Three turtle species are regularly captured in the San Jose gillnet fishery (Alfaro-Shigueto et al., 2007, 2018). The global populations of all seven extant sea turtle species are Threatened under the International Union for Conservation (IUCN)—World Conservation Union's Red List of Threatened Species (International Union for Conservation of Nature [IUCN], 2010a)—the critically endangered leatherback turtle *Dermochelys coriacea*, the endangered green turtle *Chelonia mydas*, and the vulnerable Olive ridley turtle *Lepidochelys olivacea*.

Peru has a history of sea turtle consumption (Aranda and Chandler, 1989), and turtles are still eaten despite protection under Peruvian law since 1995 (Morales and Vargas, 1996; Alfaro-Shigueto et al., 2018). Thus, we make a distinction between the capture of turtles in fishing gear, the targeted and non-target use of captured turtles, and bycatch (the capture and discard at sea, dead or injured to an extent where death is the result), following the definitions used in Hall (1996).

In Peru, the Peruvian Marine Research Institute IMARPE (Instituto del Mar del Peru) conducts government-managed marine research. The Peruvian Coastguard DICAPI (Dirección de Capitanías y Puertos) undertakes enforcement in most cases. Despite an IMARPE and DICAPI presence in San Jose, Peru's current regulatory structure does little to help mitigate the capture of marine megafauna species like sea turtles. San Jose's gillnet fishery operates as an open-access fishery—with no catch restriction [e.g., a total allowable catch (TAC)] in place for target species (Bjørndal and Conrad, 1987). In 2010, the Peruvian government implemented an effort restriction with a ban on new boats above 5 m^3 gross registered tonnage (GRT) entering the nation's small-scale fisheries (Supreme decree N° 018-2010-PE), but limited enforcement of this rule means that vessel builders still operate actively (Estrella, 2007; Christensen et al., 2014).

The fishery is predominantly beyond the reach of RFMOs, and trade measures are limited by the coastal desert environment. In San Jose, many coastal gillnetters work alternate jobs, often throughout the winter when fishing effort and catches are low because of rough weather conditions preventing fishing. Few options exist for alternate revenue streams for fishers in San Jose (e.g., "mototaxi" driver, construction worker, general store clerk). Alternate livelihoods such as these are often tied to the success of local fishing (i.e., more people will use local transport or spend money at local shops when their revenue is high from a good fishing period). With limited regulatory efficacy, not-for-profit organizations play a key role in filling data gaps and implementing conservation interventions in this data-limited, open-access fishery.

Applying the Mitigation Hierarchy

We use the mitigation hierarchy as a conceptual model and framework for structuring data and generating management recommendations toward a standardized biodiversity conservation goal. Milner-Gulland et al. (2018) present two main steps to make the mitigation hierarchy relevant to fisheries management and mitigating marine megafauna bycatch. These are (i) defining the problem (by characterizing the fishery and bycatch issue, and setting the goal, target, metric, and baseline), and (ii) exploring potential management options by systematically stepping through the mitigation hierarchy using a conceptual framing. Booth et al. (2019) explore the potential for the application of the mitigation hierarchy to shark bycatch management. These authors subdivide the two steps in Milner-Gulland et al. (2018) into five. These include (i) defining the problem, (ii) exploring potential management measures using the mitigation hierarchy, (iii) assessing the hypothetical effectiveness of management, (iv) making an overall management recommendation or decision, and (v) implementing, monitoring, and adapting implemented management measures. Here we use the steps proposed in Booth et al. (2019), in combination with data from a real-world fishery, to explore the advantages and disadvantages of the mitigation hierarchy framework. We further develop the framework by exploring its potential for integration with MSE to evaluate trade-offs between management scenarios.

Data Collection and Analysis

We adopted a mixed-methods iterative approach to data collection and analysis, drawing on primary and secondary data sources and multiple analytical methods to understand the fishery problem and explore potential management measures.

Understanding the Problem: The Fishery and Species of Concern

We collated all available information on the San Jose gillnet fishery and each of the turtle species of management concern from published and unpublished sources using a literature review and available datasets. We then collected primary data through field-based surveys to fill several key knowledge gaps.

Secondary Data

We sourced secondary data on turtle capture and bycatch in the IMG fleet from a voluntary at-sea human observer program managed by a local not-for-profit, *ProDelphinus*. This program has been operating with skippers and crew of IMG vessels along Peru's coastline since 2007. Observer surveys have been undertaken in the IMG fleet since the program's inception, but there are no site-specific turtle capture or bycatch per unit effort rates calculated. No observer data exist for the IG fleet, but we can gain insight into the turtle species captured in this fleet from existing data collected through harbor-based surveys of fishers and local government representatives (e.g., Alfaro-Shigueto et al., 2007, 2018).

We also collated relevant information on leatherback, green, and olive ridley turtles with consideration for management in our case study fishing system. We identified potential turtle capture and bycatch reduction strategies based on a literature search, which was later refined using stakeholder consultation (see section "Primary Data").

Primary Data

To better understand the fishery impacts on sea turtles, we collected primary data to quantify the fishing seasons and geographic extents of the two gillnet fleets. To quantify local fishing seasons, we conducted key informant interviews with a local IMARPE scientist and the presidents of the two at-sea fishing groups in San Jose. To estimate the geographic extent of the gillnet fleets, we used a combination of key informant interviews and focus group discussions (FGDs). We held two FGDs, one for each gillnet fleet. The FGD estimating the IG fleet's geographic extent had 15 participants, comprising 13 skippers of inshore gillnetting vessels, an IMARPE scientist, and a not-for-profit employee (JAS). The FGD estimating the IMG fleet's geographic extent had five participants, comprising three gillnet skippers and two not-for-profit employees (JAS & JCM). We used simple random sampling by number generator to select gillnet skippers from lists of 150 actively fishing IG skippers, and 18 actively fishing IMG skippers. We assigned skippers fishing within each fleet to the relevant FGD. For supplementary analysis, we present a summary of demographic data (see **Supplementary Material**).

We asked respondents to estimate the maximum geographic range that fishing vessels from their fleet traveled from San Jose (north, south, west). Respondents' maximum geographic extent was then averaged across each group's participants and displayed using ArcMap (Environmental Systems Research Institute [ESRI], 2018). We gave the respondents the option to input additional information or adjust their estimates. No respondents adjusted their estimates in this final round. We collected all primary data during field surveys in San Jose from 1 July–30 September 2017. This research has Research Ethics Approval (CUREC 1A; Ref No: R52516/RE001 and R52516/RE002).

Assessing Fishery Risks

To better quantify fishery risks, we first analyzed available onboard observer records from the IMG fleet from August 2007 to March 2019. We calculated turtle captures per trip for the IMG fleet and consider the portion of mortalities and captures returned to sea injured or unharmed. We used an analysis of variance and a *post hoc* Tukey test to compare capture rates between species groupings. All analyses were completed using core packages in R (R Core Team, 2019).

To evaluate the risks for sea turtle populations captured in the San Jose gillnet fishery, we use the consequence–likelihood (probability) matrix methodology that originated from Australian and New Zealand Standard Risk Analysis (Standards Australia, 2000, 2004) for fisheries management (Fletcher et al., 2003; Fletcher, 2005). The methodology is widely implemented (e.g., Fletcher, 2008; Food and Agriculture Organization [FAO], 2012). Iterative updates to the ERA method have followed to ensure compliance with the revised international standards for risk management (International Organization for Standardization [ISO], 2009), and to enable consideration of ecological, economic, social, and governance risks (Fletcher, 2014).

We focused on direct risks posed to turtles captured in our case study fishery (addressing both the IG and IMG fleets) relative to each species distribution and estimated population sizes throughout their respective Pacific East regional management unit (RMU), as developed by Wallace et al. (2010a). RMUs delineate global turtle populations according to regional areas that are distinct from one another based on genetics, distribution, movement, and demography, and provide a practical management unit for assessment analogous to the IUCN—World Conservation Union's Red List of Threatened Species subpopulation categorizations, but for all extant marine turtle species (Wallace et al., 2010a). RMUs allowed for an evaluation of the relative risk posed from the two San Jose gillnet fleets to each turtle species' population that is directly affected by fishing activity within our case study system. The analysis assessed how the biology and distribution of each species within the Pacific East RMUs affected susceptibility to the risk from each gillnet fleet, and whether the current management arrangements in place in our case study fishery (i.e., fishing regulations and compliance therewith) were working effectively or not. Consideration was also given to the wider fishing impacts on each species throughout their respective Pacific East RMU distributions (see **Supplementary Material**). When implementing an ERA in full, a complete evaluation of all risks posed to all target catches, non-target catches, habitat, and social and governance structures across the focus fishery is necessary (Fletcher, 2014).

Critically, risk analysis evaluates the level of risk that a given impact (e.g., incidental capture in gillnets) poses to achieving the goals and targets set over a specified assessment period with the current management measures in place (Fletcher, 2014). We evaluated the risk posed from the IMG and IG fleets against achieving the high-level biodiversity goal of population recovery of leatherback, green, and olive ridley turtle populations (Pacific East RMUs) in the shortest time possible (in line with international biodiversity targets). The mitigation hierarchy framework specifies that goals must be operationalized through quantitative targets, for which metrics and baselines can be defined (Milner-Gulland et al., 2018). The San Jose gillnet fishery does not have management benchmarks in place to meet high-level goals of turtle population recovery. Thus, we propose a fishery-specific target of reducing turtle captures from 2020 levels by 15% every year for 5 years while maintaining total catch weight. As more data become available and population models develop, we recommend a net change in population growth rate target measured against an agreed baseline (Milner-Gulland et al., 2018).

We ranked the risk from each gillnet fleet in terms of a consequence (C) level (specifying a level of impact) the fishing fleet in question is likely to have for each turtle species assessed, using a four-point scale from minor [1] to extreme [4], and the likelihood (L) that a specific consequence level will occur, also using a four-point scale from remote [1] to likely [4] (**Table 1**). Sources of risk (i.e., the two San Jose gillnet fleets) were then assigned a score for each turtle species, calculated by multiplying the consequence and likelihood values (e.g., consequence level of impact x on turtle species $y \times$ the likelihood of consequence x occurring to turtle species y). The risk posed from each gillnet fleet for each turtle species were then assigned one of four levels of impact ranging from minor to extreme (**Table 2**). If more than one combination of consequence and likelihood was plausible, we chose the combination that generated the highest risk score (i.e., consistent with taking a precautionary approach; Fletcher, 2014).

TABLE 1 | Consequence (level of impact) and likelihood (a subjective probability) descriptors used to evaluate identified risks (following Fletcher, 2014).

Level	Descriptor
Consequence for protected species	
Major (C4)	Further declines generated and major ongoing public concerns
Severe (C3)	Recovery may be affected and/or some clear public concern
Moderate (C2)	Catch or impact at the maximum level that is accepted by public
Minor (C1)	Few individuals directly impacted in most years, no general level of public concern
Likelihood of a specific consequence occurring to protected species	
Likely (L4)	A particular consequence level is expected to occur in the time frame (indicative probability of 40–100%)
Possible (L3)	Evidence to suggest this consequence level may occur in some circumstances within the time frame (indicative probability of 10–39%)
Unlikely (L2)	The consequence is not expected to occur in the time frame but some evidence that it could occur under special circumstances (indicative probability of 3–9%)
Remote (L1)	The consequence not heard of in these circumstances, but still plausible within the time frame (indicative probability 1–2%)

TABLE 2 | Consequence (C) × likelihood (L) risk matrix (following Fletcher, 2014).

		Likelihood level			
		Remote	Unlikely	Possible	Likely
Consequence level		1	2	3	4
Minor	1	1	2	3	4
Moderate	2	2	4	6	8
Major	3	3	6	9	12
Extreme	4	4	8	12	16

*The descriptions of each of the consequence and likelihood levels are presented in **Table 1**. The numbers in the cells indicate the risk score values and the colors/shades represent the levels of risk as described in **Table 1**. The level of impact is determined by summing C × L. Impact levels include: minor (1–2), moderate (3–4), major (6–8), and extreme (9–16).*

Exploring Management Options

Based on the quantified risks, we then used the conceptual framework for bycatch mitigation presented in Milner-Gulland et al. (2018) to consider how additional management strategies could be implemented to reduce the risk of fishing-related mortality for leatherback, green, and olive ridley turtles (Eq. 1):

$$\Delta\lambda_T = f(E_B \times \text{BPUE}) - O_T \tag{1}$$

In Milner-Gulland et al. (2018), the equation relates to a particular bycatch species, in which the unit ($\Delta\lambda_T$) is the rate of change in population size as a result of bycatch and its mitigation. f(E_B × BPUE) is the effect on the population growth rate of the bycatch-relevant component of fishing effort, broken down into the bycatch-relevant effort, E_B, and the bycatch taken per unit of that effort, BPUE, where f() is the effect of this effort on a given species of sea turtle's population dynamics. A reduction in E_B is equivalent to a fishery avoiding bycatch of turtle population *x*, partially or completely. A reduction in BPUE is the result of the on-site measures encompassed in the "minimize" and "remediate" steps of the mitigation hierarchy. O_T is the net effect on the population growth rate of policies aiming to improve the overall viability of turtle population *x*, representing the "offsetting" of any remaining residual damage caused, using compensatory measures away from where the fishing impact occurs (e.g., nesting site protection). In this data-limited case study, the relationship between BPUE and each turtle's population growth rate [i.e., f()] is unknown. As such, we do not attempt to solve Eq. 1 for a population growth target. Rather, we use the equation as a conceptual model for evaluating how management strategies can help reduce different components of turtle bycatch risk, and for illustrating where a potential management strategy sits within the wider mitigation hierarchy. The flexibility of the model allows for components of the equation to be further deconstructed in to separate factors. For example, BPUE can represent the sum of individual turtle species *x* that are dead on arrival to the vessel, individuals captured and dying on the vessel, and individuals dying after live release, as follows:

$$\text{BPUE} = B_{DOA} + P_{DV} \times B_{OB} + (1 - P_{DV}) \times B_{OB} \times P_{DR} \tag{2}$$

where B_{DOA} is the bycatch per unit effort that arrives at the boat dead, B_{OB} is the bycatch per unit effort that arrives at the vessel alive, P_{DV} is the proportion dying on the vessel, and P_{DR} is the proportion dying after release. This decomposition can help with identifying different points for management interventions within the fishing process (Milner-Gulland et al., 2018).

To understand the feasibility of different management measures and support the selection of multi-strategy scenarios for the MSE assessment, we interviewed a subset of gillnet skippers operating in San Jose about their personal preferences for potential management options using questionnaires that gathered basic demographic information and incorporated a quantitative five-point Likert-scale assessing strong disagreement to strong agreement with each strategy proposed. We analyzed data in R version 3.6.1 (R Core Team, 2019).

Assessing the Hypothetical Effectiveness of Management Options

To assess trade-offs in potential management strategy performance, we draw on the conceptual integration of the mitigation hierarchy with MSE (Bull and Milner-Gulland, 2019) and demonstrate the implementation of the two processes in a data-limited management scenario. MSE generates simulations within an operational model such as the Atlantis model framework, adapted from the work of Fulton (2001). However, it is possible to qualitatively assess management strategy scenarios against performance indicators (e.g., area fished, catch, BPUE) derived through a process of expert judgment and stakeholder consultation (e.g., Smith et al., 2004; Dichmont and Brown, 2010).

Qualitative MSE assessments can be undertaken in data-limited management scenarios as a preliminary assessment with the intent to undertake a quantitative evaluation of management scenarios during the next iteration of the management project (Dichmont and Fulton, 2017). The evaluation phase implemented in the current study involved a project team (the authors) made up of several subject matter experts (with over 125 years of collective experience in conservation science and fisheries management research, and over 25 years of collective experience working in the case study fishery). The analysis was undertaken through an iterative web-based evaluation process, with participants drawing on their expert opinion and the collated and collected data presented in the current study (see **Supplementary Material** for further presentation of data used during the assessment).

Based on indicators applied in MSE analyses (Smith et al., 2004), we compiled a list of performance indicators (**Table 3**) to evaluate management strategy scenarios against the high-level biodiversity goal (i.e., population recovery of the Pacific East RMU population for each turtle species), and the proposed fishery-specific target (i.e., reducing turtle captures from 2020 levels by 15% every year for 5 years while maintaining total catch weight). It is assumed that managers would maintain capture rates at or below the 5 year target level going forward, or update the target at the end of this assessment period as data become available.

The project team evaluated three management scenarios that were subjectively selected based on our fieldwork results, the compiled data, and the output of the ERA. Once we had specified the management scenarios and performance indicators (**Table 3**), we evaluated the consequence of each scenario by predicting how each performance indicator would change over a 10 year assessment period given the project team's knowledge and assumptions about the system dynamics of the San Jose gillnet fishery system (Smith et al., 2004). We chose a longer assessment period for the qualitative MSE (10 years) over the ERA (5 years) to reflect a realistic timeframe for implementing potential management strategies. We present predicted trends in performance indicators. We highlight that the current qualitative MSE assessment remains preliminary. When implementing a full MSE (whether this be qualitative or quantitative), potential management scenarios should undergo a broad stakeholder consultation and engagement process during which time, stakeholders representing different sector's interests can input ideas and submit other management strategy combinations for evaluation (as undertaken in a qualitative MSE process for Australia's South-east Shark and Scale Fishery; Smith et al., 2004).

TABLE 3 | Proposed performance indicators for assessing management scenarios against set goals, targets, and baselines for bycaught turtle species in the San Jose gillnet fishery.

Indicator
Technical/biological
Threatened, endangered, protected species
BPUE
Leatherback turtle
Green turtle
Olive ridley turtle
Ecological sustainable development (ESD)
Impact from the San Jose gillnet fishery
on biodiversity composition
Fishing effort
Geographic extent
Set number × set time
Distance traveled
Discards
Habitat and sessile communities
Socio-economic
CPUE
Management costs
Stable management
Gear conflict
Revenue per ton of fish landed
Revenue per day fished
Cost per day fished
Return on investment
Food security
Employment security
Local fish processing
Local transport, boat building, and maintenance
Access to other services
Improvement in conservation values
Desire to participate in bycatch reduction initiatives in future
Social networks (leadership)
Formation of local institutions
Public perceptions of conservation
Trust and confidence in authorities

RESULTS

The Fishery and Turtle Bycatch Rates

The San Jose gillnet fishery comprises two distinct gillnet fleets that fluctuate in vessel number and effort between the fishing seasons of summer and winter. The main uncertainties identified related to the geographic extent of the two gillnet fleets, fishing seasons length, and seasonal and annual fluctuations in fleet size (**Table 4**).

Respondents in the key informant interviews identified two distinct fishing seasons, with fishing effort varying between winter and summer conditions. In the northern regions of Peru, summer is usually December–February (3 months), but the government fisheries scientist noted that summer-like fishing conditions span December–May, with this longer seasonal division supported by the presidents of the two local at-sea fishing groups, and by capture reports from the Lambayeque region (Guevara-Carrasco and Bertrand, 2017). FGDs estimated the maximum geographic extent for two fleets comprising the San Jose gillnet fishery across the defined seasonal breaks (**Table 4**). We then overlaid observed sea turtle captures and fishing effort data from the IMG fleet to corroborate respondents' estimates of this fleet's geographic extent (**Figure 1**).

Two shore-based surveys recorded turtle captures in San Jose (Alfaro-Shigueto et al., 2007, 2018; **Table 5**). From July 2000 to November 2003, a combination of shore-based- and at-sea observer surveys recorded nine leatherback turtle captures in San Jose (across gillnet and longline gear types; Alfaro-Shigueto et al., 2007). Turtle capture and bycatch rates were available for the towns of Constante, Salaverry, and Ilo (Alfaro-Shigueto et al., 2011). In Salaverry and Constante, most turtle captures in gillnets were green turtles (Alfaro-Shigueto et al., 2011; **Figure 1**). Turtle bycatch reports from Salaverry found leatherback turtles captured close to the coast, indicating a potential coastal foraging "hot spot"; if captured, consumption rates were high (Alfaro-Shigueto et al., 2007, 2018).

There were 461 fishing trips observed from San Jose, of which observers recorded the capture of 379 turtles in gillnets. Observer coverage for the IMG fleet is low at ~1–4% fleet coverage spanning 11 years and 7 months. Species proportions were 86.8% green (n = 329), 9.2% olive ridley (n = 35), 1.8% leatherback (n = 7), and 2.1% unidentified hardshell turtle species (n = 8; **Figure 1**). Turtles released alive without visible injury made up 62% of the 379 captures. Live releases with injuries made up 28%. Mortalities 8% of captures (see **Supplementary Material**). Capture per unit effort across trips (n = 461) was significantly different between species [one-way analysis of variance; $F(2,1380) = 49.73$, $p < 0.001$]. Green and olive ridley turtle capture rates per trip differed significantly ($p < 0.05$; Tukey

TABLE 4 | Characteristics of the San Jose gillnet fishery, Lambayeque, Peru (6°46′ S, 79°58′ W).

Fishery element	Lines of evidence	Uncertainties/filled data gaps
Vessel type	Peruvian law defines SSF vessels as displacing a maximum of 32.6 m^3 gross registered tonnage (GRT), up to 15 m length, and operated predominantly manually (Legislative decree N° 012-2001-PE). San Jose vessels using gillnets can be divided into two distinct fleets: (i) the "inshore gillnet fleet" (IG) comprises vessels of 1-8 GRT, locally known as "chalana," and (ii) the "inshore/midwater gillnet fleet" (IMG) comprising vessels of 5–32 GRT, locally known as "lancha" (Guevara-Carrasco and Bertrand, 2017).	Rate of gear switching to gillnets from vessels that primary fish with another gear type.
Fleet size	IG fleet is increasing in size. The IMG fleet is thought to be decreasing in size as many fishers' switch from gillnets to squid jigging. Estimates of gillnet activity in San Jose recorded 47 gillnet vessels fishing in November 1995–April 1996 (Escudero, 1997) and 95 gillnet vessels fishing in January–April 2004 (Alfaro-Shigueto et al., 2010). Skippers typically operate with 1–4 crew (Alfaro-Shigueto et al., 2010).	**In the winter of 2017, the IG fleet was estimated at 150 actively fishing vessels, and the IMG fleet 18 actively fishing vessels.** Not always known when vessels are active and inactive.
Fishery geographic extent	Two distinct fleets with different fishing footprints. Limited GPS coordinates for observed trips from the IMG fleet (5–32 GRT). Landing site/port surveys (Alfaro-Shigueto et al., 2007, 2011, 2018) and bycatch location reported from the HF two-way radio outreach program (Alfaro-Shigueto et al., 2012).	**Focus group discussion mean estimates for the maximum geographic extent for the San Jose IG fleet was 1200 km^2 in summer and 3700 km^2 in winter, and the IMG fleet 27,000 km^2 in summer and 31,500 km^2 in winter**.
Target catch	Surface drift net: target sharks, rays mahi mahi, bonito, swordfish *Xiphias gladius*, flathead gray mullet *Mugil cephalus*, Peruvian silverside *Odontesthes regia*; Bottom set net: target sharks, rays flounder, lobster (Alfaro-Shigueto et al., 2010; Guevara-Carrasco and Bertrand, 2017).	Target catch behavior in relation to turtle bycatch reduction technologies (e.g., gillnet illumination). Impact that shifting target species would have on turtle bycatch.
Fishing seasons	Two main seasons in San Jose. Summer usually spans 3-months December–February and winter March–November. Lambayeque catch reports indicate summer-like fishing conditions span a longer period (Guevara-Carrasco and Bertrand, 2017).	**San Jose winter fishing season is June–November and the summer fishing season as December–May.**
Market type	The nearest fish market is in Santa Rosa located 21 km from San Jose (**Figure 1**). Catch is sold locally and domestically. Refrigeration trucks present daily.	Lack of oversight as to where all the catch taken using San Jose gillnets, e.g., local in San Jose, Chiclayo (largest nearby city), wider Lambayeque region, other regions, international markets.

*Here we define the bycatch problem by first collating lines of evidence on fishery type, fleet size and spatial extent, target catch, fishing seasons, and relevant markets, and then evaluating known uncertainties. Text in **bold** highlights collected data filling identified knowledge gaps.*

post hoc tests), but leatherback and olive ridley turtle capture rates were not significantly different at the trip level (**Table 6**).

Risk Assessment

Inshore-Midwater Gillnet Fleet

We ranked the leatherback and green turtle RMU (Pacific East) populations as subject to an extreme risk from the San Jose IMG fleet over the next five years, and the olive ridley turtle RMU (Pacific East) as subject to a major risk given the current management measures in place (**Table 7**). No catch restrictions or effort limits exist, but five of the estimated 18–28 vessels comprising the IMG fleet (**Table 4**) were using light-emitting diodes (LEDs) on their nets—a form of at-sea minimization. This equates to illuminated nets on 27% of IMG vessels in winter and 18% of vessels in summer. In Sechura Bay, located approximately 150 km north of San Jose (**Figure 1**), controlled gear trials were implemented testing the turtle mitigating potential of LEDs on gillnets. The study found LEDs reduced green turtle bycatch by 64.7% with no reduction in target catch (Ortiz et al., 2016). While no fishery- or region-specific data on the effect that LEDs have on leatherback and olive ridley captures exist, anatomical, physiological, and behavioral studies show leatherback turtles also have a sensitivity to ultraviolet (UV) wavelengths (Wang et al., 2013; Wyneken et al., 2013). Over the assessment period, we assumed a 64.7% reduction in captures for each of the turtle species captured, across 27% of the IMG fleet in winter and 18% in summer. Workshops training fishers on safe handling and release of captured turtles are conducted in San Jose by the not-for-profit *ProDelphinus*. We estimated a small increase in post-capture survival rates of turtles based on known sea turtle survival rates following capture in gillnets (Epperly et al., 2004; Snoddy and Williard, 2010).

The IMG fleet concentrates fishing effort nearshore between Lobos de Tierra in the north, Salaverry in the South, and west to Lobos de Afuera (**Figure 1**)–this shows fishing effort occurs in areas where each turtle species has a known presence. The fleet covers less than 5% of each turtle species' Pacific East RMU distribution. Using the lines of evidence (**Tables 4**, **5**), all four levels of consequence (i.e., some level of turtle bycatch) were plausible for each turtle species, but with different levels of likelihoods (**Table 7**).

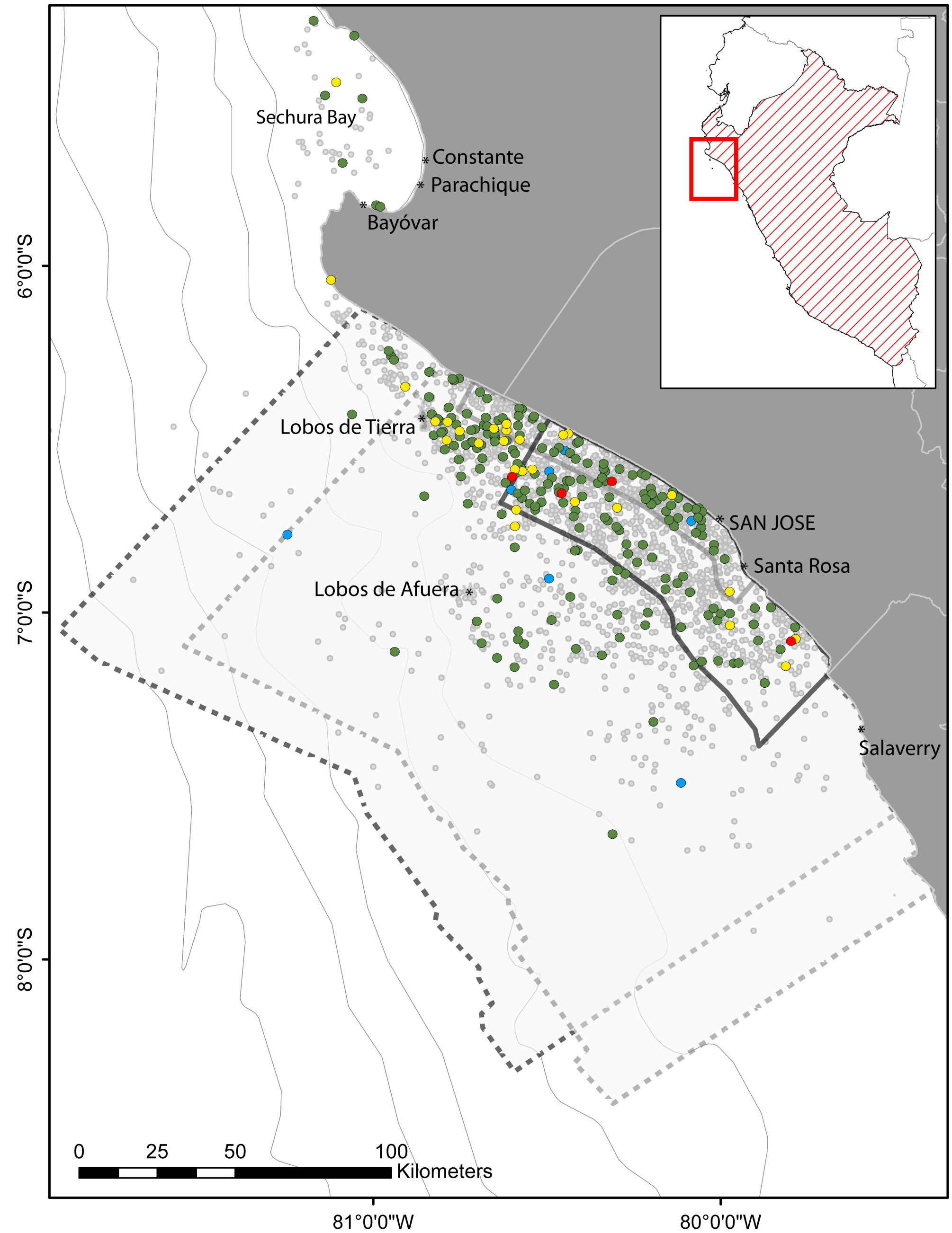

FIGURE 1 | Estimated summer (December–March) and winter (June–November) geographic extent for the San Jose inshore-midwater gillnet (IMG) fleet, and the San Jose inshore gillnet (IG) fleet (dark gray dashed line = IMG winter, light gray dashed line = IMG summer, dark gray line = IG winter, light gray line = IG summer). The fleets' geographic extents are overlaid by a distribution of sea turtle bycatch by species (green circles = green turtles, blue circles = leatherback turtles, yellow circles = olive ridley turtles, red circles = unidentified hardshell turtle species) relative to observed fishing effort for the San Jose IMG fleet from August 2007 to March 2019 (small gray circles). No turtle bycatch data from the inshore gillnet fleet have been recorded. Fishing areas were elicited from San Jose gillnet skippers during focus group discussions. Distances are the maximum (group mean) distance skippers estimated any skipper fishes from San Jose (north, south, west). Captures and fishing effort north of Bayóvar show trips that either left or landed from San Jose but began or concluded at the Bayóvar port.

The olive ridley turtle is the most abundant sea turtle in the world (Wyneken et al., 2013). The Pacific East RMU population numbers approximately 1,500,000 individuals (Eguchi et al., 2007; Wallace et al., 2010a). The population has an increasing trend in the short term but a predicted decreasing trend in the long term (Wallace et al., 2010a). The olive ridley turtle species exhibit both solitary and "arribada" nesting; the latter is a behavior unique to the *Lepidochelys* genus where large groups of females nest synchronously at a nesting site (Richard and Hughes, 1972; Wyneken et al., 2013). The observer data

TABLE 5 | Summary table of the information used to complete the risk assessment for turtle species captured in the San Jose gillnet fishery.

Species of concern	Lines of evidence		
	Biological factors	**Susceptibility to the fishery**	**Socio-economic outcomes**
Leatherback turlte *Dermochelys coriacea*	*Size:* Up to 215 cm (7 feet) *Weight:* Up to 900 kg (2000 pounds) *The average lifespan in the wild:* 45 years *Sexual maturity:* ∼16 years *Fecundity:* One female may lay up to nine clutches in a breeding season. Average clutch size is approximately 110 eggs, with up to 85% of these in a viable state. *Habitat:* Primarily pelagic (open ocean) dwelling. Females require sloped sandy beaches for laying clutches of eggs. *Nesting sites:* The East Pacific population nests along the Pacific coast of the Americas from Mexico to Ecuador. No established nesting sites for leatherback turtles are present in Peru. The closest nesting area to San Jose is located in Ecuador (Eckert, 2012). *East Pacific RMU geographic extent:* From the tip of Baja California Mexico south to Chile, out to 135 W (Wallace et al., 2010a). *East Pacific RMU population size:* ca. > 200 (Wallace et al., 2010a, 2013). Preliminary data show a small percentage of leatherback turtles present in the waters of the Pacific East regional management unit (RMU) are from the Pacific West RMU (P. Dutton, pers. comm.). *Population trend (East Pacific RMU short and long-term/Global):* decreasing short- and long-term (Wallace et al., 2010a); global population decreasing.	*Catchability in the fishery:* **BPUE per trip in San Jose is 0.02 ± 0.21 (mean ± SD). Seven recorded captures (all released alive) in San Jose inshore/midwater fleet from observer data between 2007 and 2017 (Table 6)**. *Distributional overlap:* 100% of total area within boundaries of the fishery (**Figure 1**). High leatherback captures in coastal gillnet locations near Salaverry port, south of San Jose (Alfaro-Shigueto et al., 2007). *Management restrictions:* poor—few restrictions are in place to support a reduction in leatherback turtle bycatch. Five inshore/midwater gillnet vessels are using LED lights and remote electronic monitoring systems as part of a trial community cooperative with a local not-for-profit. *Overlap in the effective fishing effort:* Most of the fishing effort is concentrated in the first 25 km of ocean from the shore (**Figure 1**). Few sets have been recorded further offshore than Lobos de Afuera. *Management effectiveness and compliance:* Unknown	*Social use:* Retention for human consumption is known to occur. Of the133 leatherback turtles captures recorded in Peru's SSF 1985–2003, 58.6% were retained for consumption (Alfaro-Shigueto et al., 2007). *Value:* Unknown *Target market if sold:* If eaten, turtles are usually consumed onboard the vessel or at home after a fishing trip. Black markets provide a platform for the sale of the illegal product (Quiñones et al., 2017). *Cultural values:* Turtle meat was historically eaten in Peru (Aranda and Chandler, 1989).
Green turtle *Chelonia mydas*	*Size:* Up to 150 cm (5 feet) *Weight:* Up to 315 kg (700 pounds) *The average lifespan in the wild:* 80 + years *Sexual maturity:* ∼25 years *Fecundity:* Nesting occurs nocturnally at 2-, 3-, or 4-year intervals. Max nine clutches within a nesting season (average 3.3). *Habitat:* Shallow waters (except when migrating) inside reefs, bays, and inlets (Seminoff et al., 2015). *Nesting sites:* Nesting occurs in more than 80 countries. The southernmost nesting sites for the species have been reported in Los Pinos, Tumbes, northern Peru (Forsberg et al., 2012), approximately 466 km from San Jose. *East Pacific RMU geographic extent:* Los Angeles south, sweeping down the coast of Chile and the Eastern Tropical Pacific out to 145 West (Wallace et al., 2013). *East Pacific RMU population size:* 3750 (Wallace et al., 2010a). *Population trend (East Pacific RMU short and long-term/Global):* Increasing short-term (Wallace et al., 2010a; Seminoff et al., 2015), decreasing long-term; global population decreasing.	*Catchability in the fishery:* **BPUE per trip in San Jose is 0.71 ± 1.98 (mean ± SD). 329 captures in San Jose inshore/midwater fleet from observer data between 2007 and 2017 (Table 6).** *Distributional overlap:* 100% of the total area within the boundaries of the fishery (**Figure 1**). Reports of high capture rates in gillnets in northern fishing locations during key information interviews in San Jose. *Management restrictions:* poor—few restrictions are in place to support a reduction in green turtle bycatch. See leatherback section for further details. *Overlap in the effective fishing effort:* Majority of fishing effort is concentrated in the first 25 km of ocean from the shore (**Figure 1**). Few sets have been recorded further offshore than Lobos de Afuera. *Management effectiveness and compliance:* Unknown	*Social use:* Human consumption, but likely not for their eggs unless northern most nest sites in Tumbes, Peru are impacted. *Value:* Unknown *Target market if sold:* See leatherback target market if sold. *Cultural values:* See leatherback cultural values.

(Continued)

TABLE 5 | Continued

Species of concern	Lines of evidence		
	Biological factors	**Susceptibility to the fishery**	**Socio-economic outcomes**
Olive ridley turtle *Lepidochelys olivacea*	*Size:* 60–75 cm (2–2.5 feet). *Weight:* Up to 45 kg (100 pounds). *The average lifespan in the wild:* 50 years *Sexual maturity:* 10–18 years *Fecundity:* Commonly nest in successive years, one to three times per season, with ~ 100–110 eggs per clutch *Habitat:* Worldwide in tropical and warm oceanic and neritic waters. *Nesting sites:* Nesting occurs in nearly 60 countries worldwide. The southernmost nesting sites for the species have been reported in El Ñuro, Piura, Peru (Kelez et al., 2009), approximately 375 km from San Jose. *East Pacific RMU geographic extent:* Baja California Sur Mexico to southern Peru, the eastern Pacific and northwest of Hawaii (Wallace et al., 2010a). *East Pacific RMU population size:* 5000 (Wallace et al., 2010a). *Population trend (East Pacific RMU short and long-term/Global):* Stable short-term—Population in East Pacific RMU may have increased since the 1990s (Eguchi et al., 2007; Wallace et al., 2010a), long-term decreasing; global population decreasing.	*Catchability in the fishery:* **BPUE per trip in San Jose is 0.08 ± 0.46 (mean ± SD). 35 captures in San Jose inshore/midwater fleet from observer data between 2007 and 2017 (Table 6).** *Distributional overlap:* ~75% of the total area within the boundaries of the fishery (**Figure 1**). Reports of high capture rates in gillnets in northern fishing locations during key information interviews in San Jose. *Management restrictions:* poor—few restrictions are in place to support a reduction in the incidental take of green turtle. See leatherback section for further details. *Overlap in the effective fishing effort:* Most of fishing effort is concentrated in the first 25 km of the ocean from the shore (**Figure 1**). No recorded olive ridley captures were recorded further offshore than Lobos de Tierra (**Figure 1**). *Management effectiveness and compliance:* Unknown	*Social use:* See green turtle social use. *Value:* Unknown *Target market if sold:* See leatherback target market if sold. *Cultural values:* See leatherback cultural values.

Three species of sea turtle are known to be regularly captured in our case-study fishery, the leatherback turtle Dermochelys coriacea, green turtle Chelona mydas, and olive ridley turtle Lepidochelys olivacea. Text in ***bold*** *highlights collected data filling identified knowledge gaps. Italicized text indicates subcategories for lines of evidence.*

did not record any olive ridley captures further offshore than Lobos de Tierra indicating the potential for a more inshore distribution within the San Jose gillnet fishery's geographic extent (**Figure 1**). Vessels number 18–28 (**Table 4**) and fishing trips average 7.5 days (see **Supplementary Material**). Drawing on the lines of evidence of fishing effort, and a capture per trip rate of 0.08 (of which mortality rates were 21%, and capture release with injury rates were 25%), the annual mortality rates of olive ridley turtles in the IMG fleet are likely in the tens rather than the hundreds. This pattern highlights a "moderate" consequence (C2) signifying the bycatch impact from the IMG fleet is at a maximum level of acceptability, is "likely" (L4) to occur during the assessed period (**Table 7**). The evidence does not suggest that a "severe" (C3) consequence level of impact "may occur" (L3) or is "expected" (L4).

TABLE 6 | Observed sea turtle captures per trip in the San Jose inshore-midwater gillnet fleet from August 2007–May 2019.

		Per trip (*n* = 461)			
Turtle species	***n***	**Mean**	**SD**	**Min 95% CI**	**Max 95% CI**
Green	329	0.71	1.98	0.53	0.89
Leatherback	7	0.02	0.12	0.01	0.03
Olive ridley	35	0.08	0.46	0.04	0.12
Unidentified	8	0.02	0.21	0.00	0.04
Total turtle captures	379	0.82	2.10	0.63	1.01

CI = confidence interval. Mortalities and capture releases with injury are provided in text (see ***Supplementary Material*** *for the table format).*

The green turtle RMU (Pacific East) population has been estimated at 3750 individuals (Wallace et al., 2010a). The population trend is projected upward in the short-term, but downward in the long-term (Wallace et al., 2010a). We found green turtle presence was likely throughout the fleet's geographic extent (**Figure 1**). Green turtle capture rates are high at 0.71 per trip (**Table 6**). The observed mortality rate of green turtle bycatch was 7% and captures released with injury 30% (see **Supplementary Material**). These data show bycatch mortality rates of green turtles may have been occurring in the tens of turtles, to low hundreds of turtles per annum in the IMG fleet (Alfaro-Shigueto et al., 2010, 2018). These patterns imply that negative fishing impact from the IMG fleet will occur to more than a few individuals in most years over the assessed period (C1). The likelihood of this consequence occurring was ranked as "remote" (L1). The capture and inferred bycatch rates could be consistent with capture or impact occurring at the maximum acceptable level (C2), or that recovery "may be affected"/"further declines are generated" (C3 or C4). The estimated short-term rising trend in the Pacific East RMU population of green turtles (Wallace et al., 2010a) in combination with existing IMG fleet management measures to mitigate turtle bycatch imply that

TABLE 7 | Results of the consequence × likelihood qualitative ecological risk assessment.

Source of risk	Turtle species	Consequence level	Remote	Unlikely	Possible	Likely	Risk score	Final risk level
			L1	L2	L3	L4		
Inshore/midwater gillnet fleet	Leatherback	C1	×				1	EX
		C2	×	×	×		6	
		C3	×	×	×	×	12	
		C4	×	×	×	×	16	
	Green	C1	×				1	EX
		C2	×	×	×		6	
		C3	×	×	×		9	
		C4	×	×			8	
	Olive ridley	C1	×	×	×	×	4	MA
		C2	×	×	×	×	8	
		C3	×	×			6	
		C4	×				4	
Inshore gillnet fleet	Leatherback	C1	×	×	×		3	EX
		C2	×	×	×		6	
		C3	×	×	×	×	12	
		C4	×	×	×	×	16	
	Green	C1	×	×			1	EX
		C2	×	×	×		6	
		C3	×	×	×		9	
		C4	×	×	×		12	
	Olive ridley	C1	×	×	×	×	4	MA
		C2	×	×	×	×	8	
		C3	×	×			6	
		C4	×				4	

*Likelihoods (as indicated by × 's) for each of the consequence levels for the bycatch (mortality following incidental capture) of leatherback turtle Dermochelys coriacea, green turtle Chelonia mydas, and olive ridley turtle Lepidochelys olivacea in the San Jose inshore gillnet fleet, and the San Jose inshore/midwater gillnet fleet. The final risk level is based on the highest risk score calculated from multiplying consequence and likelihood scores. Turtle stock size was assessed at the East Pacific RMU scale. Fleet sizes were defined by the geographic maximum extent calculated (**Table 4**). Consequence levels and associated likelihoods are based on the lines of evidence for biological factors, potential overlap/susceptibility, simple catch and effort, current management restrictions, effective effort levels, social use, and cultural values (see **Table 5**). Consequence levels: 1 = minor, 2 = moderate, 3 = major, 4 = extreme. Final risk levels: MI = minor, MO = moderate, MA = major, EX = extreme.*

further declines to the RMU population from this fleet (C4) were not "likely" (L4) or "possible" (L3) but "unlikely" (L2) over the assessment period. We assigned this consequence of impact an indicative probability of 3–9%. Both the consequence levels of "stock recovery impact" (C3) and the "maximum level of acceptable bycatch occurring" (C2), were "possible" (L3) as further data were not available to reduce uncertainty.

Leatherback turtle capture rates were the lowest of the turtle species assessed at 0.02 per trip (**Table 6**), but this BPUE could still equate to >10 leatherback turtle captures per annum. The observed mortality rate of leatherback turtle bycatches was 14% (1/7). The remaining six captures were released alive without injury (see **Supplementary Material**). Leatherback turtle presence was considered "possible" through the IMG geographic extent (**Figure 1**). The leatherback turtle's Pacific East RMU population (ca. > 200) has an estimated decreasing mean growth rate of -0.156 (Mazaris et al., 2017). These data suggest that even a low amount of fishing-related mortality from the IMG fleet (i.e., only a few individuals per year) could "likely" (L4) result in further population declines (C4) and increase the chances of extinction of the Pacific East RMU population of leatherback turtles (Spotila et al., 2000).

Inshore Gillnet Fleet

We ranked the recovery of the leatherback, and green turtle, East Pacific RMU populations, as subject to "extreme" risk from the San Jose inshore gillnet fleet, and the olive ridley turtle East Pacific RMU as subject to "major" risk given the current management measures in place (**Table 7**). The IG fleet covers an area of 1200 km^2 in summer and 3700 km^2 in winter (**Table 5**). The geographic extent of the IG fleet is considerably smaller than the IMG fleet (**Figure 1**), but vessel numbers are higher. During our 2017 winter field season, we recorded 150 inshore gillnet vessels fishing in San Jose—this represents a tripling in fleet size since 1996 (Escudero, 1997). Unlike the IMG fleet, no fishery observer data for turtle captures exist for the IG fleet. This increased uncertainty when estimating the likelihood of consequences (Fletcher, 2014).

A significant overlap between the IMG and IG fleets exists (**Figure 1**). Captures of leatherback, green, and olive ridley turtles in the IMG fleet have been recorded within the geographic bounds of the IG fleet (**Figure 1**). These data show that turtle capture in the IG fleet is also probable. However, with only these data, the captures of sea turtles in the IG fleet remain unknown. Further insight can be gained from shore-based surveys investigating sea turtle bycatch in coastal fisheries across Ecuador, Peru, and Chile between August 2010 and March 2011 (Alfaro-Shigueto et al., 2018). San Jose, Lambayeque, Peru, was a survey site. In San Jose, 44 respondents, across both the IMG and IG fleets, acknowledged turtle bycatch in their gillnets. Of these 44 respondents, 43.2% reported green turtle bycatch, 25% leatherback turtle bycatch, and 20.5% olive ridley turtle bycatch (Alfaro-Shigueto et al., 2018). This pattern of data suggests that all the levels of consequence are "possible" for green, leatherback, and olive ridley turtle species (**Table 1**). While the geographic extent of the fishing fleet is small compared to each species wider East Pacific RMU distribution, the high vessel number, inshore distribution of the turtle species overlapping with the IG fleet's geographic extent (**Figure 1**), and high uncertainty resulted in "possible" (L3) and "likely" (L4) likelihoods for most of the consequence rankings (**Table 7**).

A final risk level of extreme or major is unacceptable unless further management actions are undertaken (Fletcher, 2014). This assessment highlights the need for further management actions in the San Jose gillnet fishery if the proposed target of reducing turtle captures by 15% every year for five years while maintaining total catch weight is the be achieved—this includes adding monitoring efforts to estimate baseline BPUE rates for each turtle species captured in the IG fleet. These BPUE estimates could then be compared to potential management strategies to reduce turtle BPUE in the future. For supplementary analysis, we present summary tables of evidence for the turtle species assessed (see **Supplementary Material**).

Potential Management Measures Based on the Mitigation Hierarchy

Based on information obtained from a literature search, we defined a list of potential management measures and categorized them according to the steps of the mitigation hierarchy. This list was refined to 13 potential management strategies during key informant interviews and FGDs (**Table 8**). Management strategies included an avoidance strategy (to reduce E_B of Eq. 1), eight minimization strategies (four spatial or temporal area closures and four technology or fishing behavior changes), two remediation strategies (to reduce BPUE of Eq. 1), and two biodiversity offsetting strategies (to increase O_T of Eq. 1).

The respondents in the inshore gillnet fleet's FGD (comprising 13 of 150 possible IG skippers, a local government scientist, and a local not-for-profit employee) disagreed with more of the potential preventative measures proposed that fell into the avoidance and minimization steps of the mitigation hierarchy and agreed with more of the compensatory actions that fell into the remediation and offsetting steps (**Figure 2**). The same trend is present for the IMG fleet's FGD (comprising 3 of 18 possible IMG skippers, and two local not-for-profit employees) but responses were more mixed. Fishers were in strong disagreement with the proposed avoidance strategy of phasing out gillnets in favor of alternative fishing gear such as trolling (a form of handline fishing). The inshore group strongly disagreed with at-sea capture reduction technologies such as LEDs on nets or shifting to buoyless nets. Responses to these measures were more distributed for the IMG group. Both groups were in strong agreement with participating in training workshops teaching better handling and release practices for captured turtles and most respondents agreed with the use of remote electronic monitoring onboard their vessels (**Figure 2**). Off-site compensation strategies such as biodiversity offsetting received mixed responses in both FGDs (**Figure 2**).

The Hypothetical Effectiveness of Management Options

In the final section of this study, we illustrate the integration of MSE and the mitigation hierarchy for use in a data-limited management scenario. We explore the performance of three "management scenarios" that each combines multiple management strategies in a preliminary and qualitative assessment to reduce the risk from the San Jose gillnet fishery posed to the recovery of leatherback, green, and olive ridley turtle populations defined by Pacific East RMUs.

Scenario Synopses

Scenario 1 considers the *status quo* management over 10 years between 2020 and 2030. The scenario maintains existing management strategies in the San Jose gillnet fishery and projects an expected level of expansion over 10 years. Management strategies include expanding the use of LEDs on nets (minimization) and remote electronic monitoring (remediation) from the five IMG vessels where these technologies are currently applied to all the vessels in the IMG fleet. Safe handling and release workshops held in San Jose continue (remediation). This scenario does not implement any management measures in the IG fleet (see **Supplementary Material**).

Scenario 2 takes a protectionist approach to sea turtles. The scenario implements a gear switching program that phases out gillnet for trolling (a form of handline fishing). A quarter of the San Jose fishery is proposed to undergo the gear switch every two and a half years (avoidance). The existing management actions in place in San Jose continue as expected in scenario 1 (e.g., LEDs on nets would be implemented on IMG vessels that continued to fish during the gillnet phase out-period).

Scenario 3 takes a more incentive-based approach, implementing multiple strategies spanning at-sea minimization, post-capture remediation, and off-site compensation actions. The scenario includes an effort restriction for all vessels operating in the IMG fleet. This limits the gillnet soak time to 6 h per set as opposed to the current 14.6 ± 3.9 h (Alfaro-Shigueto et al., 2010). This equates to an approximate halving of fishing effort across the IMG fleet (minimization). The scenario also includes a dynamic spatiotemporal marine protected area (MPA) for leatherback turtles (minimization). This will make use of a local two-way high-frequency (HF) radio program that allows fishers to receive and report real-time information on turtle sightings and captures (Alfaro-Shigueto et al., 2012). The *status quo* management strategies in scenario 1 are also enacted, but in this scenario, they integrate the IG fleet as well as the IMG fleet (e.g., LEDs on all nets, remote electronic monitoring systems

TABLE 8 | Potential management measures for mitigating sea turtle captures/mortalities in San Jose's small-scale gillnet fishery.

Mitigation hierarchy step	Management measure	Examples of use in existing fisheries management/policy, or, examples of use in a similar fisher. Effects on sea turtles are highlighted	Key references
Avoid—ensure spatiotemporal overlap does not occur; E_B	Gear trade-in initiatives swapping all gillnets to lobster pots or trolling gear (a form of hand line fishing).	In 2007, a gillnet gear trade in initiative was trialed with gillnet fishers in Trinidad, where 3000 entanglements of leatherback turtles were reported in the year 2000 (Eckert and Eckert, 2005; Lee Lum, 2006). Fishers were given training in how to use trolling gear consisting of outriggers, planers, fish finders, and bandit reels. At the conclusion of the 2007 field tests, fishers were presented with the results of the experiments and asked about their willingness to try new these new methods. Average daily trolling daily income was calculated at $406 (Trinidadian dollars) with no sea turtle bycatch, relative to $334 (Trinidadian dollars) per day with traditional nets. 90% of fishers said they would be willing to switch to trolling (Eckert et al., 2008).	Eckert and Eckert, 2005; Lee Lum, 2006; Eckert et al., 2008
Minimize—limit probability of capture in times/places of overlap; B_{DOA}	No-take MPA extending the fishing restriction in place around the islands of Lobo de Tierra and Lobo de Afuera from 5 to 15 nautical miles offshore the islands (a potential turtle hotspot), all year.	The Peruvian government implemented national marine reserves around 30 offshore islands including two, Lobo de Tierra and Lobo de Afuera, located 100 and 85 km from San Jose, respectively. National marine reserves in Peru only have an equivalent protection status to IUCN category VI protected areas, offering limited protection. A prohibition of bottom trawling exists that extends for 5 nautical miles from the islands' shoreline.	International Union for Conservation of Nature [IUCN], 2010b; United Nations Environment Programme World Conservation Monitoring Centre [UNEP-WCMC], and International Union for Conservation of Nature [IUCN], 2016
	A temporal gillnet ban (August–November) with gear switching to lobster potting or trolling during the gillnet ban period every year.	The Pacific Leatherback Conservation Area is a 250,000-square mile marine protected area off the California coast that is enforced during 3 months of the start of August to end of October when leatherbacks are present, shutting off all fishing including the California large-mesh drift gillnet fishery. Consideration of the spillover effects that resulted from the Pacific Leatherback Conservation Area is necessary when considering a time-area closure—notably biodiversity loss due to displaced fishing activity, displaced production activity, and trade leakages from an increase in imports to replace the displaced domestic production (Squires et al., 2016).	50 C.F.R. §660.713; Curtis et al., 2015; Squires et al., 2016

(Continued)

TABLE 8 | Continued

Mitigation hierarchy step	Management measure	Examples of use in existing fisheries management/policy, or, examples of use in a similar fisher. Effects on sea turtles are highlighted	Key references
	A dynamic gillnet ban shifting in space and time in relation to turtle movement (enacted with existing and available information).	In an effort to provide information to fill existing data gaps and support bottom-up monitoring of compliance, an information sharing scheme was started by not-for-profit *ProDelphinus* in the form of a high-frequency two-way radio outreach program to raise awareness of fishers at sea of bycatch, and to provide them with any requested information using real-time spatial management. Now partnered with the not-for-profit's *Asociacion* and *Pacifico Laud* the initiative covers twenty-five ports and extends over 3500 km from Manta, Ecuador to San Antonio, Chile.	Alfaro-Shigueto et al., 2012; Hazen et al., 2018; Squires et al., 2018
	An offshore distance restriction with gillnetting only allowed to occur between 0 and 2 nautical miles offshore.	No-take marine reserves are established, important conservation and management tools that have proven to have positive responses in far more cases than no differences or negative responses.	Halpern, 2003; Lester et al., 2009
	Soak time (effort) restriction of 6 h per set for the IMG fleet only.	Soak time for IMG vessels is 14.6 ± 3.9 h. This strategy equates to a rough halving of the IMG fleet's fishing effort.	Alfaro-Shigueto et al., 2010; Gilman et al., 2010
	Buoyless nets which entail removing the buoys from the float line of the net.	In 136 controlled sets of conventional (control) and buoyless nets (buoys removed from float line), buoyless nets reduced mean turtle bycatch rates by 68% while maintaining target catch rates and composition.	Peckham et al., 2016
	Fixed demersal nets only, surface driftnet ban.	Most gillnets in San Jose are surface drift nets, which take more turtle bycatch than fixed demersal nets. Reductions in bycatch of surface and near-surface swimming turtles would be expected.	–
	Light-emitting diodes (LEDs) on gillnets.	(i) *ProDelphinus* are running a trial bycatch reduction community co-management scheme in San Jose where participating skipper and crew use LEDs on their nets in an effort to reduce turtle bycatch. (ii) In Sechura Bay, northern Peru, 114 pairs of control and illuminated nets were deployed. The predicted mean catch per-unit-effort of green turtles was reduced by 63.9% in illuminated nets. (iii) Turtle capture rate was reduced by 39.7% in LED illuminated nets while having negligible impacts on target catch and catch value.	Wang et al., 2013; Ortiz et al., 2016

(Continued)

TABLE 8 | Continued

Mitigation hierarchy step	Management measure	Examples of use in existing fisheries management/policy, or, examples of use in a similar fisher. Effects on sea turtles are highlighted	Key references
Remediate–limit capture–related mortality once caught; B_{OB}, P_{DV}, P_{DR}	An annual workshop on safe handling and release procedures, which includes the resuscitation of sea turtles (estimates represent mortality reduction rather than encounter reduction).	(i) Post-capture, sea turtles that appear lifeless are not necessarily dead. They may be comatose. While turtles returned to the water before they recover from a coma will drown. A turtle may recover on board a boat once its lungs have drained of water. This could take up to 24 h. By following best practice handling and resuscitation guidelines unnecessary turtle deaths can be prevented. (ii) *ProDelphinus* run workshops training fishers on safe handling and release of bycatch turtles in San Jose and other SSF communities along Peru's coastline.	Food and Agriculture Organization [FAO], 2009 Bartholomew et al., 2018; Suuronen and Gilman, 2019
	Mandatory remote electronic monitoring on vessels to reduce possibility of turtle retention post-capture.	Remote electronic monitoring has been trialed on five San Jose boats with a total of 228 fishing sets monitored. Of these, 169 sets also had on-board fisheries observers present. The cameras were shown to be an effective tool for identifying elasmobranch catch > 90% detection rates, though variable for sea turtles (with 50% positively identified). As well as improving data, remote electronic monitoring has potential to reduce the high rate of illegal consumption of leatherback turtles (Alfaro-Shigueto et al., 2007).	Bartholomew et al., 2018; Suuronen and Gilman, 2019 Dutton et al., 2005; Janisse et al., 2010; Milner-Gulland et al., 2018
Offset—compensate for harm caused by residual bycatch mortality; O_T	Green bycatch (Pigovian) tax[1] that funds turtle nesting site protection e.g., unprotected smaller nesting sites in Peru, Ecuador, Costa Rica, or Mexico (depending on species).	(i) Positive trends have been reported in leatherback turtle populations over decades as a result of nesting site protection and egg relocation (Dutton et al., 2005). (ii) The California drift gillnet industry, in 2004, financed Pacific sea turtle nesting site conservation efforts in Baja California through a voluntary bycatch tax for compensatory mitigation of sea turtle bycatch. The funds were in part driven in an effort to slow further extensive time-area closures (Janisse et al., 2010).	Dutton et al., 2005; Janisse et al., 2010; Milner-Gulland et al., 2018; Squires et al., 2018

(Continued)

TABLE 8 | Continued

Mitigation hierarchy step	Management measure	Examples of use in existing fisheries management/policy, or, examples of use in a similar fisher. Effects on sea turtles are highlighted	Key references
	Payment-in-kind program with fishers contributing their time, resource and knowledge for conservation efforts in San Jose or the wider northern region of Peru [e.g., reporting all leatherback turtle sightings and captures to the local government science (IMARPE) officer, contributing hours to protected green and olive ridley nesting sites in El Ñuro, Piura, Peru, or monitoring marine reserves for illegal fishing]	The Kiunga Marine National Reserve Conservation and Development Project is a partnership between the Kenya Wildlife Service (KWS) and World Wildlife Fund (WWF) pays local women to report turtle nests and sightings of nesting turtles to KWS or WWF employees. In exchange they are paid upon report verification and a payment conditional on hatching success is also made. Nest translocation is high (∼70%) because they are located below the high-tide mark or at other risks of depredation (Flintan, 2002; Ferraro and Gjertsen, 2009).	Flintan, 2002; Ferraro and Gjertsen, 2009

Here we limit potential management measures to 13. Additional management strategies could be evaluated in successive evaluation rounds. An effort was made to ensure representation of management strategies to address the negative anthropogenic impact that occurs throughout the life cycle of each of the sea turtle populations of management concern. [1]A green bycatch (Pigovian) can be a double dividend tax, acting as both as an offset and minimization strategy. The tax minimizes bycatch by internalizing the external costs of bycatch (for both consumers and producers as part of the tax is passed up the supply chain, depending upon the price elasticities of demand and supply). The first dividend is the welfare increase (including conservation) from minimization through the bycatch tax and the second dividend, and an additional source of welfare increase (including conservation), comes from the offset (Squires et al., 2018).

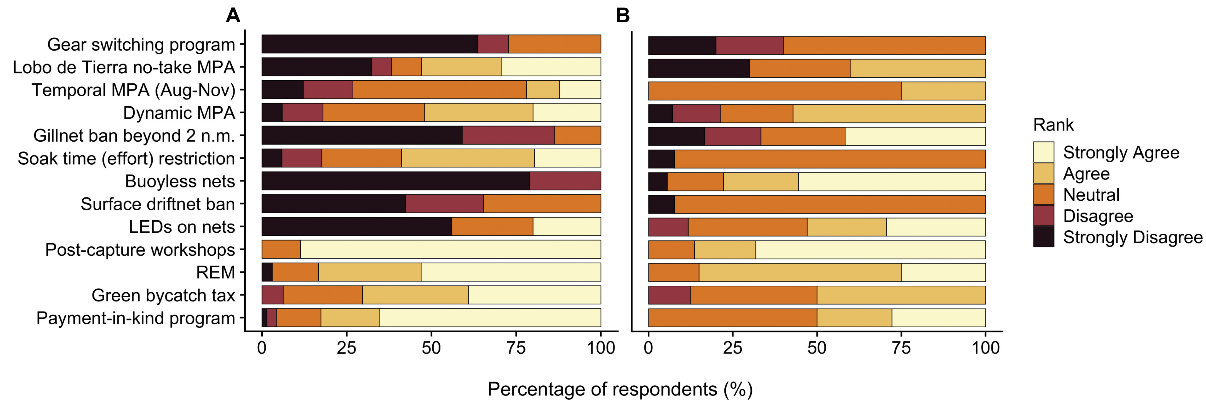

FIGURE 2 | Stakeholder preference for the evaluated turtle bycatch reduction strategies. Two focus group discussions (FGDs) were run. FGD **(A)** concentrated on the inshore gillnet (IG) fleet and comprised 15 respondents, including IG skippers (n = 13), a regional government scientist (n = 1), and a not-for-profit employee (n = 1). FGD **(B)** focused on the inshore-midwater gillnet (IMG) fleet and comprised five respondents, including IMG skippers (n = 3) and local not-for-profit employees (n = 2). REM = remote electronic monitoring. For full strategy descriptions, refer to **Table 8**.

on all vessels, and continued implementation of safe handling and release workshops across the San Jose gillnet fishery). To support further population recovery for the turtle populations impacted by our case study fishery, this scenario implements a green bycatch (Pigovian) tax as a biodiversity offset (**Table 8**; Squires et al., 2018). The tax applies to leatherback, green, and olive ridley turtles captured in an eastern Pacific pelagic longline fishery (e.g., Donoso and Dutton, 2010). The means to negotiate this tax in practice goes beyond the scope of the hypothetical scenario assessed here, but volunteer bycatch taxes have been implemented by large-scale commercial fishing fleets before (e.g., a turtle bycatch tax through the California Drift Gillnet Fishery funding nesting site protection implemented by the Mexican non-profit organization *Asupmatoma* A.C; Janisse et al., 2010). In the present scenario, funds from the tax support the monitoring of leatherback secondary nesting sites in Costa Rica, where illegal egg poaching can still occur (e.g., Ostional; Santidrián-Tomillo et al., 2017). Olive ridley turtles also nest in Ostional, Costa Rica, offering the potential for conservation actions at a single site to support the population recovery of two of the three turtle populations incidentally captured by the San Jose gillnet fishery.

Evaluation of Scenarios

Scenario 1 ("the status quo") presents the existing management of the San Jose gillnet fishery between 2020 and 2030 (see **Supplementary Material**). In this scenario, the turtle bycatch issue is expected to worsen because of a lack of management measures restricting fishing effort (**Figure 3**). With no effective effort restriction in place (such as a TAC to reduce target fish catch per unit effort; CPUE), the incidental take of sea turtles is expected to increase as the IG fleet grows in vessel number and the San Jose gillnet fishery as a whole expands in geographic extent and fishing effort (Guevara-Carrasco and Bertrand, 2017; Castillo et al., 2018). Despite increasing fishing effort (e.g., distance traveled) and fleet number (**Table 4**), we projected the target fish CPUE to trend downward in line with historical catch trends for the Lambayeque region of Peru (Guevara-Carrasco and Bertrand, 2017). The expansion of existing turtle bycatch mitigation measures trialed in the fishery (LEDs on nets reduce B_{DOA} of equation 2, and remote electronic monitoring and better handling practices reduce P_{DV}) are expected to reduce turtle BPUE rates for individual vessels, and remote electronic monitoring is expected to improve data paucity of turtle capture, bycatch, and consumption rates. Discard rate across a fishery is strongly influenced by shifts in individual human behavior, so the uncertainty in our projected trend is high (e.g., Smith et al., 2004). We drew on data that indicates LEDs on nets have little impact on the volume of target catch (Ortiz et al., 2016). This was supported by our field observations where we noted that San Jose fishers retain all but the smallest fish species for use and sale at markets—which is supported by regional catch reports (Guevara-Carrasco and Bertrand, 2017). These data highlight that current trends in discards are likely to persist under scenario 1. As fishing effort across a larger geographic extent is expected, the impact on habitat and sessile communities is predicted to have a slight upward trend (**Figure 3**).

We predicted that the overall management cost of this scenario would follow an increasing trend because of the expansion of LEDs on nets and remote electronic monitoring across the IMG fleet (**Figure 3**). Costs supporting our estimate came from price estimates reported from controlled gear trials of LEDs on nets and remote electronic monitoring in the local fishing system (Ortiz et al., 2016; Bartholomew et al., 2018). The IG fleet remains for all intents and purposes, an open-access fishery (cf. Supreme decree N° 018-2010-PE). Despite the ban on new vessel builds, we expect the IG fleet to expand in line with historical trends over the assessment period (**Table 4**). We predicted that an expanding IG fleet will decrease the stability of management across the San Jose gillnet fishery as a whole. The cost per day fished in expected to increase as distance traveled increases, forcing a higher consumption of fuel per vessel. Declines in food, employment security, and fish processing follow declining CPUE estimates (Guevara-Carrasco and Bertrand, 2017). We predicted

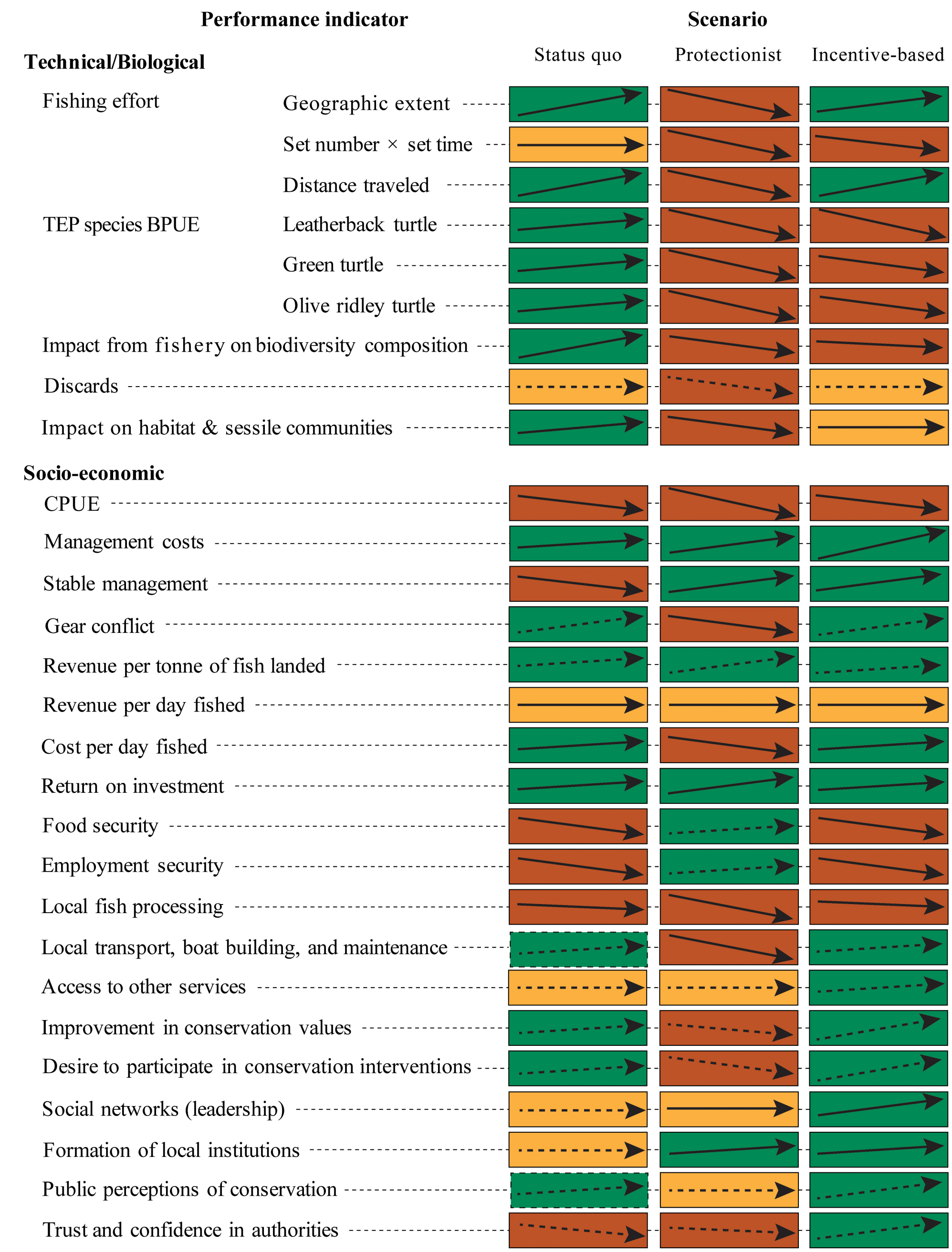

FIGURE 3 | Trends over 10 years in indicators for each of the three management scenarios in the San Jose gillnet fishery (based on informed opinion). Green boxes indicate predicted positive trends, yellow boxes indeterminate trends, and red boxes declining trends. Dotted lines represent high levels of uncertainty in the presented trend direction. We present a management summary table with a full list of the management measures contained in each scenario (see Supplementary Material).

an increasing IG fleet will drive positive trends in local transport, boat building, and maintenance, but uncertainty remains high due to the potential to increase enforcement of the ban on new vessels. Access to other services is not well known but predicted to remain stable with high uncertainty. We predicted that the expanding conservation interventions (e.g., LEDs on nets and participatory workshops) will lead to a small improvement in local conservation values. Our survey data show IG skippers disagreed with LEDs on nets but that remote electronic monitoring and training workshops had a stronger agreement (**Figure 2**). We predicted perceptions to improve over time as the existing interventions in the IMG fleet expand to the other fishers in the fleet, but this trend remains highly uncertain and warrants further investigation. Local leadership, the formation of new social organizations (e.g., a new fishing collective), and trust and confidence in authorities are not well known and are predicted as indeterminate or in decline as the current scenario does little to investigate or manage the improvement of these social conditions.

Scenario 2 ("the protectionist") reduces the negative fishing impact on sea turtles (reduced E_B of Eq. 1) over the management period (**Figure 3**). We predicted that the area fished will increase as the gillnet fishery continues to expand in effort due to a predicted decrease in target catch following historic catch trends (Guevara-Carrasco and Bertrand, 2017). We anticipate management costs to increase as the initiative expands. We considered the possibility of an increase in the gross value of the fish product as handline caught fish can offer a more sustainable consumer choice (e.g., Eckert and Eckert, 2005; Eckert et al., 2008), but the uncertainty surrounding consumer interest and willingness to pay for a more sustainably sourced fish product in our case study fishing system remains high. We predicted that conflict between gillnet, trawl, and purse seine fisheries operating in the area will decline as gillnet are traded in. Transportation, boat building and maintenance resulted in downward trends as local fish processing and indirect income decline due to trolling bringing in a lower abundance of fish products over gillnets to process (Eckert and Eckert, 2005; Eckert et al., 2008). We expect a steady decline in turtle BPUE (reducing B_{DOA} of Eq. 2) as trolling (handline fishing) takes no, or very little turtle bycatch. Public perception of conservation is highly uncertain. We predicted long-term economic improvement, but the decline in secondary fishery services anticipated to occur in the community over the short- to medium-term may negatively affect this predicted upward trend.

Scenario 3 ("the incentive-based") attempts a more balanced approach to mitigating the negative fishing impact from the San Jose gillnet fishery on sea turtles (**Figure 3**). We predicted fishing effort to continue to increase, but not as rapidly as in scenario 1, because of the effort restriction halving the allowable set time in the IMG fleet (see **Supplementary Material**). We predicted that the effort restriction will lead to an initial decline in CPUE, which would rapidly level out over the remaining management period. We projected declines in turtle BPUE (through a reduction in B_{DOA} of Eq. 2). We estimated a steeper decline for leatherback turtles over the green and olive ridley turtles because of the dynamic leatherback turtle MPA in this scenario. Our data show support from most gillnet fishers' in San Jose for the best handling and release practice workshops so we estimated high compliance and an associated small but measurable increase in post-capture turtle survival rates contributing to the declining turtle BPUE (**Figure 2**; reducing P_{DV}, and potentially P_{DR} in Eq. 2). It was also assumed that remote electronic monitoring if expanded would result in wide uptake on gillnet vessels in San Jose (**Figure 2**; reducing P_{DV}). The green bycatch tax can act as a double dividend. The first dividend comes from the tax incentivizing fishers in the large-scale pelagic fishery to change their fishing behavior in favor of mitigating turtle bycatch. The second dividend comes from the funds that the tax produces supporting nesting site protection for leatherback turtles (secondary nesting site) and olive ridley turtles (major nesting site) in Ostional, Costa Rica (Squires et al., 2018). Predicting any meaningful shift in population trends from funding the monitoring of a single nesting site in Costa Rica is difficult over the ten year assessment period (O_T of Eq. 1). Additional conservation action protecting secondary nesting sites for the East Pacific population of leatherback turtles form an integral part of planning any holistic conservation and population recovery plan for this species (Santidrián-Tomillo et al., 2017). Over longer periods (e.g., 20 + years), nesting site protection has driven long-term population recovery in several sea turtle populations (e.g., Chaloupka, 2003; Balazs and Chaloupka, 2004; Dutton et al., 2005; Troëng and Rankin, 2005). We predicted that the likelihood of public perception and fishers' desire to be part of management strategies in the future will improve, but this trend is uncertain despite scenario 3 integrating multiple strategies that fishers supported (**Figure 2**). These trends were strongly influenced by the expected decline in food and employment security expected in San Jose. Scenario 3 has the most diverse suite of bycatch reduction strategies, thus high management costs for this scenario were estimated (**Figure 3**).

DISCUSSION

We applied the mitigation hierarchy for fisheries management and marine megafauna bycatch reduction (Milner-Gulland et al., 2018; Squires et al., 2018) to the San Jose gillnet fishery where sea turtle captures are a known conservation issue (Alfaro-Shigueto et al., 2018). Working through the proposed steps of the mitigation hierarchy framework, we characterized our case-study fishery and the species of management concern. This helped prioritize research quantifying the fishery's geographic extent across fishing seasons. We identified gaps in fishery-specific turtle capture and bycatch rates, prompting us to calculate capture rates per trip for the turtle species regularly impacted by the IMG fleet. We then assessed the risk from the case study fishery (both IG and IMG fleets) on the turtle species of management concern based on a proposed qualitative turtle bycatch reduction target to contribute to a wider high-level population recovery goal (Fletcher, 2014). Drawing on the existing information collated and newly filled knowledge gaps, we compiled a list of 13 hypothetical management options to reduce key sources of anthropogenic-impact posed to the turtle populations of management concern. We then used fisher perceptions and a qualitative MSE framework to carry out a preliminary exploration of possible management scenarios by considering estimated trends for a range of biological, social, and economic indicators.

The wide migratory range of the turtle species assessed means that they spend much of their lives in waters or on beaches under other nations' jurisdictions. This necessitates a wider international effort to manage transboundary externalities for this species at ecologically relevant levels (Dutton and Squires, 2008). While we focused the current study on direct fishing impacts from a single small-scale fishery, the use of the mitigation hierarchy as an overarching framework encouraged consideration of a range of potential management strategies, from precautionary avoidance and minimization measures at-sea to supporting compensatory actions that seek to mitigate negative impacts from both large-scale pelagic fisheries, and those that occur at terrestrial-based nesting sites (**Table 8**). The framework helped drive simultaneous consideration of biodiversity losses and gains. This, in turn, allowed us to demonstrate the integration of a diverse set of management processes and tools to achieve a specific, qualitative target. This integration demonstrates how actions that are undertaken across a wide variety of fisheries and associated management structures might be summed together to evaluate progress toward high-level population recovery goals for depleted populations of marine megafauna species.

We supplemented the ERA using the turtle capture rates calculated for the IMG fleet, and existing research investigating turtle captures in Peru's coastal gillnet fisheries (e.g., Alfaro-Shigueto et al., 2007, 2018). Our analysis shows that the fishing impact from two gillnet fleets, which launch from a single port, could generate further declines of the Pacific East RMU populations of green and leatherback turtles (**Table 7**). San Jose is one of the major gillnetting ports in Peru but comprises only one of 106 landings sites or ports along the country's coastline (Castillo et al., 2018). While this assessment remains qualitative, it highlights the immediate need for additional management action to address the risk of local extinction for the Pacific East RMU leatherback turtle population (Spotila et al., 2000; Mazaris et al., 2017).

Integrating a qualitative MSE process with the mitigation hierarchy framework allowed for a preliminary evaluation of potential management scenarios incorporating a mix of turtle bycatch reduction strategies in a data-limited fishery. The assessment of how a diverse range of biological, technical, and socio-economic indicators might change through time allows for trade-offs between management goals to be transparently assessed. The trends estimated in the predictive performance indicators demonstrated that further management action is necessary to mitigate the negative impact on sea turtle populations from the San Jose gillnet fishery. The results also demonstrated that none of the three bycatch reduction scenarios presents a straightforward management picture. We predicted a wide variety of biological, economic, and social shifts across the three management scenarios evaluated. Our results provide some insight into how a range of management measures aimed at reducing turtle captures and mortalities could impact fishers, the wider San Jose community, and indirectly on biodiversity. However, based on our available data, the uncertainty in many of the predicted trends was high, particularly concerning the social indicators (**Figure 3**). Our results highlight the need for further integrating natural and social science in marine ecosystem-based management research (Alexander et al., 2018).

In several instances, it was easy to predict indicator trends under one of the three management scenarios evaluated (e.g., expecting green turtle BPUE in gillnets to decrease across the San Jose gillnet fishery as vessels switched from gillnets to handline trolling—scenario 2). In most cases, predicting the trends was difficult and uncertain based on the data available. We required an iterative process where the project team (the authors) assessed conflicting inputs to come up with the best guess of the likely trends (Smith et al., 2004). The assessment combined trends across the two gillnet fleets (i.e., IMG and IG), with weightings or emphasis applied to each fleet largely based on the project team's knowledge of the fishery and the collated and collected data. We found that emphasis on any particular input (i.e., the efficacy of the proposed dynamic spatiotemporal MPA for leatherback turtles) often had a sizeable influence on the trajectory of the trend in the indicator.

The varying experiences and personal biases each member of the project team brought to the assessment meant that several different trends in an indicator could result depending on an individual's interpretation. We undertook an iterative evaluation process aimed to address any difference in opinion. These web-based discussions allowed team members to highlight differences in interpretation. Comprehensive face-to-face workshops guided using structured question protocols and feedback would have improved the project team's ability to address different interpretations (Valverde, 2001; Burgman et al., 2011). The project team comprised representatives from academia, government, and a not-for-profit organization. We acknowledge additional bias in the overall experience of the group toward a conservation science and fisheries science background. Recognition that these biases may influence the qualitative assessment is vital and points to the importance of seeking a diverse range of stakeholder inputs across multiple sectors (e.g., industry, local community members, local government, not-for-profit organizations; Smith et al., 2004). Our experience of undertaking the assessment highlights the necessity for a quantitative evaluation of management scenarios—this could be a mid-term goal for supporting effective mitigation of turtle captures and mortalities in the San Jose gillnet fishery (e.g., Smith et al., 2004; Fulton et al., 2011a).

Numerous management options could integrate under the umbrella of the proposed mitigation hierarchy framework. While we made every attempt to include consideration of management strategies that addressed the negative anthropogenic impact that occurs throughout the life cycle of each of the sea turtle populations of management concern, many fishery management strategies were not evaluated. For example, implementation of a TAC on target fish species is a primary management mechanism in many fishery management frameworks (Gordon, 1954; Karagiannakos, 1996; Marchal et al., 2016). The decision to not include a TAC in any of the management scenarios was made because setting TACs for multiple, individual target species within a mixed-stock fishery must be carefully evaluated (Squires et al., 1998). Such an evaluation went beyond the scope of this study. Instead, we chose to include a simple effort restriction as part of scenario 3, in the form of halving the soak time within the IMG fleet. However, we note that the evaluation of proposed TACs for multiple species in a qualitative MSE process is achievable (Smith et al., 2004; Dichmont and Brown, 2010).

In collating and collecting information about the San Jose gillnet fishery and bycatch species group of management concern, case-specific issues arose. For example, 33% of San Jose gillnet fishers who self-reported turtle captures also noted that they consume turtles (Alfaro-Shigueto et al., 2018). These findings are supported by the report of 133 leatherback turtles caught between 2000 and 2003 off the coast of Salaverry (**Figure 1**). Of these captured leatherbacks, 41.4% were released alive, and 58.6% were retained for human consumption (Alfaro-Shigueto et al., 2007). These data highlight the need for an intervention focused toward shifting social norms and cultural values away from the consumption of turtle meat and toward alternate food sources. This could potentially be integrated as an offsetting measure (e.g., campaigns to engender pride in conserving turtles funded by a bycatch tax). Such an approach could be supported by compliance and monitoring in the form of the proposed expansion of remote electronic monitoring devices (**Figure 3**; Bartholomew et al., 2018).

We classified only one avoidance management strategy (a gear trade-in initiative swapping all gillnets for lobster pots or trolling equipment), out of the 13 management strategies evaluated. When we developed the theory for applying the mitigation hierarchy to fisheries management and bycatch mitigation, any spatial, temporal, and spatiotemporal area closures were classified within the avoidance step of the mitigation hierarchy (Milner-Gulland et al., 2018). Equation 1 of the mitigation hierarchy stipulates that avoidance measures ensure no spatiotemporal overlap occurs between the impacting risk and the species unit of management concern, thereby reducing E_B (Milner-Gulland et al., 2018). Thus, true avoidance measures require that the impacting fishing activity in question does not overlap with the bycatch of management concern (or has a very low likelihood of occurring; Booth et al., 2019). Because of this, spatiotemporal area closures avoid only if they are large enough or dynamic enough to ensure that fishing impact on the assessed unit of the species of management concern does not occur. For highly migratory marine megafauna such as sea turtles, this means that small spatiotemporal closures may displace the fishing impact to areas where turtles may still be located, thus creating a marginal benefit rather than ensuring the fishing impact is avoided (Halpern et al., 2004; Agardy et al., 2011). We contend that consideration must be given to the size of the proposed spatiotemporal closure in regard of the size of the assessment unit for the species of management concern. Only following this consideration should management measure be classified in the mitigation hierarchy accordingly.

We identified and filled several knowledge gaps in the current analysis, but other knowledge gaps present more substantive uncertainties and a more comprehensive data gathering process. For example, we had limited understanding of how the proposed management strategies will perform in our case-study system (except for net illumination and remote electronic monitoring; Ortiz et al., 2016; Bartholomew et al., 2018). Several trends estimated in the qualitative MSE were more uncertain as a result (**Figure 3**). In data-limited fisheries management situations such as the current study, it is often necessary to draw on elicited knowledge from fishers and local practitioners to support evaluations. Structured elicitation methods such as the IDEA protocol offer robust frameworks to reduce cognitive biases and more accurately quantify uncertainty (Hanea et al., 2016; Arlidge et al., 2020). Elicited data can then be used with fishery-specific costs of management strategy implementation, alongside consideration of the social implications of implementation.

Finally, a fully quantitative application of the mitigation hierarchy (Eq. 1) would also require an understanding of the relationship between population growth rates and bycatch rates. This was not achievable in our case study and will be challenging for many fisheries and species, particularly those in data-limited situations. As such, targets based on population growth may need to be the "gold standard," with more realistic measurable targets, such as those based on total catch or BPUE, used in the interim.

CONCLUSION

We present a case study application of the mitigation hierarchy to evaluate management options to mitigate sea turtle captures and reduce bycatch in a small-scale gillnet fishery in northern Peru. The conceptual overarching framework provided by the mitigation hierarchy helped integrate a range of fisheries management processes toward a fishery-specific quantitative target that feeds into a wider goal for biodiversity (Milner-Gulland et al., 2018; Squires et al., 2018). In data-limited fisheries like our case study, such goals remain aspirational, yet this framing clarifies how local-scale management action can translate to higher level goals for biodiversity. The proposed framework supported explicit consideration of uncertainties and highlighted future areas of research before implementing a more comprehensive assessment of management strategies in the future. The mitigation hierarchy's step-wise precautionary approach toward biodiversity encouraged a more holistic appraisal of management actions to address the negative fishing impact to sea turtles from the San Jose gillnet fishery. The framing of management options within the context of the hierarchy helped with consideration of preventative and compensatory measures throughout the life cycle of each turtle species of management focus. Integrating the mitigation hierarchy framework with MSE offers potential, as both qualitative and quantitative assessments can be undertaken, catering to a wide suite of potential fisheries. It also demonstrates how the mitigation hierarchy can add value to existing methods and procedures established within existing fisheries management processes. In identifying and filling key knowledge gaps and considering the socio-economic implications of a diverse suite of management strategies, the mitigation hierarchy shows the potential for supporting effective fishery-specific solutions that translate to aspirational national and international biodiversity goals.

DATA AVAILABILITY STATEMENT

All datasets generated for this study are included in the article/**Supplementary Material**. The survey data in anonymized form and R code underlying this research can be found at https://osf.io/ze5pw/.

ETHICS STATEMENT

The studies involving human participants were reviewed and approved by the University of Oxford (CUREC 1A; Ref No: R52516/RE001 and R52516/RE002). The patients/participants provided their written informed consent to participate in this study.

AUTHOR CONTRIBUTIONS

WA, DS, and EM-G conceived the manuscript. WA wrote the first draft. JA-S and JM supported the data analysis of the observer database. JA-S, HB, and JM contributed to the development of the framework's application in the case study fishery. All authors contributed significantly to revising the manuscript.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2020.00049/full#supplementary-material

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frontiers in Marine Science

BRIEF RESEARCH REPORT
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Novel Insights Into Gas Embolism in Sea Turtles: First Description in Three New Species

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The recent finding of gas embolism (GE) and decompression sickness (DCS) in loggerhead sea turtles (*Caretta caretta*) in the Mediterranean Sea challenged the conventional understanding of marine vertebrate diving physiology. Additionally, it brought to light a previously unknown source of mortality associated with fisheries bycatch for this vulnerable species. In this paper, we use ultrasonography to describe GE in a leatherback sea turtle (*Dermochelys coriacea*), a green sea turtle (*Chelonia mydas*), and an olive ridley sea turtle (*Lepidochelys olivacea*) from accidental capture in a gillnet, bottom trawl, and pair-bottom trawl, respectively. This is the first description of this condition in these three species worldwide. These cases of GE suggest that this may be a threat faced by all sea turtle species globally.

Keywords: sea turtle, gas embolism, decompression sickness, fisheries, bycatch

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INTRODUCTION

The incidental capture of sea turtles by fisheries, referred to as bycatch, is recognized as the greatest threat for the conservation of the species within this group worldwide (Wallace et al., 2010). Currently, six out the seven sea turtle species are listed by the International Union for Conservation of Nature as vulnerable, endangered, or critically endangered[1] (accessed 28 April 2020). Despite this, accurate assessment of the global effects of bycatch is extremely challenging (Lewison et al., 2004). The size of global fishing fleet is too numerous to be monitored effectively, especially considering that much fishing occurs in international waters and many undocumented boats (Lewison et al., 2014). To address the issue of sea turtle bycatch therefore requires research and collaborative efforts among scientist, conservationists, industry, and managers (Lewison et al., 2004). The annual average number of sea turtle bycatch reported globally between 1990 and 2008 was 4722 turtles, but estimations of true total bycatch numbers are of two orders of magnitude higher given that only 1% of fleets report these data (Wallace et al., 2010).

A review of sea turtle bycatch in the Mediterranean Sea estimated over 132,000 captures and 44,000 mortalities per year (Casale, 2011). The fishing gear that caused the most captures was pelagic longline, followed by bottom trawl, setnets (single netting wall set stationary on the bottom),

[1]www.iucnredlist.org

and gillnets (single, double, or triple walls near the surface, in midwater or on the bottom mounted together on the same frame ropes; FAO, 2001), and demersal longline, but set nets and gillnet resulted in higher mortalities than bottom trawlers (Casale, 2011). In the Atlantic Ocean, the trawl fishery industry in Gabon is responsible for an estimated annual bycatch of around 1026 olive ridley turtles with an estimated mortality ranging from 63 to 794 turtles per year, endangering the local breeding population (Casale et al., 2017). In the Pacific Ocean, mortalities of leatherback sea turtles by swordfish gillnet fisheries in Chile and Peru have contributed to the collapse of the Mexican Pacific coast breeding colony (Oravetz, 1999; Spolita et al., 2000).

In a global assessment of sea turtle bycatch in different gear, mortality rates were significantly higher in nets and trawls than longlines, emphasizing the need to mitigate bycatch impact by this gear (Wallace et al., 2013). Gear fixed to the bottom had higher mortality rates, although not statistically significant, when compared to gear used closer to the surface (Wallace et al., 2013). Some regions of the planet such as the southwest Atlantic Ocean and the Mediterranean Sea have been identify as "hotspots of bycatch intensity" for sea turtles (Lewison et al., 2014), although the authors highlight the widespread lack of data (Wallace et al., 2013).

It has traditionally been accepted that bycaught sea turtles either suffer from drowning and/or lesions caused by fishing gear (Casale, 2011). Recent research, however, has demonstrated that loggerhead sea turtles (*Caretta caretta*) caught by trawl- and gillnets have a high prevalence of intravascular gas or gas embolism (GE) (García-Párraga et al., 2014). Additionally, some of them presented clinical signs consistent with decompression sickness (DCS) (García-Párraga et al., 2014), a syndrome caused by the formation of intra- and extra-vascular gas bubbles when the summation of total dissolved gas exceeds local absolute pressure (Vann et al., 2011). In human hyperbaric medicine, Doppler flow transducers or two-dimensional echocardiography are the most common techniques used to detect intravascular gas bubbles (Mollerlokken et al., 2016). Although the detection of GE is not diagnostic of DCS, large bubble loads correlate with the probability of DCS (Evans et al., 1972; Neuman et al., 1976; Spencer, 1976; Gardette, 1979; Sawatzky, 1991), hence detection of GE by Doppler flow transducers or two-dimensional echocardiography are used as indicators of decompression stress (Pollock, 2007; Mollerlokken et al., 2016). The prevalence of GE in turtles caught accidentally in the waters of southern Brazil has recently been assessed onboard with surprising data on the development of intravascular gas in 100% of the individuals evaluated by ultrasonography (Parga et al., 2020).

In vitro studies of the vasoactive characteristics of the pulmonary and systemic arteries of loggerhead sea turtles suggest that nitrogen might accumulate and form gas bubbles as a result of an elevation of the sympathetic tone during the entanglement that, in turn, increases pulmonary blood flow and nitrogen uptake (García-Párraga et al., 2018). As this condition may affect post-release survivorship, this means the conventional estimates of bycatch mortality rates are underestimates. Thus, knowing the prevalence of GE and DCS in fisheries around the world is essential to quantify the true impact of fisheries bycatch on sea turtles worldwide.

To date, GE has mainly been recorded in loggerhead turtles in the Mediterranean yet there considering that sea turtle bycatch is a global problem for all sea turtle species, we postulate that this issue may be more widespread than currently thought. Here, we describe the diagnosis of GE in three sea turtle species, leatherback sea turtle (*Dermochelys coriacea*), green sea turtle (*Chelonia mydas*), and olive ridley (*Lepidochelys olivacea*) for which this has never been described before. This variety on species together with the new description outside of the Mediterranean Sea highlights the relevance for its consideration in the assessment of the effects of bycatch on all sea turtles and management plans worldwide.

MATERIALS AND METHODS

In October 2015, a bottom trawler fishing boat caught a live leatherback sea turtle (**Figure 1A**) along the Spanish Mediterranean coast (Castelló de la Plana, 39°58′59.99′′N, 0°1′59.99′′E). In May 2016, a coastal gillnet caught a live green turtle (**Figure 1C**) off the Spanish Mediterranean coast (Perellonet, 39°18′26.068′′N, 0°17′47.049′′W). Both animals received health examinations under the authorization of the "Conselleria d'Agricultura, Desenvolupament Rural, Emergència Climàtica i Transició Ecològica" to the Oceanogràfic Aquarium of the City of Arts and Sciences of Valencia for coordinating veterinary stranding response. The leatherback turtle was evaluated onboard once the fishing boat reached the harbor and then released back in the open ocean. The green turtle was transported and examined at the veterinary facilities of the Oceanogràfic aquarium. Additionally, in January 2017, a pair-bottom trawler in the Southern Atlantic Ocean (off the coast of Southern Brazil), caught a live olive ridley turtle (**Figure 1E**). This animal received a health examination on deck as part of a study carried out specifically to assess the prevalence and immediate evolution of DCS in incidentally captured turtles under the permits 15962-6 and 15962-7 from the Brazilian government (Parga et al., 2020).

For each turtle, we measured minimum curved carapace length (CCLmin) from the nuchal notch to the posterior tip of the caudal scute (Wyneken, 2001) or peduncle (Robinson et al., 2017). Veterinary evaluation included routine physical and neurological examinations, blood analyses, and imaging studies (García-Párraga et al., 2014). A portable ultrasound machine [General Electric Logiq E Vet ultrasound (GE Medical Systems)] with commercial 4C-RS (convex) for large individuals and 8C-RS (microconvex) and 12L-RS (linear) probes for small and medium size was used to evaluate kidney area (**Figure 1A**), cardiac chambers, major vessels at the base of the heart and liver in smaller animals where the ultrasound beam had enough penetration. A convex probe was placed in the prefemoral acoustic window of the inguinal area in the leatherback turtle with a penetration depth of 20 cm to reach kidney and adjacent vessels. A linear probe was used for kidney in the green turtle and olive ridley (4–6 cm penetration depth) and a microconvex

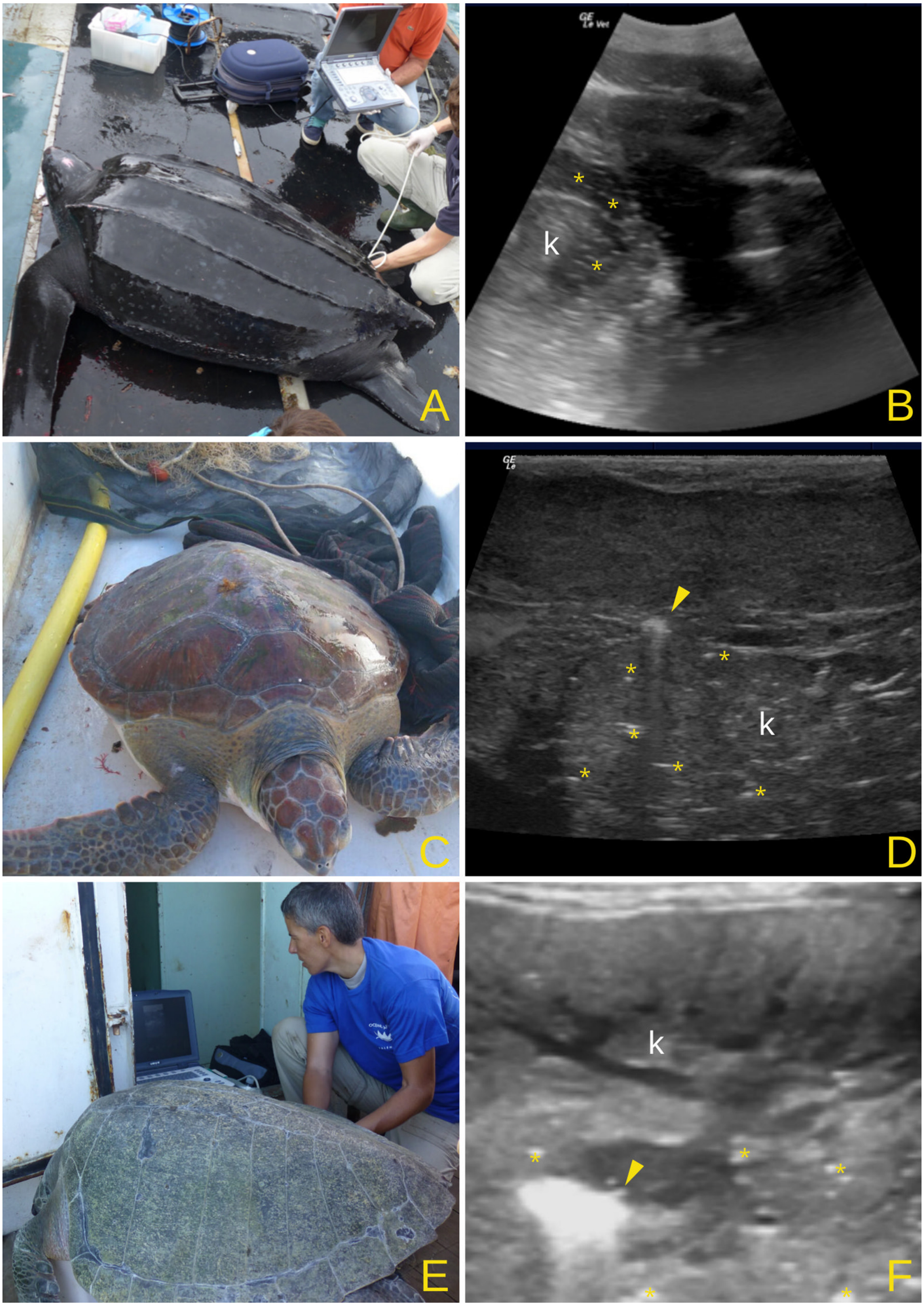

FIGURE 1 | Leatherback turtle **(A)**, green turtle **(C)**, and olive ridley **(E)** on the deck of the fishing boats. Kidney (k) ultrasound in the leatherback **(B)**, green **(D)**, and olive ridley **(F)** turtles showing the presence of gas as a hyperechoic (white) artifacts. Asterisks mark intravascular **(B)** or intra-parenchymal **(D,F)** gas seen as small hyperechogenic spots (small bubbles). Larger accumulations of gas usually produce comet tail artifacts on ultrasound examination (arrowhead).

probe to reach liver, heart, and great vessels in the green turtle (8–10 cm of penetration depth). In the three animals, the degree of GE was evaluated through ultrasound following the protocols outlined in García-Párraga et al. (2014). Based on the amount of gas detected in blood vessels and the distribution in the different organs, each turtle was categorized as: (1) mild embolism—a small amount of gas in kidneys, mainly small hyperecogenic spots moving through renal vessels and/or in renal parenchyma, few comet tail artifacts; (2) moderate embolism—a larger amount of gas in kidneys, with clear aggregation of gas in some vessels areas and evident comet tail artifacts, and presence of gas in liver and cardiac chambers; (3) severe embolism—a large amount of gas present in tissues hampering the ultrasound examination. Additionally to the ultrasound, the green turtle was also evaluated through radiography as described in García-Párraga et al. (2014). Severity of the embolism was determined for the olive ridley on necropsy as the animal died onboard 2 h after surfacing.

RESULTS

Leatherback Sea Turtle

The CCLmin of this turtle was 136 cm, corresponding to a subadult (Stewart et al., 2007). The accidental capture occurred during the 2.5 h trawl at 37 m deep. Physical examination was conducted on board about 4 h after surfacing, revealing mild bleeding abrasions on front flippers and dorsal keels, but no other significant lesions or scars were found. The animal was in good body condition, with evident fat accumulation on the neck, pectoral, and inguinal fossa, and was well hydrated. No abnormal behavioral responses were observed during physical or neurological examinations. Prefemoral ultrasound examination revealed hyperechogenic spots and comet tail artifacts in renal vessels and portal-renal vein compatible with intravascular gas bubbles (**Figure 1B**) (Crespo-Picazo, 2019a). At the neck region, the dorso-cervical sinus did not show evidence of gas bubbles. No other internal structures were accessible due to the large size of the animal and limited penetration of the ultrasound. Based on the limited amount of intravascular gas present on accessible regions, it was presumed to be a mild GE case.

Green Turtle

The CCLmin of this turtle was 43 cm, corresponding to a juvenile (Limpus and Walter, 1980). The gillnet where the turtle was captured was set at 5 m deep for 12 h. Physical examination conducted 4 h after surfacing did not show any clinical signs or external lesions. Body condition was good. Ultrasound examination revealed circulating bubbles in the right atrium and renal adjacent vessels. The renal parenchyma also presented hyperechogenic spots with comet tail artifacts (**Figure 1D**) (Crespo-Picazo, 2019b). Conventional radiographs also revealed small amounts of gas in the kidney region. Based on these findings, it was classified as a mild GE case. The turtle remained in the rescue center facilities until complete rehabilitation and was released 6 weeks after admission.

Olive Ridley Turtle

The CCLmin of this turtle was 71.9 cm, corresponding to an adult (Zug et al., 1998). Trawl conditions when captured were 19 m deep for 4.5 h. Physical examination revealed several fresh superficial wounds caused by net entanglement, a ray sting embedded in the skin immediately lateral to the tail and a recent cloacal prolapse. Body condition was good. Hyperechogenic spots compatible with intravascular gas bubbles were evident in renal and neck vessels on the first ultrasound scan taken when the animal arrived on deck (**Figures 1F**, **2A**) (Crespo-Picazo and Parga, 2019). In a second scan, 30 min later, gas bubbles were even more clearly detectable in the renal parenchyma and vessels (**Figure 2B**). One hour later, the amount of gas was so severe that it hampered the renal ultrasound examination (**Figure 2C**). Based on the amount of gas present, the animal was classified as a severe GE case. GE severity was confirmed during necropsy as the turtle died 2 h after surfacing. Main findings included abundant gas inside cardiac chambers, senus venosus, mesenteric veins, kidneys, spleen, and most of the vasculature of all other internal organs (**Figure 3**). Macroscopic lesions associated with circulatory failure due to impediment of regular blood flow were also detected in several organs and tissues including the intestinal tract (segmental congestion of the intestinal mucosa), kidneys (marked congestion in extensive areas of renal parenchyma), and lungs (congestion and hemorrhage in lung parenchyma), confirming not just the presence of GE but also DCS diagnosis. Necropsy was performed within 12 h *post-mortem*, which it was also confirmed that the turtle was an adult female with well-developed and apparently functional oviducts.

DISCUSSION

Gas embolism and DCS was originally described in loggerhead sea turtles recovered as bycatch from trawlers and gillnets of local fisheries off the east coast of Spain (García-Párraga et al., 2014). Here, we demonstrate that green, leatherback, and olive ridley turtles are also susceptible to GE after fisheries interactions. Despite only having a single representative from each of these new species, we postulate that this provides evidence that all sea turtle species may, in fact, be susceptible to GE, especially when forcibly submerged for prolonged periods of time. This conclusion is further supported by recent findings that have showed the high prevalence of GE in loggerhead turtles in the southern Atlantic (Parga et al., 2020). Overall, we therefore think that GE may be a severely underestimated form of mortality associated with interactions between fisheries and sea turtles. If true, global estimates of sea turtle mortality associated with fisheries bycatch (e.g., Lewison et al., 2004) may be far higher than initially predicted.

Leatherbacks are the deepest diving sea turtle species, and therefore are the most adapted to deal with the associated effects of decompression after diving (Davenport et al., 2009; Fossette et al., 2010; Murphy et al., 2012). Yet, we still found evidence that this species is also affected by GE when forcibly submerged. Furthermore, this occurred when this individual was caught as a relatively shallow depth (37 m), which is shallower than the

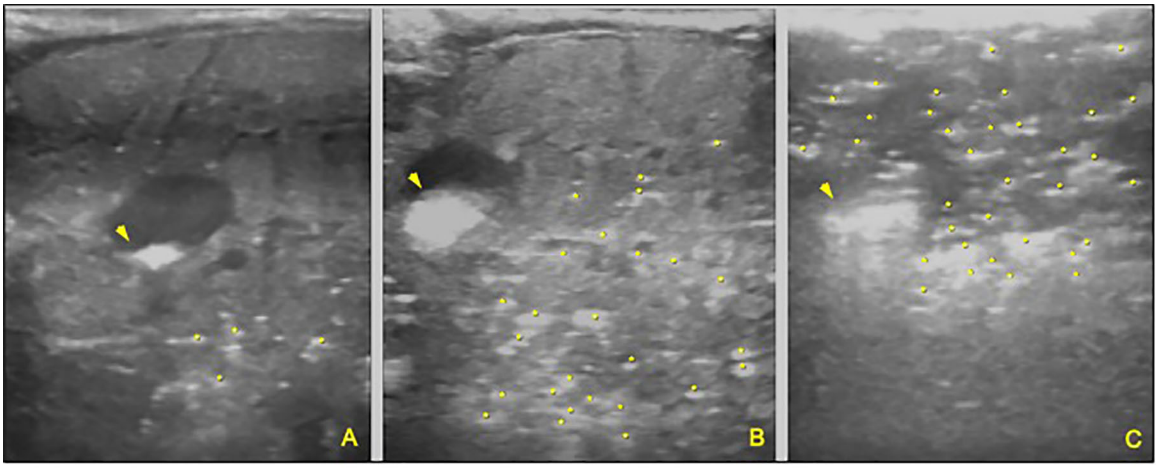

FIGURE 2 | Sequence of ultrasound images of the renal area of the olive ridley turtle, within 30–40 min intervals. A clear increase in the presence of gas in vessels and renal parenchyma is observed. **(A)** Mild gas embolism. Initial ultrasound after approximately 20 min of being surfaced. **(B)** Moderate gas embolism: The amount and distribution of gas increase in blood vessels and renal parenchyma. **(C)** Severe gas embolism. Massive amount of gas preventing ultrasound penetration and tissue visualization.

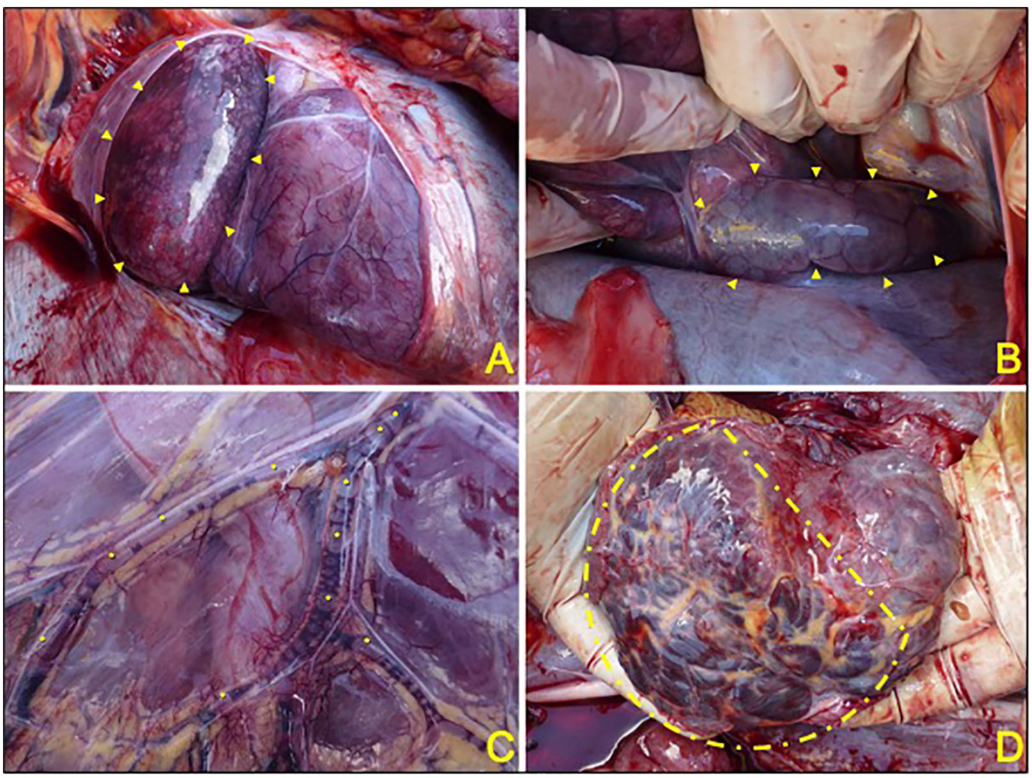

FIGURE 3 | Selected images from the olive ridley turtle necropsy affected with severe GE and DCS. **(A)** Opening of the pericardium, abundant gas bubbles are observed in the right atrium (arrowheads). **(B)** Large presence of gas in the venous sinus (arrowheads). **(C)** Presence of gas bubbles (asterisks) in mesenteric veins. **(D)** Marked congestion (dotted line area) of the renal parenchyma.

mean dive depth for this species (Robinson and Paladino, 2015). If leatherback turtles, a species that has been recorded dive to depths over 1200 m (Houghton et al., 2008), are affected by GE when retained at such shallow depths, it might suggest that all species of sea turtle may be vulnerable to GE. Alteration of the physiologic mechanism to minimize nitrogen absorption during dives (bradycardia, intracardiac shunting, pulmonary sphincters) is considered the main physio-pathological explanation driving

GE formation (García-Párraga et al., 2018). These anatomical, physiological, and behavioral mechanisms have been identified and developed to a different degree in several sea turtles species (Sapsford, 1978; García-Párraga et al., 2017). Although most work until now has been based on loggerheads as they are the most common bycaught species in the western Mediterranean, based on the present work, we cannot exclude all other species of sea turtles for being potentially affected from decompression under entanglement conditions at depth.

All three species described here appeared to suffer GE, but only the olive ridley was confirmed to have symptoms and lesions of DCS. As these symptoms and lesions were identical to those found in loggerhead turtles (García-Párraga et al., 2014), this provides further support that GE might be a common threat to all sea turtle species. Nevertheless, further research is still needed to confirm if this is the case and the real risk for the detected intravascular gas bubbles to lead to actual DCS and potential subsequent death in all other species. Perhaps some sea turtle species are able to manage any GE and thus inhibit the occurrence of any pathological events. Identifying more cases over time in different species and under different fisheries and environmental conditions will help clarify clinical implications and susceptibility to disease in the different species of sea turtles. Additionally, further investigations should also determine the role of concurrent and cumulative deleterious effects of other metabolic disorders observed in accidentally and direct capture of sea turtles (Harms et al., 2003; Innis et al., 2010, 2014; Phillips et al., 2015).

Conservation Implications

Mortality due to fishery interaction can be divided into direct and delayed mortality (Parga et al., 2017). The former is relatively easy to ascertain as it can be observed directly, given that it happens at the time of incidental capture or immediately after, with the animal still on board the fishing vessel. Delayed mortality, however, is far more difficult to assess, as it may occur within hours, days, or even months after the release of the animal. Mortality rates are commonly calculated based on two scenarios: (a) the total number of caught turtles found dead in fishery gear (all comatose turtles are assumed to survive) and (b) the total number of caught turtles found dead or comatose (assuming eventual death) in fishery gear (Casale et al., 2017).

Despite little data being currently available on post-release survivorship (Swimmer et al., 2006; Maxwell et al., 2018; Parga et al., 2020), the IUCN sea turtle specialist group recommends that live and moving sea turtles should be released from the boat and reintroduced into the sea (Oravetz, 1999). As these animals are active on release, they are therefore not typically accounted for when calculating bycatch mortality rates. Our finding as well as those of previous studies, however, suggest some those turtles may die within hours or days post-release (García-Párraga et al., 2014; Parga et al., 2020). Further knowledge on post-release survival is therefore essential to develop a more accurate measure of fisheries associated mortality in sea turtles.

It has been shown that the likelihood of fatal decompression in loggerhead sea turtles increases with gear depth; an average trawl depth of 65 m resulted in 50% estimated mortality under particular conditions (Fahlman et al., 2017). This finding is in agreement with previous global mortality bycatch estimates, which suggested that gears fixed to the bottom had higher mortality rates than shallower gear (Wallace et al., 2013), except if those gears are at depths out of the normal sea turtle diving range. As such, within the sea turtle diving range, reducing gear deployment depth (when possible) and soak time might be a mitigation measure worth taking into consideration and thus merits further investigation. On the other hand, by deploying gear at depths that exceed the typical diving ranges of sea turtles would also minimize the probability of sea turtle capture.

CONCLUSION

We found evidence that at least four sea turtle species are susceptible to GE and/or DCS because of bycatch in gillnets and trawlers at different geographical locations. As this condition can lead to reduce survivorship, there is a vital need for further studies to assess the prevalence of DCS in sea turtles for all fisheries and all sea turtle species worldwide. In turn, this knowledge will help both improve estimates of fisheries associated mortality in sea turtles and guide conservation management actions (Lewison et al., 2004). Research into effective mitigation measures and their implementation should continue with particular consideration of fishing gears types, deployment depths, soaking times, and locations that are more likely to result in bycaught sea turtles remaining forcibly submerged and developing GE and/or DCS.

DATA AVAILABILITY STATEMENT

All datasets generated for this study are included in the article/supplementary material.

ETHICS STATEMENT

Turtle health examinations were performed under the authorization of the "Valencia Ministry of Agriculture, Rural Development, Climate Emergency and Ecological Transition" to the Oceanogràfic Aquarium of the City of Arts and Sciences of Valencia for coordinating veterinary stranding response. The study in Ocean Atlantic waters was specifically carried out to assess the prevalence and immediate evolution of DCS in incidentally captured turtles under the permits 15962-6 and 15962-7 from the Brazilian government. Handling and sampling protocols were based on NMFS-SEFSC 2008 Sea Turtle Research Techniques Manual. All procedures were performed in accordance with relevant guidelines and regulations.

AUTHOR CONTRIBUTIONS

JC-P conceived the presented idea. JC-P, MP, and DM did the field work and collected the data. YB, VM-C, and DG-P verified the analytical methods and supervised the findings of this work.

JC-P and CL-B wrote the manuscript. All authors discussed the results and contributed to the final manuscript.

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Mitigation of Elasmobranch Bycatch in Trawlers: A Case Study in Indian Fisheries

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Bycatch poses a significant threat to marine megafauna, such as elasmobranchs. India has one of the highest elasmobranch landings globally, through both targeted catch and bycatch. As elasmobranchs contribute to food and livelihood security, there is a need for holistic approaches to bycatch mitigation. We adopt an interdisciplinary approach to critically assess a range of hypothetical measures for reducing elasmobranch capture in a trawler fishery on India's west coast, using a risk-based mitigation hierarchy framework. Data were collected through landing surveys, interviews and a literature review, to assess the following potential management options for their technical effectiveness and socio-economic feasibility: (1) spatio-temporal closures; (2) net restrictions; (3) bycatch reduction devices (BRDs); and (4) live onboard release. Our study provides the first evidence-based and nuanced understanding of elasmobranch bycatch management for this fishery, and suggestions for future conservation and research efforts. Onboard release may be viable for species like guitarfish, with moderate chances of survival, and was the favored option among interview respondents due to minimal impact on earnings. While closures, net restrictions and BRDs may reduce elasmobranch capture, implementation will be challenging under present circumstances due to the potentially high impact on fisher income. Interventions for live release can therefore be used as a step toward ameliorating bycatch, while initiating longer-term engagement with the fishing community. Participatory monitoring can help address critical knowledge gaps in elasmobranch ecology. Spatio-temporal closures and gear restriction measures may then be developed through a bottom-up approach in the long term. Overall, the framework facilitated a holistic assessment of bycatch management to guide decision-making. Scaling-up and integrating such case studies across different species, fisheries and sites would support the formulation of a meaningful management plan for elasmobranch fisheries in India.

Keywords: sharks, rays, scalloped hammerheads, guitarfish, mitigation hierarchy, bycatch, management, sustainability

INTRODUCTION

Fisheries constitute one of the biggest pressures on oceans today, due to their impact on marine habitats, overexploitation of fish stocks and bycatch of non-target species (Dayton et al., 1995; Myers et al., 1997; Davies et al., 2009). Bycatch threatens marine megafauna, fish and invertebrates through capture in non-selective fishing gear (Alverson et al., 1994; Hall et al., 2000). At least 20 million endangered, threatened and protected marine animals are estimated to be caught as bycatch annually (Pérez Roda et al., 2019). Traditionally discarded, bycatch is increasingly retained and sold due to dwindling catches of target species and rising demand for seafood products (Kelleher, 2005). As such, bycaught species contribute significantly to livelihood stability and food security in fishery-dependent developing nations like India (Lobo, 2007; Gupta et al., 2019). Given the socio-economic importance of bycatch and the vulnerability of many bycaught species, it is imperative to regulate and manage this complex dimension of fisheries.

Elasmobranchs (sharks and rays) are a highly threatened species group (Dulvy et al., 2014) with more than half of global fishing mortality attributed to bycatch (Stevens et al., 2000). Due to their slow growth, late maturity and low fecundity, elasmobranchs are highly susceptible to fishing pressure, with a limited capacity to recover from overexploitation (Bonfil, 1994). Elasmobranchs play important roles in marine ecosystems as top and meso-predators, and provide socio-economic value to coastal communities through fisheries and tourism, making their conservation a top priority (Ferretti et al., 2010; Gallagher and Hammerschlag, 2011; Dent and Clarke, 2015).

India is among the top three elasmobranch fishing nations in the world (Dent and Clarke, 2015). While artisanal fishers in India have practiced targeted shark fishing since at least the early 1900s (Fernando et al., 2017), the advent of mechanized fishing for shrimp and high value finfish has led to increases in total elasmobranch capture in bycatch (Kizhakudan et al., 2015). Though many elasmobranchs landed in India today are caught as bycatch (Kizhakudan et al., 2015), they are seldom discarded as their meat forms a cheap and widely consumed protein source (Dulvy et al., 2017; Jabado et al., 2018). Therefore, domestic elasmobranch meat consumption may be a major driver of their fishing pressure in India (Karnad et al., 2019). Although we use the term bycatch here, we emphasize that these species are retained, and have some commercial and socio-economic importance.

Landings of elasmobranchs have declined in India in recent decades, from 75,262 tons in 1998 (CMFRI, 1999) to 42,117 tons in 2018 (CMFRI, 2019). This reduction is despite increasing fishing effort, which suggests that elasmobranch populations are overexploited (Kizhakudan et al., 2015), and corresponds with global trends (Davidson et al., 2016). With over half the elasmobranch species in the Arabian Sea region assessed as threatened (Jabado et al., 2018), there is an urgent need for improved management of fisheries that impact these species. While India has imposed a ban on shark fin trade and protected ten species under the Wildlife (Protection) Act, 1972 (Kizhakudan et al., 2015), these regulations are hampered by limited capacity for monitoring and enforcement. Furthermore, with incidental catch being a major issue, such regulations likely have limited success in reducing fishing mortality. They need to be accompanied by practical measures to reduce capture of priority species at the fishery level (Booth et al., 2019a).

The complex issue of elasmobranch bycatch leads to trade-offs between elasmobranch conservation and livelihoods of fishers (Booth et al., 2019a). Trawlers, in particular, have high levels of elasmobranch catch (Kizhakudan et al., 2015). Trawling in India is increasingly driven by exports of shrimp and other high value species, as well as high demand for fishmeal (Bhathal, 2005; Gupta et al., 2019). This is producing a biomass-based fishery with trawlers frequently fishing in shallow inshore waters with small mesh sizes to catch large volumes of fish (Kumar and Deepthi, 2006). Coastal elasmobranchs are collateral damage in this complex, multispecies trawl fishery, and form a small percentage of the total catch (Kizhakudan et al., 2015). However, conservation measures for elasmobranchs are likely to impact catches of high-value species, and hence reduce earnings of fishers. Given that there are 3.8 million active fishers in India (Department of Fisheries, Ministry of Fisheries, Animal Husbandry and Dairying, 2019); it is critical to develop shark management strategies that are science-based, economically viable and socially just.

The mitigation hierarchy is a framework for preventing and compensating for the negative impacts of development projects on biodiversity (BBOP, 2012). It has recently been proposed as a framework for mitigation of fisheries bycatch (Milner-Gulland et al., 2018), and follows four sequential steps: (1) avoidance of bycatch, e.g., through fisheries closures, (2) minimization of fisheries impacts, e.g., through gear modifications, (3) remediation of bycaught species, e.g., through live release protocols, and (4) offsetting of the residual impact through conservation measures elsewhere (Squires et al., 2018). The framework assembles a range of mitigation measures under each step, and assesses their effectiveness in meeting a quantitative bycatch reduction target (Milner-Gulland et al., 2018). It aims to balance conservation with economic development, by facilitating the sustainable use of natural resources with minimal or no net loss of biodiversity (Arlidge et al., 2018). Booth et al. (2019b) expanded and adapted the mitigation hierarchy for shark fisheries management. They provide a risk-based framework which integrates biological and operational aspects of species and fisheries with socio-economic context to manage potential trade-offs between conservation objectives and human needs. Set within the overarching framework of the mitigation hierarchy, this approach can be applied to develop holistic, context-specific and adaptive measures for shark fisheries management.

We used the mitigation hierarchy for sharks (Booth et al., 2019b) to assess options for shark and ray catch mitigation in an Indian trawl fishery. Our study was conducted at Malvan, a fishing town with a coastal, mixed species fishery, making this the first practical application of such an approach to managing elasmobranch bycatch for a data-limited, fisheries-dependent site. Our specific aims were to: (1) evaluate reduction measures for elasmobranch bycatch in the Malvan trawl fishery using the mitigation hierarchy framework, and (2) assess the applicability of this framework for bycatch reduction in a multi-species fishery with a complex socio-economic context.

Following the process outlined in Booth et al. (2019b) we first present an overview of elasmobranch fisheries at the study site and risk factors to the study species. We then propose different options for bycatch mitigation and assess them in terms of their technical effectiveness and socio-economic feasibility. Finally, we discuss the outcomes of the framework and its applicability as a decision-making tool for bycatch management, and propose recommendations for interventions and further research. We do not intend for this to be a complete assessment of management options; rather, we aim to initiate structured and interdisciplinary thinking for elasmobranch bycatch mitigation in India, identify data gaps and highlight potential management solutions going forward.

MATERIALS AND METHODS

Study Site and Fishery

Malvan is a region on the west coast of India (16.052027°N, 73.468247°E; **Figure 1**). Its coastline is interspersed with a range of marine habitats including estuaries, mangrove forests, coral outcrops and a shallow shelf ranging from about 20 to 30 m in depth to about 20 km offshore (UNDP, 2013). This shallow shelf forms a habitat for many marine species, as well as highly productive fishing grounds. Malvan's waters also host the Malvan Marine Sanctuary, one of India's marine protected areas (Sundaramoorthy et al., 2001). However, while the Marine Sanctuary has been designated in 1987, it is not functional as it is yet to be implemented on ground (UNDP, 2013).

There are 22 fishing villages in the greater Malvan region (i.e., the Malvan *Taluka*), with 10,635 resident fishers as well as a significant population of migrant fishers (CMFRI, 2012). This region is home to diverse fisheries, with 80–100 trawlers, at least 600 gillnet boats, and some artisanal fisheries, including those using shore seines. Our study was based at the main town in this region, also known as Malvan. All fishing boats, owned by fishers from different villages throughout the Malvan region, land and sell their catch at this site. Trawlers in Malvan constitute a multi-species fishery that target a range of species: prawn, crabs, and demersal fish using a benthic net (i.e., bottom trawl net) or pomfret (*Pampus* sp.), mackerel (*Rastrelliger kanagurta*) and other pelagic fish using a pelagic net (i.e., mid-water trawl net). Trawl fishing takes place between August to May, with a mandatory seasonal ban imposed by the government during June and July to protect spawning fish (Narayanakumar et al., 2017). Trawlers operate across the region, from Panaji in Goa State in the south to Ratnagiri in the north (approx. 180km; **Figure 1**). They are relatively small-sized (100–140 HP, 40–55 feet vessel length), fishing nearshore within a depth of 100m. Fishing trips typically last 1–5 days. Elasmobranchs are frequently captured as non-target or secondary catch, particularly in trawlers, but also in gillnets and artisanal fisheries. Most elasmobranch catch is retained and sold for meat, which is salted, dried and consumed within the region. Elasmobranchs are generally considered low-value products, with sharks relatively more profitable than rays.

Process

We adopted the risk-based mitigation hierarchy for elasmobranchs developed by Booth et al. (2019b) for this study (**Table 1**). The process involves understanding the fishery and assessing the risk to the species of concern; developing mitigation measures for incidental catch under the 4 mitigation hierarchy steps; and assessment of these measures in terms of technical and socio-economic feasibility. We used a mixed-methods approach to collect data for this process (section "Data Collection"), which were analyzed and assessed to populate the framework and identify management measures for elasmobranchs (section "Analysis and Assessment").

A combination of landing site surveys and interviews were used to collect primary data for the framework. In addition, secondary data were used in the assessment of mitigation measures where no primary data were available (explained in section "Technical Assessment").

Data Collection

Trawler Landing Surveys

To understand the biological and operational aspects of the fishery, elasmobranch landings from trawlers were sampled over two seasons: March–May 2018 and October 2018–May 2019. Sampling was conducted at the Malvan landing center 3 days per week on alternate days, starting on different days to avoid any bias in sampling the same 3 days. Every boat that had landed any elasmobranchs was sampled. Biological data on the elasmobranchs (species, abundance, size, and sex) were recorded, and operational data on the fishing trip (effort, fishing location, depth, and gear) were collected through informal interviews with the fishers. Some captured elasmobranchs, particularly juveniles, are discarded at sea. We were not able to estimate these discards, and our data are therefore restricted and potentially biased to landed elasmobranchs only. We also acknowledge that our sampling was conducted over a relatively short time period; however, based on informal discussions with key informants, we believe it to be representative of the present fishery scenario in Malvan.

Interviews

To supplement the biological data and understand the socio-economic context of the fishery, interviews were conducted with fishing community stakeholders (fishers and trawler owners). Owners (n = 11) were selected at the landing center through convenience sampling, and represented about 20–25% of the trawlers in Malvan. Fishers (n = 7, two of whom were also boat owners) were selected through purposive sampling, as we intended to interview key informants with in-depth knowledge about elasmobranchs. Although our sample size of fishers was small, we believe it was sufficient as saturation was reached in terms of the information provided.

We used a semi-structured questionnaire in our interviews (Milner-Gulland and Rowcliffe, 2007), with the following sections: (1) background information, fishing experience and behavior; (2) catches and trends in sharks and rays (fishers only); (3) costs and revenues for an average fishing trip (owners only); and (4) opinions of and preferences for the proposed

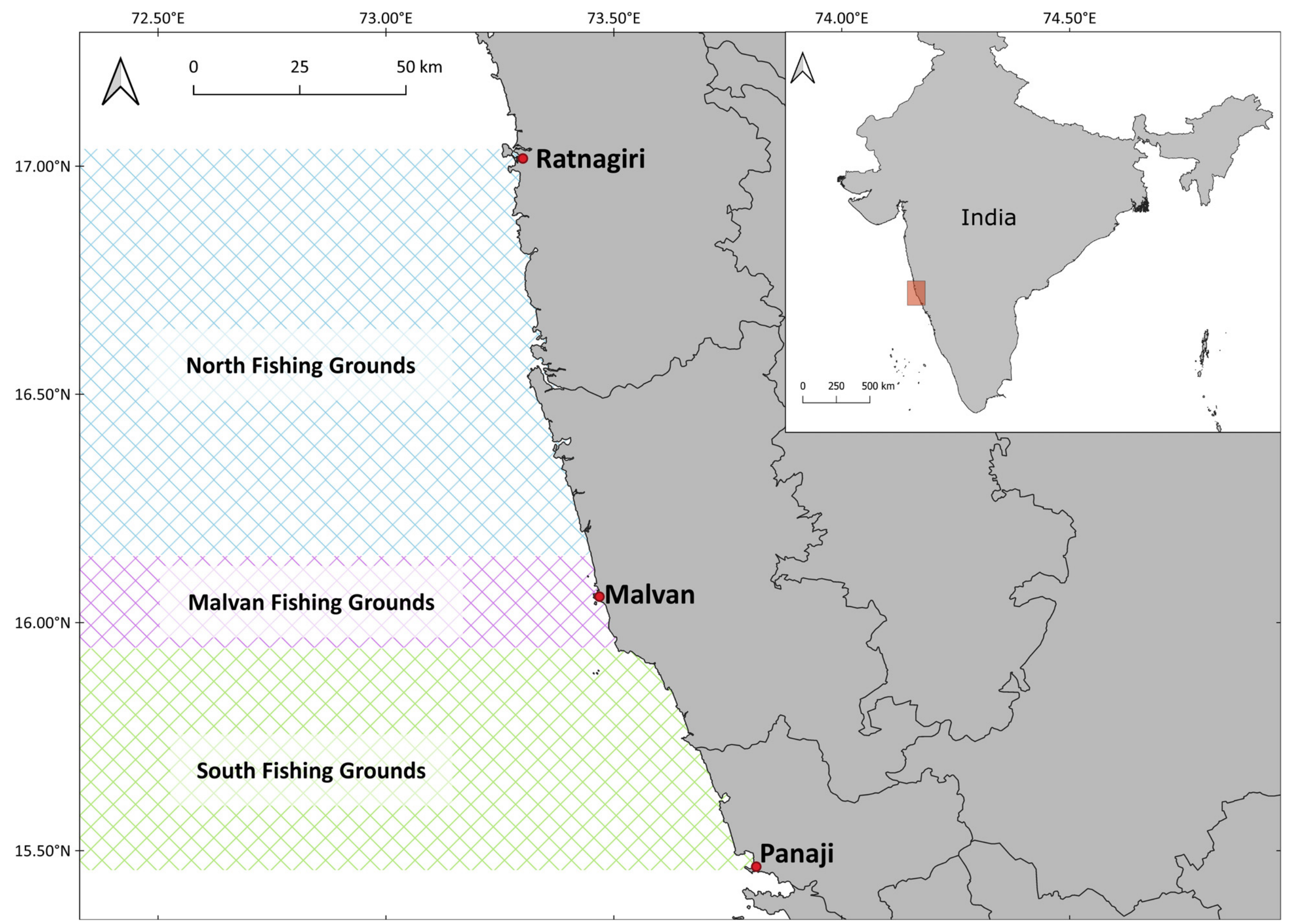

FIGURE 1 | The study site and spatial extent of the fishery, showing the different fishing grounds categorized by this study (north, Malvan, and south). The fishing grounds delimited here represent the latitudinal extend of the fishing areas only, the distances traveled offshore are unknown. Black lines in the map represent the borders of each district, while the red dots represent the main fishing towns in this region.

mitigation measures for elasmobranch catch (**Supplementary Table 1** and **Supplementary Figure 1**). Respondents were asked to quantify their opinions about the impact of each mitigation measure on elasmobranchs, on their target species, and on their profits using a Likert scale (Likert, 1932). The scale ranged from −2: More than a 50% decrease, −1: Up to a 50% decrease, 0: No impact, +1: Up to a 50% increase, +2: More than a 50% increase. Interpretation of the Likert scale responses was carried out with some caution to avoid any potential bias arising from the respondents' understanding of the scale.

Analysis and Assessment

Defining the Study Species

Multiple elasmobranch species are caught and landed by trawlers in Malvan. Given the lack of species-specific data and their similar economic values, this study first considers elasmobranchs in two broad groups of sharks and rays to assess management measures, particularly from an economic and feasibility perspective. However, elasmobranchs are highly diverse in their biological and ecological characteristics, and the same strategy will not fit all species (Dulvy et al., 2017). Therefore, we also focus on a few priority taxa (IUCN, 2019); scalloped hammerhead sharks (*Sphyrna lewini*), sharpnose guitarfish (*Glaucostegus granulatus*), and widenose guitarfish (*Glaucostegus obtusus*). These were chosen due to their threatened status and vulnerability to fishing. *S. lewini* is a long-lived species with late maturity, slow growth (Miller et al., 2013; Zacharia et al., 2017) and low intrinsic potential to recover from fishing pressure (Smith et al., 1998). It has a higher risk of capture in fishing gear due to the unique shape of its head and the aggregating behavior of juveniles in nearshore waters (Gallagher et al., 2014). *S. lewini* is listed as Critically Endangered by the International Union for Conservation of Nature (IUCN; Rigby et al., 2019a), and previous studies have noted an apparent decline of this species in Malvan (Karnad et al., 2019). Giant guitarfish (Glaucostegidae) have relatively high population productivities with moderate recovery potential if fishing mortality is kept low (D'Alberto et al., 2019). However, due to the high levels of exploitation throughout their range, both study species (*G. granulatus* and *G. obtusus*)

TABLE 1 | The risk-based mitigation hierarchy framework for elasmobranchs adapted from Booth et al. (2019b).

Framework step	Method used
A. Define the Problem	
Define the species of concern Understand the fishery and identify species of concern for assessment	Preliminary analysis of landings data. **Section "Defining the Study Species"**
Risk assessment Technical factors of the fishery affecting capture and mortality of the species of concern	Modeling catches from the landings data on fishery variables. **Section "Risk Assessment – Factors Affecting Elasmobranch Capture"**
Set a goal and target The desired catch reduction goal and quantitative reduction target	Preliminary analysis of landings data. **Section "Reduction Target"**
B. Management Measures Under Each Step of the Mitigation Hierarchy	
Avoidance Spatio-temporal closures of pupping grounds	
Minimization Restriction of benthic nets during the pupping season	
Remediation Bycatch Reduction Devices (BRDs)	Management options were selected based on preliminary analysis of landings data. **Section "Developing Mitigation Measures"**
Remediation Onboard release of live individuals	
Offset Mitigation in other fisheries	
C. Assessment of Management Options	
Technical assessment To what degree can each measure reduce mortality risk of the species of concern, based on biophysical and technical factors?	Secondary data + model of factors influencing catch + interview data. **Section "Technical Assessment"**
Feasibility assessment To what degree can each measure be feasibly implemented in the fishery, given costs, benefits & social context?	Interview data + Secondary data. **Section "Feasibility Assessment"**
Management recommendations Which management measures and instrument mix are likely to have the greatest impact?	Collation of results. **Table 4**

Method used for each step, and the corresponding section of the paper are given. Graphics courtesy of The Noun Project (2014), under the creative commons license.

have recently been listed as Critically Endangered (Compagno and Marshall, 2019; Kyne and Jabado, 2019), with a need for urgent global action for their conservation and recovery (Kyne et al., 2019).

Due to small sample sizes and their similar biology, we grouped the two guitarfish species for analysis (hereafter guitarfish). We use the term "hammerhead" to refer to the scalloped hammerhead shark, "sharks" to all shark species (i.e., Selachimorpha – including scalloped hammerhead sharks) surveyed at the study site, and "rays" refers to all ray species (i.e., Batoidea – including guitarfish).

Risk Assessment – Factors Affecting Elasmobranch Capture

We first evaluated factors affecting the number of sharks and rays captured in trawlers, by modelling catches from the landings data against a number of operational fishing variables and their interactions (**Table 2**; see **Figure 1** for the locations of the fishing grounds). We used a lognormal linear mixed model (package: lmerTest, Kuznetsova et al., 2017), where the response variable was log-transformed to meet model assumptions. The best fitting model was selected using AIC model selection. Following this, differences in catches associated with the explanatory variables

TABLE 2 | The different fishing factors and interactions used as explanatory variables in the models.

Explanatory Variable	Description	Expected relationship with response variables			
		Number of sharks caught	Number of rays caught	Probability of hammerhead capture	Probability of guitarfish capture
Fishing effort	Number of fishing days (Continuous variable)	Higher with more effort			
Fishing season	**Pre-Monsoon**: January–May **Post-Monsoon**: September–December	Higher in post-monsoon			
Depth	**Shallow** if ≤25 m **Deep** if >25 m	Higher in shallow waters	Higher in shallow waters	Higher in deep waters	Higher in shallow waters
Gear	**Benthic** or **Pelagic** nets	Higher in pelagic nets	Higher in benthic nets	Higher in pelagic nets	Higher in benthic nets
Fishing Location	**Malvan:** within Malvan waters **North:** north of Malvan **South:** south of Malvan	N/A			
Location × Gear Location × Season Season × Gear Season × Depth	Interaction terms	N/A			
Boat ID	Random effect	N/A			

Four different models were fitted, with the following response variables: number of sharks captured (log-transformed), number of rays captured (log-transformed), presence of hammerhead sharks (1 = yes, 0 = no) and presence of guitarfish (1 = yes, 0 = no). Expected relationship of each explanatory variable with the response variable of each model is stated.

retained in this best fitting model were assessed, using *t*-tests within the lmerTest program. For the categorical explanatory variables, the coefficients and *p*-values were calculated with respect to a reference category – for location: south, for gear: benthic, for depth: shallow and for season: pre-monsoon. For example, *p*-values for the north and Malvan fishing grounds presented in the results are each in comparison to the south fishing grounds. These models were separately constructed for sharks and rays.

We then evaluated factors influencing capture of hammerheads and guitarfish. We created a binary response variable for whether these species had been captured in each of the fishing trips sampled (1 = yes, 0 = no). As we had only sampled trawlers that had captured elasmobranchs, this gave us the probability of capturing a hammerhead or a guitarfish, given that an elasmobranch had been captured. We fitted a generalized linear mixed model (GLMM; package: lme4, Bates et al., 2015) with a binomial logit distribution to this binary response variable, with AIC model selection. The same explanatory variables as before were used (**Table 2**), and coefficients and *p*-values calculated similarly. The models were constructed separately for hammerheads and guitarfish. All analyses were conducted in RStudio version 1.1.463 (R Core Team, 2014; RStudio Team, 2015).

Reduction Target

The mitigation hierarchy calls for defining a goal in terms of a desired change in biodiversity, accompanied by a quantitative catch reduction target against which the mitigation measures can be evaluated. The target can be defined using a metric such as population growth rate or Potential Biological Removal (PBR) threshold (Milner-Gulland et al., 2018).

We defined the overarching conservation goal for this study as the minimization of incidental catch of the study species within the socio-economic constraints of the study site. However, we were unable to specify a quantitative reduction target due to the data-limited nature of the site and species. We instead set a relative target, which is a reduction in the number of animals caught of the study species as compared to current catch rates, which is more realistic in the present scenario.

Developing Mitigation Measures

A number of potential bycatch reduction measures can be categorized under each step of the framework (Milner-Gulland et al., 2018; Booth et al., 2019b). Using a preliminary analysis of the landings data, as well as an understanding of the logistics and socio-economic context of the study site, we proposed the following potential management measures for assessment:

Avoidance: spatio-temporal closures of pupping or nursery grounds

Based on pilot surveys, the southern fishing ground (**Figure 1**) was identified as a possible nursery ground with high catches of juvenile sharks and rays in the post-monsoon season. We therefore proposed a closure of these grounds for 2 months from October to November, which may be the pupping season (when elasmobranchs give birth to their young) for many species. However, due to lack of data we did not define the exact spatial extent of the closure area.

Minimization: restriction of benthic net use during the pupping season

Pilot interviews and landing site surveys found a higher catch of juvenile elasmobranchs in benthic nets as compared to pelagic nets. We proposed a restriction in the use of benthic nets for trawlers during the same time period (October–November) to minimize elasmobranch catch.

Remediation: bycatch reduction devices (BRDs)

Based on studies elsewhere, and trials being undertaken in India, we proposed the use of BRDs such as turtle excluder devices (TEDs) and other similar designs with escape panels as the third option to reduce mortality of elasmobranchs as a result of bycatch.

Remediation: onboard release of live individuals

We proposed the safe handling and release of all captured elasmobranchs, if alive, onboard the trawler to reduce mortality.

Offset: mitigating elasmobranch catch offsite

We proposed to mitigate elasmobranch mortality through improving management measures in other fisheries in the region that are likely to target the same populations as Malvan.

Technical Assessment

The hypothetical effectiveness of the proposed mitigation measures was assessed through a combination of primary and secondary data. For avoidance and minimization, model coefficient values from 2.4.2 were used to evaluate whether changes in particular operational fishing variables would have an impact on elasmobranch catches. This was supplemented with fisher perceptions of the impact of the proposed measures on the populations of elasmobranchs, and of their target fish species, using the Likert scale described in section "Interviews."

Due to a lack of data on the effectiveness of BRDs and live release for the study species in Malvan trawlers, we assessed the hypothetical impacts of these two measures using secondary data from previous studies in tropical trawl fisheries. We used Google Scholar to search for studies that have assessed the impact of BRDs and live release on bycatch rates and survival of the study species. Search terms included "bycatch reduction device," "brd," "turtle excluder device," "ted," "fishing mortality," "post-capture survival" or "release" combined with "elasmobranch," "shark," "ray," "hammerhead," or "guitarfish." We found a total of 10 relevant studies to help infer the effectiveness of these measures.

Feasibility Assessment

The feasibility of each mitigation measure was assessed using perceptions of boat owners, who quantified the potential impact of each measure on their income using a Likert scale. All respondents (fishers and boat owners) were also asked for their overall opinion of each mitigation measure, and to select their most preferred option. Feasibility of the measures and compliance of the fishing community were also discussed. The qualitative responses obtained for these sections were noted and used to understand why each measure would or would not work.

Permits and Ethics

No permits were required for landing site surveys. Ethics clearance for the interviews was obtained through an institutional ethic committee review (Reference number: DF_Ethics committee_HS_2019_May_01). The interviews were conducted and voice recorded only after obtaining informed verbal consent from the participants, and assuring them that they could omit questions or end the interview at any stage. All interview data were kept confidential and anonymous.

RESULTS

Risk Assessment

We sampled a total of 985 fishing trips over the two sampling seasons. November and December were the peak months for shark capture (**Table 3**). Hammerhead sharks were captured only between November and January, and all recorded individuals were juveniles. Catch rates of rays were more consistent throughout the year, with November being the peak month. The two guitarfish species were sporadically captured in low numbers throughout the year (**Table 3**). In general, most of the sharks and rays captured in trawlers were <1 m in size (**Table 3**), as they were composed of small-sized coastal species and juveniles of larger species like hammerheads. Adults of large ray species were infrequently captured, whereas those of larger sharks were never captured by trawlers.

On modeling factors affecting elasmobranch capture, the number of sharks captured was found to be strongly influenced by season, with the post-monsoon having significantly higher catches ($p < 0.001$). To disentangle the effect of season from the effects of the other fishing variables shark catches were separately modeled for each season. For the post-monsoon season, fishing location was the only significant variable, with the south fishing grounds having higher captures of sharks as compared to Malvan ($p < 0.001$) and the north grounds ($p = 0.01$). For the pre-monsoon season several variables were found to be significant. Pelagic nets had higher shark catches than benthic nets ($p < 0.001$), and higher fishing effort was linked to higher catches ($p < 0.001$). Deep waters had slightly higher shark captures than shallow waters ($p = 0.03$; **Figure 2** and **Supplementary Table 2**).

Hammerhead sharks also had a significantly higher probability of capture in the post-monsoon season ($p < 0.001$), in pelagic nets ($p < 0.001$) and in deep waters ($p = 0.009$). Contrary to the trends for pooled shark species, the probability of catching hammerheads was slightly higher in the northern fishing grounds ($p = 0.02$ with reference to the south; **Figure 2** and **Supplementary Table 2**). For rays, the numbers captured were not significantly related to season. Benthic nets had significantly higher catches of rays than pelagic nets ($p = 0.003$), as expected. Like sharks, the southern fishing grounds had significantly higher captures of rays than the northern grounds ($p = 0.01$, **Figure 2** and **Supplementary Table 2**).

No interaction terms were included in any of the best fit models. The full set of models and coefficients are presented in the **Supplementary Tables 2, 3**. Due to the small numbers of

TABLE 3 | Summary of the elasmobranch landings data, given for sharks, rays, hammerheads and guitarfish.

		Sharks	Rays	Hammerheads	Guitarfish
Total sampled		6380	3788	80	17
Size range of caught individuals (cm)		10.5–89.5	10.5–148	42–64	29–148
Overall CPUE (catch/trip)		6.4 ± 12.8	3.8 ± 7.5	0.08 ± 0.5	0.02 ± 0.1
CPUE per Season	Pre-monsoon:	4.6 ± 11.6	4.0 ± 8.0	0.01 ± 0.1	0.02 ± 0.1
	Post-monsoon:	11.1 ± 14.6	4.0 ± 7.7	0.30 ± 0.9	0.02 ± 0.2
CPUE per Gear	Benthic:	3.5 ± 7.4	4.7 ± 8.4	0.01 ± 0.1	0.03 ± 0.2
	Pelagic:	13.9 ± 18.7	2.3 ± 6.4	0.30 ± 0.9	0.01 ± 0.1
CPUE per Depth	Shallow:	5.9 ± 10.7	4.3 ± 8.6	0.01 ± 0.1	0.02 ± 0.2
	Deep:	5.4 ± 11.9	3.8 ± 8.0	0.10 ± 0.7	0.02 ± 0.2
CPUE per Location	South:	9.9 ± 15.5	4.1 ± 8.8	0.10 ± 0.6	0.02 ± 0.1
	Malvan:	4.1 ± 8.4	4.4 ± 7.1	0.05 ± 0.3	0.03 ± 0.2
	North:	4.3 ± 9.2	3.5 ± 6.4	0.10 ± 0.6	0.02 ± 0.2

Total length (TL) is the size measurement used for sharks and guitarfish, whereas disc width (DW) is used for rays. Catch per unit effort (CPUE) is calculated as the number of individuals caught per fishing trip for that taxa. An overall average CPUE for each taxa over the entire sampling duration is given, as well as for each fishing season, gear, depth, and location. Standard deviation is given along with the average CPUE values (i.e., CPUE ± standard deviation).

guitarfish encountered over the sampling period ($n = 17$), we were unable to model fishing variables affecting their capture.

In order to deduce potential population trends in the study species, fishers and boat owners were asked about changes (if any) in elasmobranch catches over the past decade. All respondents stated that catches of all species, including elasmobranchs, had significantly reduced over the past 10 years. Most respondents (11 of 16) suggested that poor fishing practices like purse seining, light-emitting diode (LED) fishing (an illegal fishing technique where mechanized vessels use strong LED lights to attract and capture large volumes of fish) and high-speed trawling were the primary reasons for this decline (**Supplementary Table 4**). Other reasons suggested were overfishing (i.e., high fishing effort) and environmental factors (e.g., climate change). Most fishers were aware of the impacts of their fishing practices on fish populations:

> "Because of overfishing and constant killing, the fish have reduced. If there are 50 fish that have been produced and we kill 40–50 of them, then how are they supposed to replenish?"
>
> – A fisher, age 52.

Technical Assessment of Mitigation Measures

The technical assessment focused on the potential impact of the proposed measures on elasmobranch populations, if fully implemented (**Table 4**).

Avoidance: Spatio-temporal closure of the southern fishing ground for 2 months during the pupping season.

Model coefficients indicate that this measure is likely to have a significant positive impact for shark populations, due to the higher captures of sharks in the southern fishing grounds and post-monsoon months for which the closure is proposed. This holds true for rays as well, as the southern fishing grounds were related to higher captures (**Figure 2**). Although the likelihood of catching a scalloped hammerhead was significantly higher in the post-monsoon season, the southern fishing grounds (where the closure was proposed) had a lower likelihood of catching this species (**Figure 2**). Therefore, the impact of a spatio-temporal closure of the southern grounds on hammerhead catch in the post-monsoon season is somewhat uncertain.

Most fishers perceived that there would be a positive impact of spatio-temporal closures on both elasmobranch populations and populations of their target species (**Figure 3**). A summary of the fisher responses to all the proposed mitigation measures can be found in **Supplementary Table 4**.

Minimization: Restriction of benthic net use for 2 months during the pupping season.

Gear was not included in the best fit model during the post-monsoon for sharks (**Figure 2**). This may be because, while shark catches are higher in pelagic nets on the whole, juveniles of most species are largely caught in benthic nets in the post-monsoon season. Hence, the variation due to size and age may be affecting this result, and benthic net restriction during this period may in fact be effective in reducing juvenile shark capture. This measure is likely to have a positive impact for rays due to the significantly higher captures in benthic nets. However, the opposite relationship was found for hammerheads, indicating that this measure may increase hammerhead capture due to a potential switch to higher use of pelagic nets (**Figure 2**).

Most fishers believed that this measure could be beneficial for populations of elasmobranchs and of their target species (**Figure 3**).

Remediation: Bycatch Reduction Devices (BRDs).

This measure was assessed through a literature review of BRDs tested for elasmobranchs in other tropical trawl fisheries, most of these being modifications of TEDs (**Table 5**). A wide range of impacts on shark catches was observed across the studies, from 4.9 to 94% reduction (**Table 5**). Bycatch reduction rates for small sharks were on the lower side. A reduction of 25–59% of bycatch of small rays was observed from tropical prawn trawl fisheries in South America (**Table 5**). For hammerheads, TEDs achieved a reduction rate of 55%, whereas another study found a reduction of 31% in bycatch of the closely related bonnethead sharks (**Table 5**). Courtney et al. (2007) found mixed and limited effectiveness of TEDs for bycatch reduction of guitarfish and wedgefish.

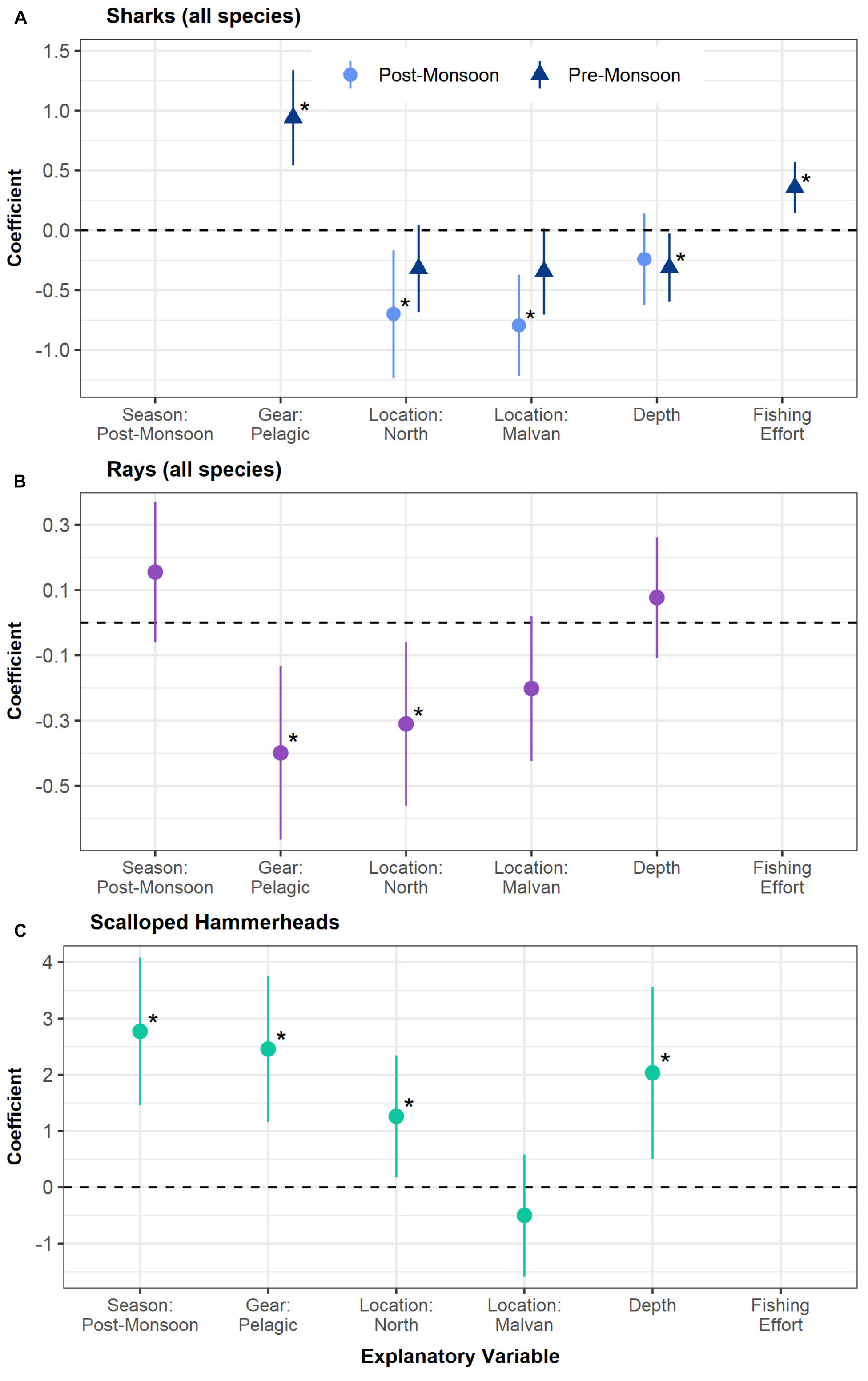

FIGURE 2 | Coefficients and confidence intervals of the best fit mixed models of the number of sharks captured **(A)**, number of rays captured **(B)**, and probability of hammerhead shark capture **(C)** plotted against various fishing variables. Panels **(A,B)** are lognormal models, whereas panel **(C)** is a binomial model. The coefficients of the categorical variables are given with respect to a reference category – for Location: South, for Gear: Benthic, for Season: Pre-monsoon, for Depth: Shallow. A positive coefficient for a category (for example, pelagic gear in panel **(A)**) indicates a higher catch of that taxa as compared to the reference category (i.e., benthic gear). Similarly, positive coefficients of a continuous variable such as fishing effort indicate a higher catch at greater effort. Significant variables are indicated with an Asterix (*). Shark catches were modeled separately for the pre- and post-monsoon seasons.

TABLE 4 | Assessment of the mitigation measures and recommendations for all study species.

Mitigation measure	Technical assessment				Feasibility assessment	Recommendations
	Sharks	Rays	Hammerheads	Guitarfish		
Avoidance: Spatio-temporal closures	✓✓✓ May significantly reduce capture	✓✓✓ May significantly reduce capture	? Effect uncertain. Lack of data on habitat use	? Effect uncertain. Small sample size and lack of data	✓ May cause high economic losses, enforcement may be a challenge	a. Live release programmes for guitarfish, implemented through awareness and landings regulations b. Participatory monitoring for research on habitat use, post-capture survival rates and long-term fisheries c. Closures and gear regulations developed bottom-up as long-term solutions
Minimization: Restriction of benthic net	✓✓ May reduce capture of juveniles.	✓✓✓ May significantly reduce capture	✓ May increase capture due to greater use of pelagic nets	? Effect uncertain. Small sample size and lack of data	✓ May cause high economic losses, enforcement may be a challenge	
Remediation: BRDs	✓✓ TEDs may be moderately effective	✓✓ TEDs may be moderately effective	✓✓ TEDs may be moderately effective	✓✓ TEDs may be moderately effective	? May cause loss of target catch and valuable bycatch. Positive feedback from respondents. High uncertainty.	
Remediation: Onboard release	✓✓ Some species may have high survival rates (e.g., bamboo sharks)	✓✓ Some species may have high survival rates (e.g., guitarfish)	✓ Low post-capture survival rate	✓✓ Post-capture survival may be high	✓✓✓ Small economic losses. Most preferred option by the respondents	

A semi-quantitative traffic-light categorization is used to summarize the assessment (following Booth et al., 2019b) based on collation of the results and expert opinion. Offset measures were deemed unfeasible and hence have not been included in this assessment. Green color and 3 "✓" symbols indicate a high perceived effectiveness for that field, whereas red color and 1 "✓" symbol indicates a low effectiveness. Gray color with a "?" symbol indicates uncertainty.

Locally designed TEDs have been developed for sea turtles caught in Indian trawl fisheries by the Central Institute of Fisheries Technology (CIFT) and are undergoing testing and improvement. However, their effect on elasmobranchs has not been specifically assessed, with one case study mentioning a lack of exclusion achieved (**Table 5**).

As fishers were not aware of bycatch reduction devices and techniques, they were not able to estimate their impact on elasmobranch and target fish populations (**Figure 3**).

Remediation: Onboard release of live individuals.

This measure was also assessed through a literature review. Survival rates upon capture and post-release is species-specific, hence sharks and rays as collective groups could not be assessed, and we focused on hammerheads and guitarfish. Hammerheads had high mortality rates of up to 98% upon capture in trawlers in South Africa and Northwest Africa (Fennessy, 1994; Zeeberg et al., 2006). Furthermore, hammerheads captured in the present study are all juveniles (<70 cm TL) and are likely to have very high mortality rates upon capture, with little scope for live release. Although we could not find any literature on the post-capture survival of the guitarfish species under study, related species like *Rhinobatos* sp. and *Rhynchobatus djiddensis* were found to have low to moderate mortality rates of 10–53% upon capture by trawlers (Fennessy, 1994; Stobutzki et al., 2002).

Fishers indicated that release of elasmobranchs will only have a slight positive impact on their populations, due to the high mortality rates when captured (**Figure 3**).

Offset: Mitigating elasmobranch catch offsite.

The final step of the mitigation hierarchy involves compensating for fishing mortality of the species of concern, by investing in actions which increase the probability of another individual in the same stock living to the same age. In other applications of the mitigation hierarchy, this typically involves a financial offset, such as a "bycatch tax," which is invested in conservation elsewhere (Squires and Garcia, 2018; Booth et al., 2019b). However, the low socio-economic status of fishers in Malvan renders such measures unfeasible at present, and we did not assess them further for this analysis.

Feasibility Assessment of Mitigation Measures

This assessment evaluated the feasibility of implementing the measures, both in terms of its social and economic impact on the fishers, and the likelihood of compliance (**Table 4**).

Avoidance: Spatio-temporal closure of the southern fishing ground for 2 months during the pupping season.

Most boat owners (6 of 11) indicated that this measure would negatively affect their incomes. A few believed it would have no impact (n = 4), as they could fish in other locations, and one respondent suggested that his profits would increase once the closure was lifted. The overall opinion of the respondents to this measure was mixed; most respondents had a negative view of this measure, whereas a few indicated their willingness to follow it as it may benefit them in the long term (**Figure 3**).

The months of the post-monsoon season (August–December) were also cited by most owners (n = 8) as the peak months

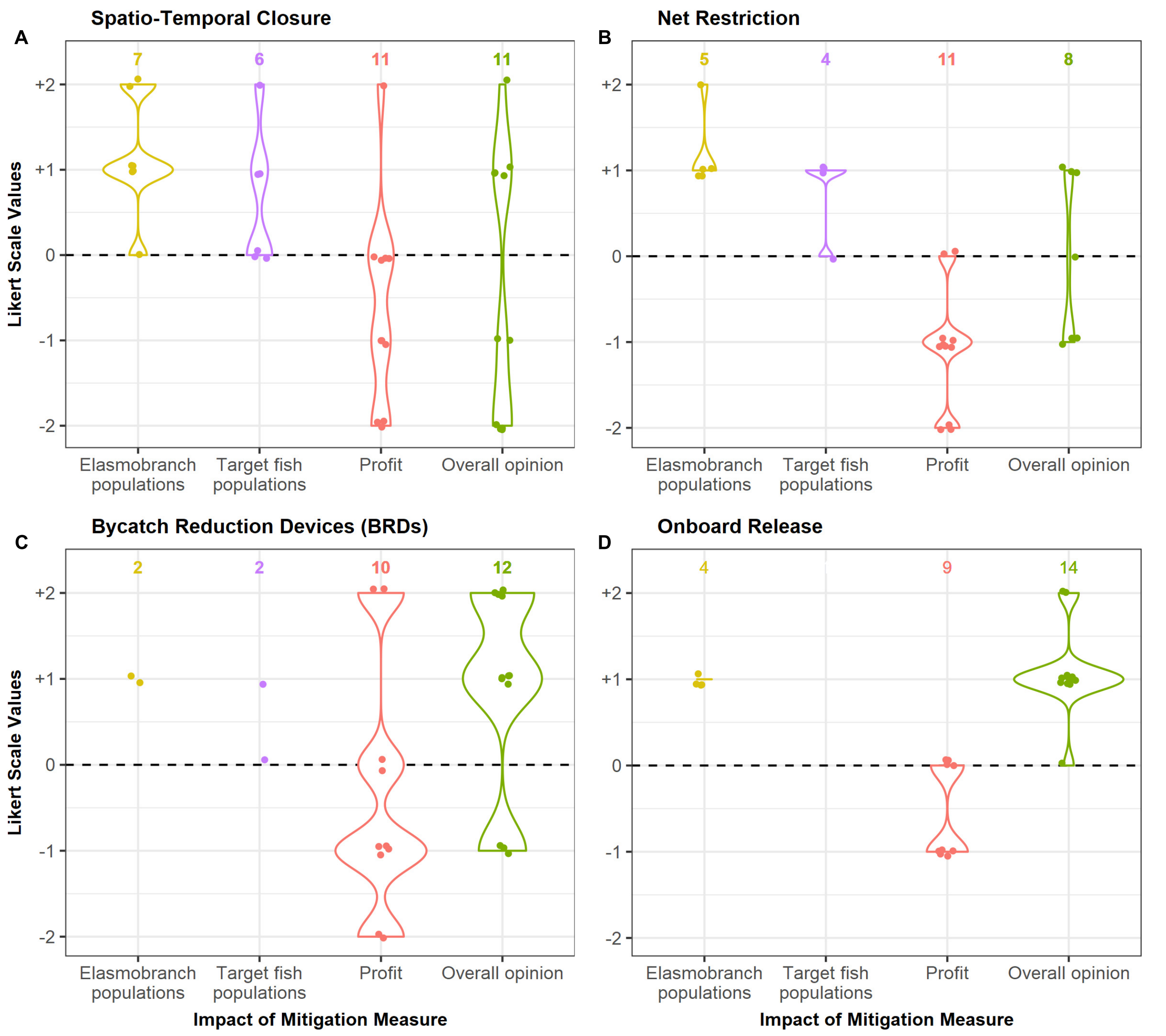

FIGURE 3 | Perceived impact of each mitigation measure **(A–D)** on elasmobranch populations, target species populations and profit of the boat owners, obtained through interviews. Data is represented on a five-point scale from –2 (greater than a 50% decrease) to +2 (greater than a 50% increase). The overall opinion of the respondents on each mitigation measure is also presented here on a similar scale (–2: Very negative opinion to +2: Very positive opinion). Sample size for each response is given above the plot. Questions on the impact of each measure on elasmobranch populations and target fish populations were directed to fishers (total sample size = 7), whereas questions on the impact on profits were directed toward boat owners (total sample size = 11). An overall opinion on the mitigation measure was asked to both fishers and owners. Not all respondents were able to provide answers for every question, hence the sample size represented on the graph may be lower than the total in some cases.

for catch and profit. Hence the closure of a fishing ground for 2 months during this period would likely significantly affect their incomes.

> "If any fishing grounds are closed, we'll have to shut down our boats and go hungry".
>
> – a trawl fisher, age 61.

Some respondents (n = 3) raised concerns regarding the impact of spatial closures on small-scale fishers in the closure region, as their boats would not have to capacity to travel further to fish. Respondents also indicated that compliance may be a problem, as it would be difficult to monitor and enforce such a closure. A summary of the respondents' opinion on all the proposed mitigation measures is provided in **Supplementary Table 4**.

Minimization: Restriction of benthic net use for 2 months during the pupping season.

Respondents stated that although pelagic nets were the primary gear used during the post-monsoon season, benthic nets would occasionally be deployed as well. Most boat owners (n = 9 out of 11) believed that restricting benthic net use would negatively affect their profits. Due to the highly variable nature of

TABLE 5 | Review of bycatch reduction techniques in trawl fisheries for elasmobranchs.

References	Study Location	Fishery	BRD Technique	Elasmobranch species of concern	Exclusion achieved (% of catch reduced)
Zeeberg et al., 2006	Northwest Africa	Pelagic trawl fishery	Escape Tunnel in trawl net	Hammerheads and pelagic sharks	55% of hammerheads 20% of other sharks
Brewer et al., 2006	Northern Australia	Prawn trawl fishery	TED	Small sharks and rays (<1 m)	4.9% of sharks, 25% of rays
Garstin and Oxenford, 2018	Guyana	Prawn trawl fishery	Modified TED	Small rays (<1 m)	59% of rays (especially larger-sized species)
Willems et al., 2016	Suriname	Prawn trawl fishery	TED	Small rays (<1 m)	36% of rays (especially larger-sized species)
Raborn et al., 2012	Gulf of Mexico	Prawn trawl fishery	TED	Blacknose, bonnethead and sharpnose sharks	94% of Blacknose, 31% of Bonnethead, No effect (0%) for Sharpnose
Boopendranath et al., 2006 and Prakash et al., 2016	East coast of India	Prawn trawl fishery	CIFT–TED	Sharks and rays	No effect (0%)

TED refers to a Turtle Excluder Device, and the CIFT–TED refers to the TED developed by the Central Institute of Fisheries Technology (CIFT) in India.

the catch, they believed that restriction of any gear may result in a severe loss for them. The overall opinion toward this measure was negative or mixed (**Figure 3**).

Remediation: Bycatch Reduction Devices (BRDs).

Due to differences in socio-economic contexts, most of the reviewed studies would not serve as suitable proxies to understand the feasibility of BRDs for Malvan. We therefore focused on the Indian case studies testing TEDs (**Table 5**), which found losses of 0.5–3.3% of prawns and target fish at different sites under highly controlled usage by experts. This number is likely to be higher in Malvan due to the commercial value of non-elasmobranch bycatch species that may also escape through the TED, which was not evaluated by these studies. Furthermore, adoption of TEDs by Indian fishers has been extremely limited despite their mandatory use in some states, probably due to their perceived impact on catch and profits (Rao, 2011).

The general concept of a BRD was explained to boat owners, and its possible effects on target catch and bycatch described. Given this information, most (6 of 11) believed that it could significantly reduce their profits. However, a few owners (n = 2) indicated that their profits may increase over the long-term. The overall opinion toward this measure was mostly positive, with nine respondents indicating their willingness to try it and four selecting BRDs as their most preferred of the four options. Respondents were concerned that buying nets with BRDs would be costly, especially as trawl nets need regular replacement due to wear and tear, and suggested these devices be given to boats for free. Some also believed that any technological modification to reduce bycatch may not suit their boats, and would be willing to use a BRD only if it had been developed for and tested in local conditions.

We also referred to the square-mesh trawl nets introduced in Malvan in 2015 to reduce bycatch of juvenile fish (UNDP, 2017), and discussed this with interview respondents as another type of BRD. Although most boat owners (n = 7) had received this net when it was being promoted, only a few (n = 3) stated that they occasionally used it.

> "The problem is, we also catch fish that are small to begin with, like anchovies and sole fish. The square mesh nets reduce our catch of these, and decreases our profits".
>
> – a boat owner, age 28.

Remediation: Onboard release of live individuals.

5 of the 11 boat owners said that release of live animals would have a small negative impact on their profits, while 4 indicated that it would have no effect (**Figure 3**). Sharks and rays were considered low value catch and formed only 1–2% (n = 7) or 5% (n = 3) of their income. Overall opinion toward this measure was positive, and it was the most preferred option among the four proposed measures by most respondents (n = 8) as it caused minimal economic loss and involved little time and effort. Some owners (n = 2) stated that their crew already released live juveniles of many species whenever possible.

Key Uncertainties

There is a high degree of uncertainty associated with the proposed mitigation measures. Lack of data regarding critical habitats of elasmobranchs, and spatio-temporal variation in their use of different habitats, is a major hindrance to designating effective avoidance and minimization strategies. Few bycatch reduction technologies have been developed specifically for elasmobranchs in trawl fisheries, and none in India. Similarly, survival of the study species, if released onboard or even through a BRD, is not specifically known for this fishery, but is likely to be low to moderate. Overall, there is limited understanding of the effectiveness of these measures with respect to both catch and mortality reduction of elasmobranchs, as well as their socio-economic impact. Future research needs to focus on these specific data gaps to address this uncertainty (**Table 4**). Moreover, our dataset was collected over a relatively short time period, and lacks onboard data on discards. Long-term landings and discards data are essential in developing optimal management strategies.

Compliance with the management measures, if implemented, is another major challenge. Only 5 out of the 16 respondents believed that the fishing community would comply with any of the proposed measures, and only if some form of compensation was provided. Respondents mentioned the prevalence of illegal fishing activities such as LED fishing around Malvan, indicating that compliance with any new measures would be unlikely given the challenges in enforcing these existing regulations.

DISCUSSION

Insights From the Mitigation Hierarchy Assessment

Bycaught elasmobranchs in India have a social and economic value (Jabado et al., 2018), which makes bycatch mitigation highly challenging. This study used a novel framework that allowed the systematic assessment of management measures for elasmobranch bycatch mitigation, based on a range of evidence sources. Landings surveys indicated how operational fishery variables affected elasmobranch capture toward designing effective mitigation measures, while interviews provided insights on the perceptions of local stakeholders on the proposed measures. Our study provides the first evidence-based, nuanced and case-specific understanding of elasmobranch bycatch for this fishery, and suggests ways forward for management.

Area-based strategies like Marine Protected Areas (MPAs) are widely used in marine conservation (Shiffman and Hammerschlag, 2016; MacKeracher et al., 2019) and are generally advocated for shark protection (e.g., shark sanctuaries; Ward-Paige, 2017). However, such strategies have had little success in India where they tend to be strict MPAs with little inclusion of the fishing community in the design, implementation or access to the area, leading to violations of MPA rules and conflict between fishers and managers (Rajagopalan, 2009; Bijoor et al., 2018; Muralidharan and Rai, 2020). The Malvan Marine Sanctuary is not yet operational as it has faced considerable opposition from the fishing community due to their exclusion from the entire process (Rajagopalan, 2009). It is clear that area-based strategies in their present format have little scope for success, and need to be approached differently. Our findings suggest that if flexible and case-specific closures or gear regulations were designed with the local community as partners and co-managers, they may be effective (see also Karnad et al., 2019; Rigby et al., 2019b).

Bycatch reduction technologies (BRDs) are generally plagued with implementation challenges (Campbell and Cornwell, 2008). Although some respondents in Malvan provided positive feedback about the adoption of BRDs, their perception may be biased by lack of knowledge regarding this measure. The limited use of square-mesh trawl nets in Malvan to reduce bycatch reported by interview respondents suggests that other BRDs may face a similar response. Furthermore, high levels of uncertainty regarding the effectiveness of BRDs for elasmobranchs, combined with the increasing commercial value of most non-target species, makes this measure somewhat unfeasible at present.

Onboard release of live individuals, particularly species like guitarfish, appears to be the most viable option from a socio-economic perspective. Given that catch rates of guitarfish in trawlers is low, this may be the most cost-effective method to potentially minimize fisheries mortality of these species. The whale shark (*Rhincodon typus*) conservation campaign in Gujarat, on the north-west coast of India, is an example of a successful intervention where fishers have released several hundred sharks caught in their nets, receiving compensation for any damage (Matwal et al., 2014). However, this measure may be applicable to a few species only; post-capture mortality rates for obligate ram ventilators like scalloped hammerheads are too high to support live release (Ellis et al., 2017). Nonetheless, the greater feasibility of this measure should be taken into consideration, even if its direct impacts are low. In a situation where fishers are generally excluded from management decision-making, and there is high uncertainty and a conservation need, building trust and engagement through feasible management options such as release of live individuals is an important first step (Redpath et al., 2013). This can be followed with solutions that have better conservation outcomes.

Our results provide clear evidence for the need for species-specific management strategies, due to the diversity in elasmobranch species characteristics (Dulvy et al., 2017). Capture trends varied between and within taxa; for instance, hammerheads had a higher likelihood of capture in the northern fishing grounds, but when all shark species were pooled, catches were found to be higher in the southern fishing grounds. We highlight the need for different and complementary management measures, which would together provide conservation benefit to a range of vulnerable species (Shiffman and Hammerschlag, 2016).

Market-based approaches, such as economic incentives, form an important component of the mitigation hierarchy as conceptualized for fisheries (Squires and Garcia, 2018). Our study has not considered these due to the lack of such approaches in Indian fisheries at present. However, incentives in the form of eco-labeling schemes or compensation for lost catch can effectively produce behavioral change in fishers (Gjertsen et al., 2010). For instance, an incentive scheme to give premium prices to fishers abiding by bycatch regulations is currently under trial in a small-scale fishery in Peru (Arlidge et al., 2020). Such mechanisms could be explored to encourage uptake of mitigation measures and compensate for lost profits, once motivations and constraints of fishers are better understood (Booth et al., 2019a). Furthermore, biodiversity offset measures can be made more feasible through market-based approaches like taxation of traders or other nodes of the supply chain.

In summary, our study identifies potential steps to ameliorate the complex and seemingly intractable issue of elasmobranch bycatch in Indian coastal fisheries (**Table 4**). On the whole, stakeholders in Malvan were not opposed to elasmobranch conservation as long as it did not compromise their earnings. A good first step would be to promote the live release of guitarfish, through extensive outreach and workshops. Participatory monitoring could aid in addressing research gaps for elasmobranchs and collecting long-term fisheries data, while further building community engagement (Estrella and Gaventa, 1998; Sheil and Lawrence, 2004). Development and implementation of fishery closures or gear modifications using a bottom-up approach may then be successful as long-term management measures. The findings and recommendations of this study will be presented to the local Fisheries and Forest Departments and will also be disseminated among the fishing community in Malvan in the local language.

The Mitigation Hierarchy Framework as a Tool for Bycatch Management

Although the mitigation hierarchy has been discussed conceptually for marine bycatch management (Milner-Gulland et al., 2018; Squires and Garcia, 2018), it has only previously been applied to one case study (Arlidge et al., 2020), which investigated marine turtle bycatch in a coastal gillnet fishery in Peru. While still considered a data-limited fishery compared to large-scale industrialized fisheries, there was richer bycatch data available than the current Malvan case study. In addition, unlike Malvan's elasmobranchs, the marine turtles in that case were not commercially sold. Therefore, our study serves as an important test of its benefits as a decision-making framework for a very challenging situation and identifies scope for improvement.

Given the complexity of the bycatch problem, a single mitigation strategy following a one-size-fits-all approach is not an effective solution (Momigliano and Harcourt, 2014; Shiffman and Hammerschlag, 2016; Squires and Garcia, 2018). The framework facilitated the systematic compilation and critical assessment of multiple strategies to identify nuanced, case-specific, solutions. Moreover, we were able to better understand the challenges associated with classic management measures such as space-time closures. Therefore, the mitigation hierarchy was a useful framework for structuring thinking toward bycatch management of threatened species.

However, there were challenges with applying the mitigation hierarchy to our case study. For instance, setting a quantitative bycatch reduction target was difficult, as elasmobranchs are an exceptionally data-limited group with limited understanding of population dynamics and true fishing mortality for many species, particularly in developing countries (Booth et al., 2019b). Using a less quantitative, more feasible target of reducing elasmobranch bycatch over current observed levels was adequate for this preliminary exploration. Nevertheless, it emphasized the need to better adapt the framework for multi-species fisheries in developing nations with complex socio-economic contexts. In India, this is further complicated by local differences in social, political and economic contexts. For Indian fisheries, it may be more useful to start with a socio-economic assessment of what degree of bycatch mitigation is feasible, followed by the risk and technical assessment to identify priority species for conservation and develop effective management measures. We suggest an adaptive approach iterating the framework over time as trust and capacity, as well as the information base, are developed.

Elasmobranch and Bycatch Management in India

Our assessment began to unpack the problem of elasmobranch conservation at a case study site, and lessons learnt can be applied to elasmobranch management in India more broadly, as well as in other developing countries facing similar challenges. The present study site represents a very small fraction of Indian fisheries, and studies such as this need to be scaled up for sharks and rays across sites and gear types, to develop meaningful mitigation and conservation strategies. Research efforts are currently patchy, and frameworks such as the mitigation hierarchy can guide systematic research to produce scientific data that is relevant to policy making and management (Momigliano and Harcourt, 2014; Shiffman and Hammerschlag, 2016; Milner-Gulland et al., 2018). Most importantly, the human dimensions need to be explicitly studied. Our findings establish that socio-economic feasibility and stakeholder perceptions, rather than technical effectiveness, may be the deciding factors for management. Therefore, understanding the views and socio-economic characteristics of fishing communities is critical to developing conservation interventions (Karnad et al., 2014; Mason et al., 2020).

Lastly, it is important to consider the broader picture. Elasmobranchs form a small component of the incidental catch in Indian fisheries, which ranges from sea snakes, marine turtles and cetaceans to juvenile fish and invertebrates that are either discarded or retained for various commercial uses (Lobo, 2012). The fisheries and gears are equally complex, with a wide assortment of small and large-scale fisheries targeting a variety of species, which often overlap spatially and temporally. Bycatch management will need to integrate specific strategies for these different species and fisheries into a comprehensive action plan at multiple jurisdictional scales. Such an approach needs to be supplemented by research on the drivers of unsustainable fisheries, such as exports and fishmeal production for aquaculture. This improved understanding can then feed into regulatory changes at the local, national and international levels. Interdisciplinary frameworks like the mitigation hierarchy can play a role in operationalizing conservation and fisheries management goals, and shaping policy that integrates environmental sustainability and social justice.

DATA AVAILABILITY STATEMENT

All datasets presented in this study are included in the article/**Supplementary Material**.

ETHICS STATEMENT

The studies involving human participants were reviewed and approved by an institutional ethics committee affiliated with the Dakshin Foundation. The clearance reference number is DF_Ethics committee_HS_2019_May_01. Written informed consent for participation was not required for this study in accordance with the national legislation and the institutional requirements. Ethical review and approval was not required for the animal study because this study conducted research on sharks and rays captured by fishing vessels for the purpose of sale for food. The sharks and rays sampled were hence already dead. None of the study species are Protected under India's Wildlife Protection Act (WPA). Therefore no permits or animal ethics approval was required, in accordance with national legislation and institutional policy.

AUTHOR CONTRIBUTIONS

TG, HB, WA, and EM-G conceptualized this study. TG carried out the data collection and analysis and wrote the first draft. All authors contributed significantly to revising the manuscript.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2020.00571/full#supplementary-material

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Managing Bigeye Tuna in the Western and Central Pacific Ocean

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The Western and Central Pacific Fisheries Commission (WCPFC) is responsible for managing highly migratory species in the Western and Central Pacific Ocean (WCPO), and has been interested in managing bigeye tuna as stock assessments prior to 2017 indicated that the stock was experiencing overfishing. This paper provides some background on the primary fisheries catching bigeye tuna in the WCPO, describes the various policies within the conservation and management measures adopted by the WCPFC, discusses the effectiveness of such policies, and concludes with some suggestions for future policies for consideration.

Keywords: bigeye tuna, purse seine, longline, western and central Pacific Ocean, bycatch

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INTRODUCTION

The Western and Central Pacific Ocean (WCPO) contains the largest tuna fisheries in the world, with catches in 2018 contributing to 55% of global tuna catch (Williams and Reid, 2019). Several tuna species are caught in the WCPO including skipjack (*Katsuwonus pelamis*), yellowfin tuna (*Thunnus albacares*), bigeye tuna (*T. obesus*), albacore (*T. alalunga*) and Pacific bluefin tuna (*T. orientalis*), and the predominant gear types include purse seine, longline, pole and line and troll.

The Western and Central Pacific Fisheries Commission (WCPFC or Commission) is the regional fisheries management organization (RFMO) responsible for managing highly migratory species in the WCPO. The WCPFC entered into force in 2004 and has the largest area of application (hereafter WCPFC Convention Area) of the five tuna RFMOs. The WCPFC Convention Area covers almost 20% of the earth's surface and generally encompasses the Pacific Ocean west of 150° W to the Asian continent. As of December 2019, the WCPFC is comprised of 26 members, 7 participating territories and 8 cooperating non-members[1] (collectively referred to as CCMs). The Commission meets annually and, to date, all decisions on conservation and management measures (CMMs) have been made by consensus.

Bigeye tuna has been a stock of particular interest in the WCPO. Although the 2017 stock assessment indicated the stock was not experiencing overfishing and was not overfished, previous assessments indicated that the stock was experiencing overfishing, and the 2014 stock assessment

[1]Member to the WCPFC include Australia, China, Canada, Cook Islands, European Union, Federated States of Micronesia, Fiji, France, Indonesia, Japan, Kiribati, Korea, Republic of Marshall Islands, Nauru, New Zealand, Niue, Palau, Papua New Guinea, Philippines, Samoa, Solomon Islands, Chinese Taipei, Tonga, Tuvalu, United States of America, and Vanuatu. Participating territories to the WCPFC are American Samoa, Commonwealth of the Northern Mariana Islands, French Polynesia, Guam, New Caledonia, Tokelau, and Wallis and Futuna. Cooperating non-members to the WCPFC are Curacao, Ecuador, El Salvador, Nicaragua, Panama, Liberia, Thailand and Vietnam.

indicated that the stock was overfished (Harley et al., 2010, 2014; McKechnie et al., 2017). The two primary fisheries in the WCPO that catch bigeye are the deep-set longline fishery, which targets adult bigeye, and the purse seine fishery, which targets skipjack and yellowfin, and catches juvenile bigeye incidentally. The WCPFC has since its inception grappled with reducing fishing mortality for bigeye tuna, and adopted many CMMs aimed at managing and conserving this species of tuna. Since the 2017 stock assessment, the management focus in the WCPFC for bigeye tuna has shifted from reducing overfishing to maintaining average spawning biomass at 2012–2015 levels.

The purpose of the paper is to provide relevant background on bigeye tuna and the longline and purse seine fisheries responsible for significant bigeye tuna extraction in the WCPO, describe the various CMMs adopted by the WCPFC, evaluate the effectiveness of the various CMMs, and provide some recommendations for future consideration. This paper primarily focuses on the scientific side of management and the potential role of incentive-based strategies. We recognize that many factors play a role in multilateral decision-making and that there is an extensive body of economic, game-theoretic, institutional and politics literature available, but detailed discussions of such considerations are beyond the general scope of this paper (Barrett, 2003; Hanich, 2012; Libecap, 2014; Norris, 2015; Barret, 2016). We further recognize that bycatch policies may contain implicit or explicit allocation among CCMs, which may be highly contentious.

BACKGROUND

Bigeye tuna is an important component of the WCPO tuna catch. In 2018, the provisional catch estimate of bigeye tuna was 142,402 mt and was estimated to be valued at $780 million United States dollars (Williams and Reid, 2019). For many years, stock assessments conducted by the Oceanic Fisheries Programe of the Pacific Community (commonly known as SPC-OFP) and endorsed by the Commission's Scientific Committee concluded that WCPO bigeye have experienced rates of fishing mortality above the rate of fishing mortality at maximum sustainable yield (MSY) (Harley et al., 2010, 2014; Davies et al., 2011). The 2014 stock assessment also indicated that the stock was overfished, as the spawning biomass was below the limit reference point (Harley et al., 2014). In 2017, the WCPFC Scientific Committee reviewed a new stock assessment which included a new growth curve and regional structure, and these factors along with estimated increases in recent recruitment contributed to a much rosier outlook on stock status (WCPFC, 2017). The Scientific Committee noted that biomass was now greater than the limit reference point so the stock was not overfished, and that fishing mortality was below fishing mortality at MSY so the stock was not experiencing overfishing. Although the stock status for bigeye tuna improved, the Scientific Committee noted that some regions have large juvenile mortality and recommended that the Commission continue to reduce fishing mortality on juveniles in order to increase stock size (WCPFC, 2017). This change in stock status was surprising to some observers given that some earlier accounts had stressed the failure of the Commission to adopt adequate conservation measures (Hanich, 2012).

Bigeye tuna are predominantly caught by longline or purse seine vessels with those two gear types accounting for 85–90% of the WCPO bigeye catch each year. From 2014–2018, longline catch of bigeye tuna represented ∼45% of the total bigeye catch while purse seine catch of bigeye tuna was ∼43% of total bigeye catch (SPC, 2019d). Most of the purse seine caught fish are considered juvenile (∼3 kg), while the longline fishery generally catches adult sized fish (∼40 kg) (Abascal et al., 2014; McKechnie, 2014).

Longline vessels in the WCPO target several species of tuna and billfish depending on the area fished, and set type. Longline fleets—from Japan, Korea, Taiwan, and China (along with smaller localized fleets out of Hawaii, Fiji, etc.) – target bigeye and yellowfin tuna for sashimi markets. Longline vessels have operated in the WCPO since the early 1900s, and numbers of vessels have generally fluctuated between 3,000–6,000 vessels for the last 30 years (Williams and Reid, 2019). The number of longline vessels and overall catch peaked in the early 2000s, and both vessel numbers and bigeye catch have subsequently declined over the past 15 years. The WCPO longline tuna catch from 2018 was valued at over $1.7 billion United States dollars, with the value of the longline bigeye catch ($660 million United States dollars) accounting for nearly 40% of the total (Williams and Reid, 2019).

Purse seine vessels in the WCPO generally target skipjack and yellowfin tuna, but also catch several other species, including juvenile bigeye tuna [fish under 103 cm (Farley et al., 2017)]. Since the inception of the purse seine fishery in the WCPO in the 1970s, the number of vessels as well as the total catch of tunas have steadily increased. In 2015, there was a record high of 308 purse seine vessels fishing in the WCPO purse seine fishery (excluding the domestic purse seine fisheries in Philippines, Indonesia and Vietnam) and in 2014, there was a record high WCPO purse seine catch of 2,059,008 mt of tunas (Williams and Reid, 2019). The WCPO purse seine catch from 2018 was valued at over $3.4 billion United States dollars (ex-vessel), which represents over 50% of the total ex-vessel value of the 2018 WCPO tuna catch (Williams and Reid, 2019).

Unlike the longline fishery, the WCPO purse seine fishery does not target bigeye tuna, but catches juvenile bigeye tuna incidentally. Purse seine vessels set large nets that act as big areas that confine the tuna, which are then pursed into a smaller sized net and fish are then scooped (brailed) onto the vessel and put immediately into the fish hold for freezing. In the WCPO, vessel operators generally engage in two types of sets; unassociated sets or sets on free schools of yellowfin and skipjack tuna, or associated sets or sets made on fish aggregating devices (FADs), which can be naturally or man-made floating objects. Up until the mid-1990s, purse seine vessels made the majority of their sets on free schools and on naturally floating objects such as logs. This pattern changed in the mid-1990s, when purse seine vessels started to increasingly rely on man-made FADs. In 2018, the proportion of unassociated sets was 64% and the proportion of associated sets was 36% (Williams and Reid, 2019) in the fishery as a whole, however, some national fleets rely more on FAD sets than others due to economic factors. FAD sets tend to have a higher catches in weight per set so although associated sets only made up 36% of the sets, catch from associated sets made up 51% of the total catch (Williams and Reid, 2019).

FAD sets not only have higher catches per set than sets on free schools, but also improve the odds that a purse seine vessel will have a successful set (fewer "skunk" sets – which means no catch). Although the opportunity to use FADs increases the economic success of a purse seine vessel, FAD sets tend to result in catches of smaller-sized fish, greater bycatch, and catches with higher proportions of bigeye tuna as compared to free-school or unassociated sets (Dagorn et al., 2012). Purse seine-caught bigeye tuna prior to the 1990s represented 30% or less of the total WCPO bigeye catch, and since the 1990s increased to represent 30–49% of the total WCPO bigeye catch (SPC, 2019d). Although, the purse seine fishery catches less bigeye by weight than the longline fishery, the purse seine fishery catches far greater numbers of small bigeye, and this removal of small bigeye has effects on the level of maximum sustainable yield (Davies et al., 2011; Harley et al., 2014).

Juvenile bigeye can be difficult to distinguish from juvenile yellowfin, and obtaining accurate estimates of purse seine-caught bigeye tuna has proved challenging. It has been found that fishermen generally underestimate catch of bigeye tuna on their logbooks, and bigeye is commonly misreported as yellowfin or skipjack (Lawson, 2014). CCMs annually submit catch data to the WCPFC for their fisheries and in reporting catches by their purse seine fleets, most CCMs do not make adjustments from what is reported by vessel operators in logbooks (i.e., the information is unadjusted for what is known to be underestimates of bigeye tuna catch). SPC–OFP, which is the science and data provider to the WCPFC, not only compiles reported catch by members, but also uses observer data on catch by species and size to estimate each member's purse seine bigeye tuna catch. Cannery data has also been used by some CCMs to better estimate bigeye tuna catch from purse seine vessels as canneries produce reports on quantities of fish accepted by weight and species. Canneries may pay different prices for fish depending on species and size class, and cannery estimates are believed to be relatively accurate for larger sized bigeye tuna (>3 kg). However, there is little incentive for canneries to accurately identify smaller-sized fish, as there tends to be no price differential between species for the smallest sized fish. Most small sized (<2 kg) bigeye tuna, yellowfin tuna, and in some cases skipjack tuna, are mixed together and reported by the canneries as a mixture or as purely skipjack or yellowfin tuna.

APPLICATIONS OF POLICIES

Reducing fishing mortality – especially on juvenile bigeye – has been a priority for the WCPFC since the Commission's establishment, and the WCPFC adopted its first CMM for bigeye and yellowfin tuna in 2005. As of June 2020, the CMM for bigeye, yellowfin and skipjack tuna has been revised and replaced nine times. The most current version, adopted in December 2018, is CMM 2018-01. Most changes to the original measure have been fairly minimal, however, greater changes occurred in the CMMs adopted in 2008, 2014, and 2017 roughly concurrent with when changes were made to the management objectives for bigeye tuna. CMM 2018-01 is effective through February 2021, and the Commission is expected to work on a replacement measure at its 2020 annual meeting. The Commission adopted a biomass-based limit reference point of 20% of unfished spawning stock biomass in 2012, and the most recent objective for bigeye tuna management comes from paragraph 12 of CMM 2018-01 which states, "Pending agreement on a target reference point the spawning biomass depletion ratio ($SB/SB_{F=0}$) is to be maintained at or above the average $SB/SB_{F=0}$ for 2012–2015.

The WCPFC has adopted a number of restrictions for the purse seine fishery. The input or effort-based restrictions used by the WCPFC to manage the purse seine catch of bigeye tuna have included prohibiting the use of FADs during certain time periods, and limiting the number of FAD sets by each CCM over a year. These effort-based restrictions (process standards) are all examples of command-and-control policies (regulatory measures that mandate specific vessel behavior through limits or standards on technology, process of production, or the catch and bycatch – performance) and the FAD limits have primarily been flagged-based (counting against the limit of the CCM to which the vessel is flagged or chartered) though there have been some zone-based exemptions some years for small island developing states (SIDS). The WCPFC has also adopted a "full"-retention policy for tropical tunas, which could be construed as incentive or market-based. The retention policy, adopted as a new provision in 2008, requires vessels to retain all bigeye, yellowfin and skipjack tunas caught except in some limited circumstances[2]. As mentioned previously, canneries pay different prices by species and size, and as prices for small fish are much less than prices for big fish, this retention policy was adopted as stated in paragraph 27 of CMM 2008-01, "to create a disincentive to capture small fish and to encourage the development of technologies and fishing strategies designed to avoid the capture of small bigeye and yellowfin tuna. . .." Full retention also creates an indirect or opportunity cost of foregone catches of larger sized tunas. It is not clear full retention has ever worked or was actually adopted with the cited reasoning being the main objective of the policy.

For the longline fishery, the Commission has used catch limits (performance standards) to conserve and manage bigeye tuna catch. Initially, CCMs that historically caught over 2,000 mt of bigeye were not to exceed either the average annual catch from 2001–2004 or the catch in the year 2004 (at the discretion of the CCM), and any CCM that historically caught less than 2,000 mt were not to exceed 2,000 mt. As the Commission later believed more reductions were needed, CCMs that caught greater than 2,000 mt were required to reduce catches by anywhere from 10 to 30% starting in 2009, and many of these same CCMs were required to reduce catches further in 2015. For several CCMS, longline catches had declined in years leading up to the adoption of CMM 2008-01 so the reductions were not necessarily limiting.

[2]The exceptions for the catch retention policy are (1) if on the final set of the trip, there is insufficient well space to accommodate all fish caught on the set, (2) when the fish are unfit for human consumption for reasons other than size, and (3) when serious malfunction of equipment occurs.

EVALUATING PERFORMANCE

Over time, the WCPFC's objectives for bigeye tuna have shifted in part due to changes in stock status as well as due to progress the Commission has made in developing reference points for bigeye tuna. In this section, we will evaluate the three objectives for bigeye tuna from 2008–2011, 2012–2016, and 2017–2020. We will also evaluate the effectiveness of some of the various types of policies undertaken in the different CMMs over time.

In CMM 2008-01, the WCPFC's objective for bigeye tuna was to reduce bigeye fishing mortality by at least 30% from the annual average from 2001–2004, or 2004. Despite reductions in longline limits, catch retention and seasonal FAD closures, annual fishing mortality for both adult and juvenile bigeye from 2009–2012 remained at or above the levels from 2001–2004 (McKechnie et al., 2017).

In CMMs adopted from 2012–2016, the WCPFC's objective for bigeye tuna was to reduce fishing mortality for bigeye tuna to a level no greater than F_{msy}. SPC investigated the potential effectiveness of the various CMMs (CMM 2013-01, CMM 2014-01, and CMM 2015-01) on the bigeye tuna stock and in general, found that fishing mortality would only remain below F_{msy} under optimistic fishing scenarios where the measure worked as intended and the FAD closures remove FAD sets from the fishery (SPC, 2014, 2015, 2016). The 2014 stock assessment also found that recent (2008–2011) fishing mortality was greater than F_{msy} (Harley et al., 2014). However, as noted above, the 2017 stock assessment had a number of changes that led to very different conclusions in which recent fishing mortality (2011–2014) was less than F_{msy}.

As a result in the shift in stock status, the WCPFC modified its objective for bigeye in CMMs 2017-01 and 2018-01 to read "Pending agreement on a target reference point the spawning biomass depletion ratio (SB/SBF = 0) is to be maintained at or above the average SB/SBF = 0 for 2012–2015." SPC has conducted a number of analyses to evaluate the potential for CMM 2018-01 to achieve its objectives for the three stocks of tropical tunas including bigeye tuna (SPC, 2017, 2018, 2019c). In general, achieving the objectives for bigeye tuna are strongly influenced by the recruitment scenario in that scenarios with recent recruitment tend to achieve the spawning biomass depletion ratio objectives while scenarios using long-term recruitment indicate that the objective is not likely to be met (SPC, 2018, 2019c). The analyses also evaluate varying levels of effort and compliance such that maintaining average effort levels from 2013–2015 result in slightly higher levels of spawning biomass in 2045 than an optimistic scenario and a pessimistic scenario (SPC, 2018, 2019c). A new stock assessment is being conducted in 2020, and results from that stock assessment should help inform whether the current objectives are being met.

Longline catches of bigeye tuna have declined over time, and CCMs have collectively been successful in reducing longline catch of bigeye tuna. However, it is difficult to determine whether the decline in bigeye catches in the longline fishery is due to the restrictions imposed by WCPFC members or their respective domestic fleets or if other factors, notably market forces, have played a larger role in the decline. As noted above, longline fleets for some CCMs had been declining before CCM 2008-01 came into effect and so though several fleets had significant reductions from historical levels, some CCMs were easily able to ensure their catches were below their limits without any active management. As these same CCMs with limits have consistently stayed well under their bigeye catch limits since the adoption of catch limits in 2008, their reductions in catch have offset overages by other CCMs with limits as well as increases by CCMs that are not limited. Longline effort in the core area of the of tropical WCPFC longline fishery was higher from 2011–2015 than levels in 2000–2004, but catch and catch per unit effort (CPUE) have been declining over time. Declines in longline effort may also be attributed to other factors such as rising operating costs, decreases in market prices, and increased regulation (Miyake, 2007). Although there have been significant declines in bigeye catch over time, the fishery impact of the longline fleet has only declined slightly over the past 10 years.

Reducing bigeye mortality in the WCPO purse seine feet has also been challenging and the primary mechanism for constraining bigeye catch has been a seasonal FAD fishing/setting prohibition period where vessels are not allowed to set on FADs. The initial period was for 2 months in 2009 and then 3 months from 2010 to 2012. From 2013 to 2017 CCMs had the option to use an additional 4-month of FAD closure or reduce their total FAD set number below a certain level. From 2018–2020, CCMs have a 3-month FAD closure as well as a 2-month FAD closure on the high seas. While not ceasing completely due to a number of exceptions, the catches of bigeye tuna by purse seine vessels decreased dramatically during the FAD closure months, while in general CPUE of skipjack and yellowfin only slightly decreased below average in some months of the 2014 and 2015 closure (Pilling et al., 2013; Williams and Terawasi, 2016). The fishery has experienced classic "effort creep" (productivity growth) over time with increased catchability as well as increases in the number of sets per day over time (Tidd et al., 2015). Since 2009, the number of unassociated sets has nearly doubled from levels in 2000–2004. The average annual number of FAD sets initially remained similar to those from 2006–2009, but declined around 12% from 2015–2018 (SPC, 2019a).

CMMs 2014-01, 2015-01, and 2016-01 contained a footnote whereby if a CCM could show that their bigeye tuna catch levels had dropped to 55% of its 2010–2012 levels, then that CCM did not have to apply the complete FAD prohibition on the high seas in 2017. After some controversy, several CCMs were found by the Commission to have met this requirement in 2016 and stated that they would be applying this exemption in 2017 (WCPFC, 2017). A few CCMs achieved these reductions through attrition in their fleets unrelated to any efforts to decrease their own bigeye catch, but since the passage did not have any limitations on how those reductions were made, they were still able to apply the exemption. A few CCMs worried that CCMs that applied the exemption as written in CMMs 2014-01 and 2015-01 could result in high bigeye catch due to unlimited FAD sets on the high seas (WCPFC, 2017). The Commission thus adopted a revised footnote in CMM 2016-01 in an attempt to limit the bigeye catch from unlimited FAD sets on the high seas by adding a provision that CCMs need to ensure that their bigeye levels remain under the limits needed to achieve the exemption, but it was agreed that this is difficult to monitor in a timely fashion due to issues noted above with estimating bigeye catch in the purse seine fleet (WCPFC, 2017).

The WCPFC has adopted one incentive-based policy, the above cited full retention policy for small tropical tunas. The retention policy was adopted in the hopes that retaining small fish would be a disincentive for vessels (due to the costs of lower revenues from lower prices with smaller fish and foregone revenues from foregone catch) and this would induce technological or behavioral responses to avoid catching small fish. The full retention policy has led to declines in discards, and WCPO purse seine discard rates fell from ∼3% of estimated catch before the catch retention requirement went into effect to 2% of estimated catch after the catch retention requirement came into effect (Chan et al., 2014, SPC, 2019b). Although the retention policy has led to decreases in discards, it is unclear whether this has created any disincentives for fishermen to actually catch small tuna and perhaps could be an area of future study. Canneries may pay low prices for small fish, and it would be interesting to investigate whether this has led to vessels retaining the fish for sale rather than changing their behavior to avoid small sized fish. In times of especially high ex-vessel fish price (e.g., in excess of $2,000 USD/ton) – operators will catch and land as much small fish as the market demands (R. Clarke, pers.com.). The direct costs of a longer trip could also exceed the incremental increase in revenue given the high cost of fishing days.

SUGGESTIONS FOR CONSIDERATION

In December 2018, the Commission adopted CMM 2018-01, which is set to expire in February 2021, and a new measure will need to be renegotiated in December 2020. This section discusses various alternative policies that could be considered in managing bigeye tuna. Some of these ideas could be implemented by WCPFC, whereas others may be beyond the scope of WCPFC, but could be supported by members themselves, regional groups such as PNA, or even by consumer groups.

Adopting a TAC

The Commission may consider in a more material way adopting a total allowable catch limit (TAC) for bigeye tuna, which could help ensure that the total catch of bigeye from all its fisheries would be within a level that would meet its objectives. Although the Commission has adopted limits for some CCMs, it has not set limits for *all* CCMs nor has it set an overall TAC. In the longline fishery, this lack of limits on all CCMs allows some SIDS to expand their catch histories, but could potentially be problematic if increasing catches eventually lead to overfishing. The Commission tasked itself in paragraph 44 of CMM 2018-01 to adopt a longline limit for bigeye tuna on the high seas by 2020, but as of June 2020, this has yet to occur. The Commission can estimate levels—by general size class and adopting an overall TAC within these levels could help to ensure the overall conservation objectives could be reached.

Some of the tensions amongst Commission CCMs in adopting limits are how to divide the conservation burden between the various fisheries that catch bigeye at different life stages. As mentioned previously, the purse seine fishery primarily catches juvenile bigeye whereas the longline fishery primarily catches adult bigeye. Although removals by both fisheries impact the size of the spawning stock biomass and the maximum sustainable yield for the stock, catches—on a by-weight basis—of the relatively younger bigeye from the purse seine fishery have a much greater impact than the relatively older bigeye in the longline fishery (McKechnie et al., 2017). The WCPFC has thus far tried to limit both the purse seine and longline fishery sectors, but could consider focusing its efforts more heavily on the purse seine sector as purse seiners have a greater fishery impact with their catch of juvenile bigeye particularly in the tropical regions and the purse seine fishery is not targeting bigeye, but catching them incidentally (McKechnie et al., 2017). The WCPFC could consider dividing the overall limit by fisheries based upon fishery impact, and could extend this to a market-based scheme where there could be transfers between fisheries-when needed or deemed appropriate though any allocation either zone-based or flag-based is likely to be contentious.

Longline catches in the WCPO have declined since 2004, but allocations may not necessarily be efficient as some CCMs do not fully utilize their quotas whereas other CCMs fully use or exceed their quotas. To date, WCPFC has not really discussed transfer of limits, though transfers regularly occur in other RFMOs such as the International Commission for the Conservation of Atlantic Tunas (ICCAT) and the Inter-American Tropical Tuna Commission (IATTC). The United States allows its territories to transfer some of their bigeye limits[3] to sections of the United States fleet in exchange for funds for fisheries development projects. This has had the benefit of allowing that fleet to continue fishing through the year and has benefited the territories through funds for fisheries development projects. The WCPFC does allow for charters where vessels flagged to one CCM can enter into agreements with second CCM and catch under the charter is attributed to the second CCM. However, it has not always been easy to ensure catch from charters is attributed correctly and there has been some problems with double-counting. Perhaps if the WCPFC had a clear effort or catch transfer mechanism between CCMs participating in the fishery then this would help to ensure that limits are being used efficiently and transparently in a way that everyone is aware of where the catch is occurring.

Some of the struggles in setting limits are also due to issues around allocation of fishing privileges amongst CCMs. The WCPFC Convention lists a variety of elements to be considered in formulating allocations and the specific articles are referenced in paragraph 44 of CMM 2018-01 in discussing the development of a framework for allocating limits – although no prioritization scheme has been agreed upon. Dividing up the WCPO fisheries pie is very contentious particularly since most purse seine fishing in the WCPFC takes places within the EEZs of SIDS whereas much of the effort comes from the fleets of distant water fishing nations. The Commission has avoided making concrete decisions

[3]The United States territories do not have bigeye limits in the WCPFC, but the United States government has established domestic limits for each territory.

about allocation to date though they have recognized the need to do so. Paragraph 42 of CMM 2018-01 states, "The limits set out. . .. do not confer the allocation of rights to any CCM and are without prejudice to future decisions of the Commission." Many of the Pacific Islands countries are advocating the continuation and expansion of zone-based management, and the PNA has stated it intends to operate a longline VDS system in its members' zones, and it unclear how this will influence the development of future Commission conservation measures. Each Commission member is motivated to protect their interests, and this can result in policies that may not necessarily promote sustainability much less economic efficiency.

Although the WCPFC has not discussed an overall TAC for bigeye, adopting limits and the allocation of those limits will be a focus of the WCPFC in the near-term as CMM 2018-01 contains provisions that state that the Commission will agree to hard limits in the purse seine fishery (catch or effort in the high seas) and longline (bigeye catch) as well as a framework for allocation of those limits by 2020.

Incentive-Based Approaches to Bycatch Reduction

The seasonal FAD closures have been effective at maintaining fishing mortality for bigeye tuna. If circumstances for bigeye tuna were to change such that the length of FAD closures become sufficiently long then the cost to the vessels can become prohibitive and incentive-based approaches can lead to lower costs, flexibility in supplying processors, and bycatch reduction. In the following discussion, we explore some incentive-based approaches that could lead to least cost bycatch reduction.

Invest in Methods to Better Estimate Purse Seine Bigeye Catch in Real-Time and Consider Transferable Purse Seine Limits

To date WCPFC has placed primarily input controls on the purse seine fishery which maintained recent levels of fishing mortality and stock biomass. The WCPFC could consider output controls for the purse seine fishery as they could help ensure catch reductions. However, one key issue preventing the adoption of output controls for bigeye catch in the purse seine fishery is that bigeye catches are difficult to estimate in real-time or near real-time with certainty because they generally represent a very small percentage of the total catch. Additionally, independent verification of landings in multiple countries is difficult and costly. SPC-OFP can adjust CCMs' catch estimates using fishery observer and port sampling information, but generally only months after the fishing year is complete. As reporting and monitoring move to more timely electronic methods, it should be possible to develop schemes that combine logbook and observer data to better estimate bigeye catch in near real-time. These near real-time estimates could be compared to or audited by port samplers as well as cannery receipts.

Is should be noted that the problem of accurately identifying species in tuna catches is not unique to the WCPFC, but also plays out in the other tuna-RFMOs. In fact, the IATTC, the counterpart to the WCPFC in the eastern Pacific, faces many of the same issues and struggles. The IATTC has chosen to manage its fisheries in a similar fashion to the WCPFC, with catch limits for the longline fishery and effort limits for the purse seine fishery. In general, the tactics are similar in that both Commissions adopt command-and-control type provisions. The IATTC has adopted full closure periods instead of FAD closure periods, and has a fixed time area closure for an area of the high seas referred to as the "corralito." At the 91st Extraordinary Meeting of the IATTC in February 2017, the IATTC considered a proposal to have bigeye performance limits that each vessel would have to abide by, but this option was difficult because the IATTC would be responsible for deriving in near-real time vessel-specific catch estimates of bigeye tuna. At some point, reducing the uncertainty in real-time bigeye catch estimates is a critical missing piece to allowing better management of bigeye catch in the purse seine fishery.

If it becomes possible to accurately estimate bigeye catch in real-time or near real-time, the WCPFC might consider developing bigeye catch limits for purse seine vessels as this could likely create direct incentives to reduce bigeye catch and in turn fishing mortality from the purse seine fishery. This limit could be implemented on a by-vessel basis or for a particular fleet. The Commission and/or members could consider allowing transfers of limits or unused portions of limits, called credits, as needed through a credit system so that the purse seine bigeye catch limit would be used efficiently. Some flexibility in landings throughout the year could potentially smooth ex-vessel prices and assure a more steady supply for processors. If bigeye catches can be easily estimated, CCMs could potentially consider invoking some sort of tax or penalty for catching juvenile bigeye or even yellowfin. This tax may not be monetary, but in-kind such as additional days fishing, such as was implemented in the Scottish troll fishery for cod bycatch (Squires and Garcia, 2018).

Effort Incentives

The WCPFC, other members, or regional organizations could consider initiating fees for FADs (deploying or setting) in the WCPFC area, in effect pricing FAD usage to account for otherwise uncosted ecological impacts and an incentive-based approach. The PNA recently announced their intention to have vessels that fish in their zones pay an additional fee for any FAD sets made in their zones. Most purse seine fishing takes place in PNA waters, and so this could be an effective mechanism to control FAD sets on top of any FAD limits that the Commission adopts. By pricing FAD sets, residual catch of juvenile bigeye (and unpriced bycatch such as oceanic sharks) receives a cost, which is shared among fishers, supply chain firms, and consumers according to their ability to pass on or absorb these costs. This indirect way to price juvenile bigeye (and bycatch) is less effective and efficient than direct pricing of juvenile bigeye catch but is less expensive to implement and more likely to achieve compliance and easier to enforce. The Commission does not currently have a mechanism to enact charges on vessels, but if fees were initiated in other areas such as within other EEZs or on the high seas, these fees could be used to support research into ways to improve data estimates or others ways to fund bycatch reduction technologies.

Consumer Preference and Ecolabels

The WCPFC has a limited ability to affect consumer preference; however, this is another area that could exert greater influence on bigeye catch. Consumer demand for sustainably caught tuna has led several companies in the Western Pacific to pursue certifications such as those offered by the Marine Stewardship Council (MSC) for their free school catch. Some consumers are willing to pay a premium for MSC-certified tuna, and this price premium is theoretically passed down to vessel operators and owners from canneries seeking tuna caught from FAD free sets. The approach could also be implemented through a simple industry standard through supply chain requirements. Market demand for FAD-free tuna could prove to be beneficial for bigeye conservation as purse seine vessels typically catch bigeye in association with FAD sets. If the price premium is sufficiently high, this might further incentivize vessels to catch more tuna without using FADs through a positive incentive-based approach. Currently, only certain markets appear willing to pay a material premium for FAD-free fish and large markets like the United States continue to show limited preference to FAD-free sourced fish (Gutierrez et al., 2016; van Putten et al., 2020, R. Clarke, pers.com.).

Real-Time Spatial Management

One approach that has successfully reduced bycatch in many fisheries, with potential in the purse seine industry for limiting juvenile bigeye (and bycatch of oceanic sharks and other species) is real-time spatial management (RTSM) implemented under either a co-management or self-governance approach (Hobday and Hartmann, 2006; Little et al., 2015). Should technology improve in the future such that accurate estimates of bigeye tuna are possible, real-time and near real-time information from the electronic sensors of buoys attached to FADs on species density and mix under the FADs could be shared among fishers to incentivize vessels to leave areas and/or set on FADs of high juvenile bycatch. The information can be shared through a private, specialized company to insure data privacy and integrity. RTSM can also integrate this data with real-time biological, oceanographic, and economic data from satellites and remote sensing, and animal tracking and tagging, and using advanced analytical techniques such as machine learning, to either predict key species distributions and/or to indicate real-time "hotspots." Predictions from models can be provided by either private or public bodies as a public good available to all or as a private good only available by subscription (e.g., Turtlewatch).

RTSM needs to be incentivized. Credit systems discussed above, credit systems through reward of extra FAD sets otherwise held in reserve, rebates from FAD pricing, penalties and fines – either explicit or implicit through longer closed seasons or fewer allowable sets, are all possibilities.

Deposit-Refund Systems

Finally, one speculative approach is a deposit-refund system to clear the water of FADs during closed periods, limit ghost FADs, reduce marine debris, and incentivize more "eco-FADs." Deposits are required for each FAD, which is refunded to any party returning the FAD at the end of an open season. More "eco-FAD" designs that reduce bycatch might have lower deposit and refund rates that incentive adoption if there are not RFMO technology standards that mandate such designs. Economic lifetimes of FADs are relatively short due to high rates of physical depreciation, leading to more FADs that receive deposits than those that exist to receive the refund. 'Revenue neutrality' may require higher refund rates.

CONCLUSION

The Commission's objectives for managing bigeye have shifted over time as stock status changed from one experiencing overfishing prior to 2017 to one that is not experiencing overfishing from 2017 forward. The Commission has adopted a number of CMMs to work toward the different objectives over time, and these have resulted in mainly "command-and-control" type policies for the purse seine and longline fisheries. Evaluations of CMM 2018-01 indicate that the objectives for bigeye may be achieved if recruitment remains at recent levels, but declines in spawning biomass may occur is recruitment levels are more similar to the long-term average (SPC, 2019c). If current approaches for bigeye management become no longer tenable, the WCPFC may want to consider incentive-based approaches that lead to least-cost bycatch reduction and help maintain vessel profitability. The Commission will be challenged to develop a new CMM for tropical tunas, and hopefully bigeye can be managed in ways to meet the Commission's objectives of long-term sustainability whether that be the result of decisions by the Commission, individual CCMs or other regional groups or consumer demand.

AUTHOR CONTRIBUTIONS

VP wrote most of the manuscript with contributions from DS. Both authors contributed to the article and approved the submitted version.

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Switching Gillnets to Longlines: An Alternative to Mitigate the Bycatch of Franciscana Dolphins (*Pontoporia blainvillei*) in Argentina

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The franciscana dolphin (*Pontoporia blainvillei*) is considered the most threatened cetacean in the South Western Atlantic due to bycatch in gillnet fisheries of Argentina, Uruguay, and Brazil. As gillnet fisheries operate in the same areas inhabited by dolphins, methods and strategies to reduce bycatch require particular attention. This study investigated the potential of switching gillnets to bottom longlines to reduce franciscana bycatch rates while maintaining economic returns in a small-scale artisanal fishery in Argentina. Trials were conducted in Bahía Samborombón and Cabo San Antonio between October 2004 and January 2007, in cooperation with artisanal fishermen who simultaneously fished using bottom longlines and gillnets. Target and non-target catch composition, fishing yield, catch size distribution and quality of catch, as well as bycatch of dolphins, sea turtles, and interaction with sea lions were compared between the two fishing methods to assess the profitability of switching fishing gears. Hauls of both gear types deployed simultaneously in the same locations showed similar fish catch composition and catch size with both gears but reduced catch of juvenile fishes in longlines. Bycatch of franciscana in bottom longlines was limited to only one dolphin in three consecutive years of trials, and no direct interaction between turtles and hooks were recorded. The economic analysis showed financially acceptable perspectives under a 5-year scenario. Reducing gillnet effort by switching to bottom longlines appears a practical approach to creating a sustainable fishery that could result in significant mitigation of current bycatch of franciscana dolphins in Argentina. However, implementation requires acceptance and compliance by the artisanal gillnet fishery.

Keywords: artisanal gillnet fishery, dolphins, incidental capture reduction, marine conservation, South Western Atlantic, sustainable fishing

INTRODUCTION

Incidental capture of non-target species during fishing operations (bycatch) is a major source of mortality to many marine animals that form critical parts of marine food webs. It has been identified as the most immediate threat to the survival of many endangered marine mammals, elasmobranches, sea turtles, and seabirds (Lewison et al., 2004a; Werner et al., 2006; Goldsworthy and Page, 2007; Read, 2008; Anderson et al., 2011; FAO, 2018a). Cetacean bycatch is particularly serious given their inherent low reproductive rates, long life span, and later maturation age (Lewison et al., 2004a), all of which limit their ability for rapid population recovery. A number of approaches have been implemented to mitigate cetacean bycatch including changing fishing practices, limiting fishing effort by implementing time-area closures, modifying fishing gear and using technological devices such as acoustic alarms ('pingers') (Hall, 1996). Some progress has been made in mitigating this problem in commercial fisheries located in developed countries (Dawson et al., 2013; Northridge et al., 2017). On the other hand, there are only a few examples from small-scale non-industrial fisheries in developing countries (Mangel et al., 2010; Bielli et al., 2020).

The majority of bycatch of small cetaceans is believed to occur in gillnets (Read et al., 2006). Gillnets, which are one of the most popular fishing gears worldwide (He, 2006) are widely used in small-scale coastal artisanal fisheries because they are relatively inexpensive, require little infrastructure, and can be deployed and retrieved easily on land or using small boats. An approximation of the significance of the artisanal sector using FAO (2018b) information on the global fishing fleet is that 86% of all fishing vessels have a length of 12 m or less and are mostly undecked. Small-scale fisheries are important to local economies, involving more than 90% of the world's fishing workforce, producing about half of global annual fish catches and also providing most of the fish for human consumption in the developing world (Berkes et al., 2001).

Bycatch of franciscana dolphins (*Pontoporia blainvillei)* in artisanal gillnet fisheries has been identified as the primary conservation threat throughout most of its range in Brazil, Uruguay, and Argentina (Reeves et al., 2012). Franciscana reach sexual maturity between 2 and 4 years -one of the earliest age ranges of maturity reported for any cetacean- and have a maximum life span of approximately 20 years (Kasuya and Brownell, 1979; Danilewicz, 2000; Panebianco et al., 2012; Negri et al., 2014). As a result of this short life span and low reproductive potential the species has a limited ability to recover from current high levels of bycatch across its range (Cappozzo et al., 2007; Negri et al., 2012; Prado et al., 2013, 2016; Szephegyi et al., 2015; Cremer et al., 2016). Based on a projected range wide decline of more than 30% over three generations, franciscana is classified as a "vulnerable" species (Reeves et al., 2012).

Four "management areas" have been proposed for the species based on a combination of morphological, ecological, and genetic differentiation; two inhabiting coastal central Brazil, one in southern Brazil and Uruguay, and one in Argentina (Secchi et al., 2003). The species' distribution in Argentine waters is mainly restricted to coastal Buenos Aires Province, where bycatch of franciscana has been estimated at 360–650 individuals per annum (Pérez Macri and Crespo, 1989; Corcuera, 1994; Bordino and Albareda, 2004; Cappozzo et al., 2007; Negri et al., 2012). Franciscana abundance in Argentina has been estimated at about 14,000 individuals (Crespo et al., 2010), with 2.6–4.6% removed each year by gillnets. Annual bycatch rates that exceed ∼2% of a population size are generally considered unsustainable for small cetaceans (Perrin et al., 1994). This estimate of the proportion of the population removed by bycatch is based on the assumption of a single Argentinean population; however, genetic data indicate subpopulation structure in the region (Mendez et al., 2008; Gariboldi et al., 2015) that occur in environmentally distinct areas (Mendez et al., 2010). Although dolphins are considered highly mobile marine animals, franciscana show restricted movement patterns (Bordino et al., 2008; Wells and Bordino, 2013). The identification of subpopulations within a relatively small geographic area further underlines the importance of reducing the impact of bycatch on this species.

The multi-fleet coastal fishery of Northern Buenos Aires targets a coastal demersal association of about 30 fish species called 'variado costero,' using demersal trawls and bottom set gillnets. The gillnet sector of the fishery is a small-scale fishery that is economically vital to local coastal communities (Lagos, 2001; Garcia, 2010). Despite its importance, many aspects of the fishery, including management options, have not been comprehensively studied. The increase in fishing effort by trawlers in this region (Carozza, 2010), a decrease in the average catch size, an increase proportion of juvenile catch in the area (Ruarte and Aubone, 2004; Aubone and Lagos, 2007; Carozza and Hernandez, 2007), and the intensification of gillnet effort are directly linked to an increase in both bycatch risk and competition with marine mammals for prey species (Crespo et al., 1994; Weiskel et al., 2002).

A range of strategies to reduce franciscana bycatch in the gillnet sector of the fishery have been trialed, including the use of acoustic deterrent devices (ADDs) (Bordino et al., 2002, 2004) and testing the effectiveness of chemically modified gillnets (Bordino et al., 2013). The use of time-area closures to mitigate bycatch in the fishery are not considered a viable management option owing to the anticipated lack of infrastructure and support for monitoring and enforcement.

While the use of ADDs (pingers) resulted in a reduction in franciscana bycatch rates (Bordino et al., 2002, 2004), widespread adoption of these devices is hampered by the financial cost of purchase, unit maintenance and enforcement. There are also concerns that the repeated use of ADDs may result in habituation and/or habitat exclusion (Dawson et al., 2013). As ADDs are the only bycatch reduction technology so far shown to be effective with franciscana, additional investigation on how to reduce their cost and ruling out any unintended ecological consequences should be continued. Even though recent studies have shown that bycatch of small cetaceans can be reduced by LED devices (Bielli et al., 2020), this technology has never been tested in the area due to the characteristic murky waters. In the meantime, it is important to evaluate other fishing techniques that may offer incentives for fishermen to adopt them.

The following study was conducted to investigate if changing fishing gears in a small-scale artisanal fishery could reduce franciscana bycatch rates while maintaining comparable economic returns. Specifically we recorded total catch, comparative bycatch rates and commercial catch between gillnet and bottom longline operations fished concurrently, including target and non-target catch composition, fishing yield, catch size frequency distribution, discards, and quality of catch. An economic analysis of the fishery was performed to assess the potential profitability of switching fishing gears.

MATERIALS AND METHODS

Study Areas

The experiment was conducted in the artisanal gillnet fisheries of Bahía Samborombón (BSB) and Cabo San Antonio (CSA), in northern Buenos Aires Province (**Figure 1**). The BSB and CSA are important reproductive and spawning areas for several fish species (Lasta and Acha, 1996; Acha et al., 1999; Acha and Macchi, 2000) but vary in both static and dynamic environmental features. Usually, around 80 small artisanal gillnet boats and 120 trawl vessels have licenses to operate in these two areas. Gillnet fishing is conducted using small inflatable and fiberglass boats 5–8 m in length, powered by 40–120 HP outboard engines, and operating at 0.5 to 7 km from the coast in depths ranging from 3 to 12 m. Gillnets consist of 50 m panels of white monofilament (diameter 0.50–0.60 mm) with a height of 3–6 m and a hanging ratio of 0.25. Stretched mesh sizes range from 90 to 140 mm. In general, each fisherman deploys 400 m of gillnets at a time consisting of 100–200 m strings separated from each other. Occasionally, some fishermen use up to 2000 m of gillnets in response to competition with local trawlers for what they consider an overfished resource. While approximately half the vessels fish year round, the majority of fishing effort occurs between September and April. At BSB there is a natural harbor (San Clemente del Tuyú), with fishing activity restricted by tidal state, while at CSA boats are launched from the beach using trailers and 4 × 4 vehicles. In both areas, the fishery is highly dependent on the wind especially in the austral spring and summer limiting the operational days (Lagos, 2001; Garcia, 2010). The two main target species of the fishery are Whitemouth croaker (*Micropogonias furnieri*) and Stripped weakfish (*Cynoscion guatucupa*). Landed catches are sold to local restaurants and tourists during the summer season, or to intermediates for exportation to Asia, Africa, Europe, and Brazil.

Fishing Gear Evaluation

The experiment was conducted between October 2004 – February 2005 at both study areas, and between October 2005 – February 2006 and November 2006 – January 2007 at BSB only due to logistical problems at CSA. The first fishing period involved ten fishermen (five fishermen at each study area) simultaneously fishing with bottom longlines and gillnets, while the second and third fishing periods involved five fishermen at BSB only. Fishermen used standardized fishing gear provided to them for the experiment. Gillnets were new and of the same dimensions commonly used by fishermen, all rigged identically, with 120 mm mesh size and monofilament 0.6 mm diameter. Each fisherman used one string of 100 m length and 3.5 m high. The experimental bottom longline consisted of 180 m of 5 mm ground line and 2 mm flat nylon drop lines (snoods) attached with knots approximately every fathom (1.8 m). Buoys were placed at each end and along the ground line every 27 snoods, and 1000 g of weight was placed every 14 snoods. Additional weight (100 g of tubular lead units) was placed every 6–7 snoods corresponding with buoy positions to control the depth of gear operation. Each bottom longline was rigged with 80 medium size J-shaped hooks (*Mustad* 2330 N°7). The design of the experimental bottom longlines was previously discussed with local fishermen and is shown in **Figure 2**. Each fishermen fished with one of these bottom longline units in addition to the gillnet string. During the first trial, hooks were baited with approximately 10–15 g pieces of fresh Brazilian menhaden (*Brevoortia aurea*), Argentine conger (*Conger orbignyanus*), or frozen Argentine shortfin squid (*Illex argentinus*), with a single bait type randomly assigned to each haul. Brazilian menhaden and Argentine conger, which are abundant and considered non-commercial species locally, were caught using the experimental gillnets and longlines, while Argentine shortfin squid was provided to fishermen. Gillnet strings and bottom longlines deployed by each boat were anchored in close proximity (approximately 100 m apart), in depths ranging between 4–12 m. Each fishing boat carried an independent observer who was rotated among boats throughout the course of the experiment. Observers recorded geographic position, soak time, type of bait, and biomass of fish caught by each fishing gear, as well as environmental conditions and bycatch.

The presence of and/or any interaction between marine mammals, seabirds and sea turtles with the fishing gear were also recorded at the beginning and at the end of each fishing haul, as well as the condition of every baited hook and the presence of any damage to gillnets. After hauling, the daily catch from gillnets and bottom longlines were separated by species and weighed. Discards of commercial fish in unsellable condition and non-commercial fish species were also recorded.

An index of commercial fish species occurrence (*Isp*) was calculated by the equation:

$$Isp = (n_i/N) \times 100 \tag{1}$$

Where,

n_i: number of times any given commercial species was caught by each fishing gear; and

N: total number of fishing sets per gear.

For each fishing gear type, all commercial species with $Isp \geq 50\%$ were considered to be common species, while $Isp > 75\%$ and $Isp < 25\%$ were considered to be very common and rare species, respectively.

Total length (*TL*), measured to the nearest half centimeter was recorded for a random subsample of commercial fish species from each fishing haul. The selectivity of each fishing gear type was calculated from the catch size distribution by species, assuming that the selection range of the fishing gear

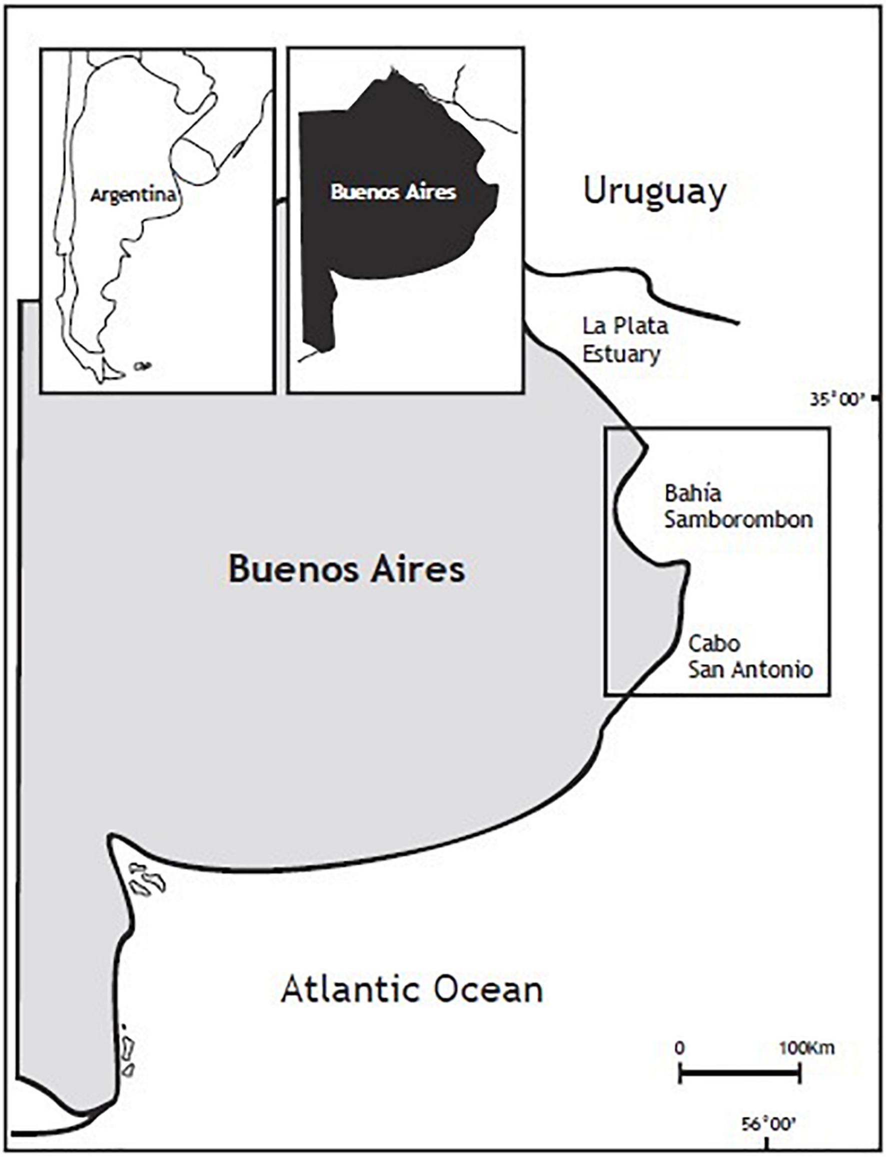

FIGURE 1 | Location of the study areas, Bahía Samborombón (BSB) and Cabo San Antonio (CSA).

is narrower than the stock size distribution. Thus, selectivity presented from *TL* frequencies of the catch, ignoring the stock size distribution, should be interpreted with caution as it provides only a rough estimate of the selection curve, and is therefore useful only in the context of this experiment comparing gear types. The *TL* frequency distributions by species were compared between fishing gears using Kolmogorov–Smirnov test (Siegel and Castellan, 1988).

The quality of the commercial fish catch was graded into four classes: (1) Fish in perfect condition (red gills or alive), (2) Fish in good condition (pink gills), (3) Fish partially deteriorated due to predation by sea lions, and (4) Fish in bad condition (gray gills and/or extensively depredated). Fish graded as class 3 or 4 were considered discards.

Fish catch per unit effort (CPUE) was used to assess the influence of factors such as fisherman, area and year on fishing yield. For gillnets and bottom longlines, the CPUE was calculated as kg of fish/m^2/hr, and kg of fish/80 hooks/hr, respectively, for each set. The relative fishing performance between fishing gears was defined by CPUE of bottom longlines/CPUE of gillnets as a way to evaluate if any gear was more efficient than other throughout the trials. The number of bottom longline hooks required to obtain the average catch of 100 m of gillnets was estimated using the ratio between fish biomass (kg) caught per gillnet haul/fish biomass (kg) caught per bottom longline haul.

The effect of bait type on the relative fishing performance of bottom longlines was analyzed using the CPUE and bait loss rate. The bait loss rate was calculated as the number of unbaited

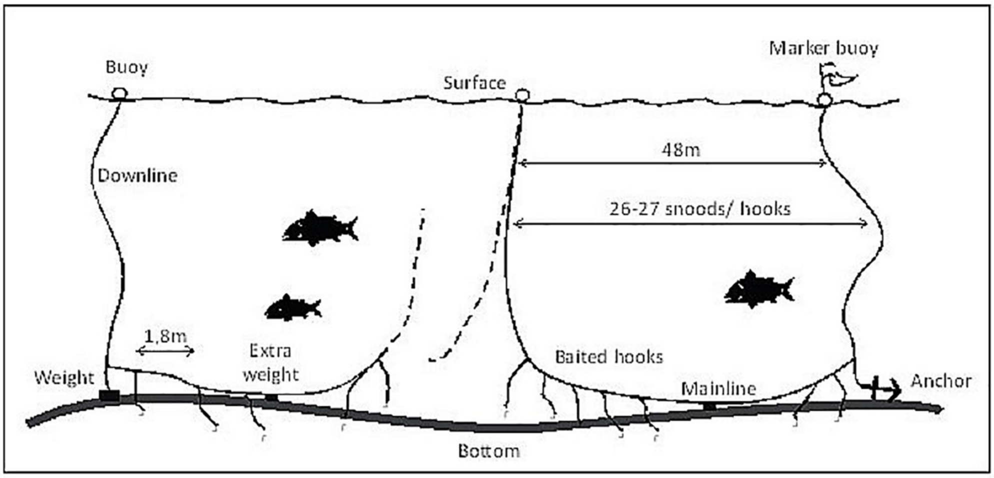

FIGURE 2 | Bottom longline experimental design showing both ends.

hooks/number of baited hooks with no catch at the end of each haul. Due to the smaller sample size from CSA, analysis of the effect of soak time on catch rates in gillnets and bottom longlines was restricted to data collected at BSB.

As units of fishing effort were different for the two gear types, the bycatch per unit effort (BPUE) for dolphins, sea turtles and seabirds were calculated as the number of individuals captured per total biomass (kg) of commercial fish catch for each fishing gear type. To allow comparison with previous studies in the fishery BPUE*dolphins* in gillnets was recalculated as the number of bycaught dolphins per km net^{-1} hr^{-1}.

South American sea lion (*Otaria byronia*) depredation on both gear types was calculated as the number of attacks/kg of total fish catch (APUE). An attack was recorded when a fisherman determined, based on experience, that observed damage to fish and/or the fishing gear had been caused by sea lions, as opposed to sharks. Multiple attacks within a single fishing operations were assumed to be the same event.

Independence among fishing gear hauls was assumed for both gears. The daily CPUE in gillnets and bottom longlines were analyzed with Kruskal–Wallis non-parametric test once assumption for normality was not met (Cohen and Fowler, 1990).

Economic Analysis of the Fishery

To determine the economic effect of switching from gillnets to bottom longlines economic analyses were conducted using data collected through interviews from 21 local artisanal fishermen at BSB (5 of them were also involved in the trials). Information provided by the fishermen included the following: cost of manufacturing and maintaining fishing gear, depreciation of fishing gear, fuel consumption and labor (usually either fixed or as a percentage of daily fishing production). The average of these costs were used for the analysis. All costs were valued in kg of fish as this is the way that many local fishermen estimate their costs and using this metric avoids expressing any monetary value that would be highly variable as affected by local economic conditions. The average commercial fish catch rate for gillnets recorded during this experiment was compared to a combination of historical catch rates obtained by interviews with local artisanal fishermen and records from the Coast Guard. The analysis assumed that the fleet is homogeneous in terms of average cost-per-unit-effort, and that market price is not affected by landing volumes. Effort was set as an average of 80 operational days per fishing season. Due to the smaller sample size, economic analysis was not conducted for CSA.

The total gross revenue (*TGR*) was estimated as the average seasonal biomass of commercial fish caught per fisherman. The operational cost (*OC*) included fuel gas and oil, boat maintenance, insurance, taxes, and fishing gear and manufacturing. Gross value added (*GVA*) was calculated as the *TGR* minus *OC*. Labor cost (*LC*) was calculated as crew personnel cost including hooking bait and cleaning. Gross cash flow (*GCF*) was calculated as *GVA* minus *LC*. The economic profit (*EPR*) was calculated as the *GCF* minus depreciation (*D*) of manufactured fishing gear. The gear depreciated at 100% over the duration of the study period for gillnets, and was calculated at 25% of the initial cost for bottom longlines. It was assumed that *D* was not a function of maintenance (*M*). Depreciation on boat and engine and the interest on owner's capital were not considered. The profit margin (*PRM*) and the return on investment (*ROI*) were defined, respectively as:

$$PRM = EPR \times 100/TGR \tag{2}$$

$$ROI = EPRx100/Totalassets \tag{3}$$

where total assets represent the value of the initial investment including a used boat with a 60–90 HP outboard engine, fishing gear (considering the comparative gillnet and bottom longline effort determined during this experiment), annual insurance and taxes, cost of fuel, labor crew, and bait cleaning hooks. The economic performance, based on the commercial fish catch rates obtained during the trials at BSB was estimated for three scenarios: 100% gillnet effort, 100% of bottom longline effort, and a hypothetical combination of 50% gillnet and 50% bottom longline effort.

An additional comparative economic analysis was conducted using a residual approach, which allows evaluation from an investor's perspective to decide if a project is financially acceptable or not. The evaluation was made using a financial model considering the risk of investment by the variables: (a) biomass of commercial fish caught for sale, and (b) their local market price. This risk was analyzed through the Hertz Simulation Model (Hertz, 1964). The analysis used the Incremental Internal Profitability Rate (IRP) because the scale of the investment is dependent on the type of fishing gear used. The Hertz Simulation Model is one of the most frequently used in financial evaluations, providing a probability function to the IRP, which is built from the probability functions of the aleatory variables. The model selects the variables of higher influence associated with income, market, investment and costs, and assigns them probabilities according occurrence to ranges of variation. If these ranges are combined, different possible alternatives are established which will have a return rate and a value determined by the associated probability. The result is a function of probability for IRP, and the model will then produce the probability of reaching or overcoming a determined IRP. To determine if the project is viable or not, it is necessary to define a minimum acceptable revenue level. The advantage of this model is that investors may define their own minimum expectations for profitability. In this analysis, the model used the nominal monthly interest for deposits at the *Banco Central de la República Argentina*[1] in 2007, which was 0.89%. Although this is the rate used for the current study, it will not be static and may not reflect the actual interest rate that should be used at a given point in time. Different scenarios for 1 and 5 years were tested, considering initial investment for boat and type of fishing gear, as well as basic expenses such as fuel, oil, crew, maintenance, insurance and taxes. To define the combination of the independent aleatory variables, probabilities were assigned to the total fish catch and the price range assuming a normal distribution. The following assumptions were made in the model: (1) All catch of commercially targeted species is sold in the market at homogeneous price range; (2) Fish catch and quality are the same for both gear types; (3) The price range is distributed normally in seven categories represented by 3.5%, 10%, 19%, 35%, 19%, 10%, and 3.5%; (4) The costs for crew, fuel oil and boat maintenance is independent of the fishing gear used; (5) There is a minimum of 80 fishing days per fishing season regardless of gear type used; (6) Each fishing gear is hauled only once a day; (7) The gillnetting fleet has a common cost structure, and; (8) The skill of skippers and crew is homogeneous and fishing capacity can be extrapolated to the fleet.

Ethical Approval

The experiment described in this paper was carried out in accordance with the current laws of the country regarding artisanal fisheries practice. Fish were taken from the fishing gears that artisanal fishermen fish legally on a daily basis. Data was obtained by counting and measuring normal fish catch and no manipulation of live animals was done. No additional permits were required but there was a consensus from the researchers to stop the project if the new fishing gear had higher rates of bycatch than the normal rate. Information regarding income, expenses and profit for performing the economic analysis included in this manuscript was obtained through a series of interviews to the fishermen which were carried out by the local conservation NGO, AquaMarina.

AquaMarina used a semi-structured interview format with fishermen who also participated in the fieldwork. None of the organizations that funded this work required an ethics review for these interviews. Nevertheless, AquaMarina adopted some standard protocols cognizant of important ethical considerations. Interviews were carried out by students hired by the project who received orientation about its objectives and interview protocols. Consent was received from all participants prior to conducting the interviews. The interviews were performed individually to each fisherman at the beach sites were they usually launch their boats on a daily basis for going out fishing. Prior to each interview, the objectives of the project were restated for the fishermen who were afforded the opportunity to ask any clarifying questions about the study or interviews. The names of respondents were recorded but their responses kept anonymous. There was no commercial or other use for the information recorded except for this study. That being the case, the project was carried on normally and thus exempt from ethical approval.

RESULTS

Fishing Gear Evaluation

Locations of gillnet and bottom longline hauls are shown in **Figure 3**. Mean depth of fishing operations was 5.2 $\pm$ 2.1 m at BSB, and 7.2 $\pm$ 2.3 m at CSA. The mean soak time (mean $\pm$ standard deviation) at BSB was 23.3 $\pm$ 2.9 h and 3.1 $\pm$ 0.93 h for gillnet and bottom longline, respectively, and 20.8 $\pm$ 2.1 h and 3.0 $\pm$ 0.89 h, for gillnet and bottom longlines, respectively, at CSA. Although in some cases both gears at both areas had soak times of up to 75 h due to adverse weather conditions that prevented fishermen from retrieving gear earlier, these hauls were not considered for the analysis. There was no significant difference in depth or soak time for each gear in gillnets or bottom longlines throughout the three consecutive fishing seasons at BSB ($P > 0.05$, Kruskal–Wallis test, pair-wise comparison with Dunn's test). A summary of fish catch by fishing gear type is presented in **Tables 1**, **2**. In total, 13 fish species were caught with both fishing gear types at the two study areas, nine of which were local commercial species, with eight and seven commercial species caught in gillnets and bottom longlines, respectively. Of the very common (*Isp* $> 75\%$) and common species (*Isp* $\geq 50\%$) being targeted, gillnets and bottom longlines caught three and four species, respectively, that dominated the catches of both gear types (**Table 3**). The two

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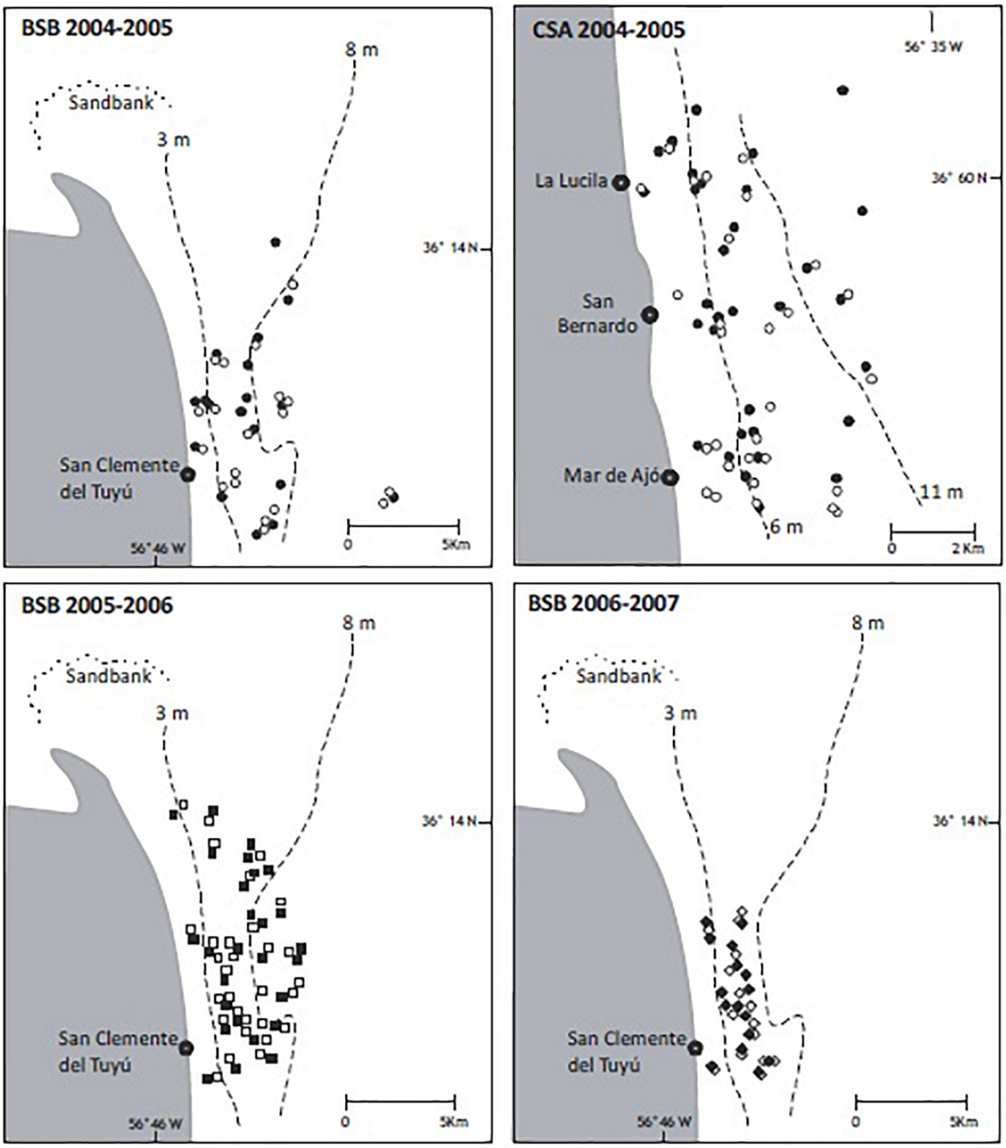

FIGURE 3 | Location of gillnets (black) and experimental bottom longlines (white) hauls in Bahía Samborombón (BSB) and Cabo San Antonio (CSA).

main target species were Stripped weakfish and Whitemouth croaker, which represented 80% and 52% of the total biomass caught in the pooled study areas (excluding discards), in gillnets and bottom longlines, respectively. Parona leatherjack (*Parona signata*) and Mullet (*Mugil liza*) were caught only in gillnets, and Argentine conger (*Conger orbignyanus*) was only caught in bottom longlines and was also the most frequently caught species at CSA. Among all commercial species with the highest value in the local market, Brazilian codling (*Urophycis brasiliensis*) and Catfish (*Genidens* sp.) represented only 1.5% and 31% of the total catch (excluding discards) in gillnets and bottom longlines, respectively. Mullet and Catfish were exclusively caught in BSB, while Patagonian smooth-hound (*Mustelus schmitti*) was only caught in CSA.

No significant differences in the CPUE among fishermen with gillnets or bottom longlines were found at BSB or CSA ($P > 0.05$, Kruskal–Wallis test, Pair-wise comparison with Dunn's test), and therefore the CPUEs were combined at each study area. Preliminary data analysis for the first trial (2004–2005) showed a significantly higher CPUE for commercial species with gillnets at CSA than BSB, and with bottom longlines at BSB than CSA ($P < 0.001$, Mann–Whitney test). These differences were confirmed by the relative fishing performance of bottom longlines to gillnets which was 19 at BSB and 8.5 at CSA during the first trial. This relative fishing performance of bottom longlines was 14.3 and 13.8 in the second and third trials, respectively, at BSB. Considering only the data set from BSB, the CPUE in gillnets or bottom longlines were similar during the

TABLE 1 | Summary of commercial fish catch and discard in gillnet hauls.

Gillnets	Total FE ($m^2 \times h$)	Targeted species		Catch (kg)	% (kg)	CPUE
BSB 04–05	1262532	*Cynoscion guatucupa*		4815	62.8	3.8×10^{-3}
(*N* = 135)		*Micropogonias furnieri*		827	10.8	6.5×10^{-4}
		Parona signata		569	7.4	4.5×10^{-4}
		Genidens sp.		177	2.3	1.4×10^{-4}
		Mugil liza		102	1.3	8.1×10^{-5}
		Urophycis brasiliensis		31	0.4	2.5×10^{-5}
			Subtotal	6521		5.2×10^{-3}
		Discard		1137	14.8	9.0×10^{-4}
			Total	7658		
CSA 04–05	882185	*Micropogonias furnieri*		2623	34.7	3.0×10^{-3}
(*N* = 138)		*Cynoscion guatucupa*		1931	25.5	2.2×10^{-3}
		Mustelus schmitti		1605	21.2	1.8×10^{-3}
		Parona signata		250	3.3	2.8×10^{-4}
			Subtotal	6409		7.3×10^{-3}
		Discard		1152	15.2	1.3×10^{-3}
			Total	7561		
BSB 05–06	1001204	*Cynoscion guatucupa*		3822	57.1	3.8×10^{-3}
(*N* = 118)		*Micropogonias furnieri*		807	12	8.1×10^{-4}
		Parona signata		622	9.3	6.2×10^{-4}
		Macrodon ancylodon		286	4.3	2.9×10^{-4}
		Urophycis brasiliensis		68	1	6.8×10^{-5}
			Subtotal	5605		5.6×10^{-3}
		Discard		1089	16.3	1.1×10^{-3}
			Total	6694		
BSB 06–07	412121	*Micropogonias furnieri*		1540	36	3.7×10^{-3}
(*N* = 61)		*Cynoscion guatucupa*		1382	32.3	3.3×10^{-3}
		Macrodon ancylodon		411	9.6	1.0×10^{-3}
		Parona signata		186	4.3	4.5×10^{-4}
		Urophycis brasiliensis		54	1.2	1.3×10^{-4}
			Subtotal	3573		8.7×10^{-3}
		Discard		706	16.5	1.7×10^{-3}
			Total	4279		

FE, fishing effort; N, number of hauls.

three trials ($P > 0.05$, Kruskal–Wallis test, pair-wise comparison with Dunn's test), although the CPUE was more variable in bottom longlines than gillnets (**Figure 4**). The mean daily catch rate of bottom longlines at CSA was 0.19 ± 0.06 kg/hook, and 0.31 ± 0.05 kg/hook at BSB.

The range of catch sizes of the two main targeted species, Whitemouth croaker and Stripped weakfish, overlapped with no evidence of differences in size frequency distribution between BSB and CSA in either gillnets or bottom longlines ($P < 0.05$, Mann–Whitney test). The pooled total length frequency distributions for gillnets and bottom longlines were similar for each paired species caught by both fishing gear types ($P > 0.001$, Kolmogorov–Smirnov test, **Figure 5**). Six percent of Whitemouth croaker and 2% of Stripped weakfish caught in gillnets were below the minimum allowable catch sizes, as were 3% of Whitemouth croaker and 2% of Stripped weakfish caught by longlines in pooled study areas. Over 85% of the commercial fish catch was of the two highest quality classifications (perfect condition/alive and good condition) when caught by gillnets or bottom longlines, although bottom longlines had a higher percentage of fish hauled in perfect condition/alive (**Table 4**).

There was a significant difference in both the CPUE and catch composition in bottom longlines in relation to the type of bait used ($P < 0.05$, Kruskal–Wallis test, $P < 0.001$, Kolmogorov–Smirnov test). The results of the first trial at BSB and CSA showed that the use of Brazilian menhaden resulted in higher CPUE and a lower bait loss rate than the other two baits used (**Figure 6**). Consequently, Brazilian menhaden was used as the sole bait type during the subsequent trials.

Discards comprised commercial fish in poor condition or depredated by Southern sea lions or unidentified sharks. Within the pooled study areas, average discard in gillnets represented 15.7% of the total catch, while in bottom longlines it was 7.5%. From the pooled data set from BSB and CSA, sea lion interaction in gillnets and bottom longlines was recorded in 36% and 19% of hauls, respectively, with average depredation by sea lions occurred on 3% and 5% of commercial fish catch in gillnets and bottom longlines, respectively. The most frequently caught non-commercial species in gillnets was Brazilian menhaden,

TABLE 2 | Summary of commercial fish catch and discard in bottom longline hauls.

Bottom longlines	Total FE (hooks × h)	Targeted species		Catch (kg)	% (kg)	CPUE
BSB 04–05	32506	*Cynoscion guatucupa*		1058	31.1	0.032
(*N* = 138)		*Micropogonias furnieri*		692	20.4	0.021
		Genidens sp.		666	19.6	0.020
		Urophycis brasiliensis		434	12.8	0.013
		Conger orbignyanus		368	10.8	0.011
			Subtotal	3218		0.099
		Discard		178	5.2	0.005
			Total	3396		
CSA 04–05	28687	*Conger orbignyanus*		540	27.1	0.019
(*N* = 121)		*Cynoscion guatucupa*		484	24.3	0.017
		Urophycis brasiliensis		384	19.3	0.013
		Micropogonias furnieri		204	10.2	0.007
		Macrodon ancylodon		158	7.9	0.005
		Mustelus schmitti		95	4.8	0.003
			Subtotal	1865		0.062
		Discard		126	6.3	0.004
			Total	1991		
BSB 05–06	30101	*Cynoscion guatucupa*		744	29.8	0.025
(*N* = 112)		*Micropogonias furnieri*		594	23.8	0.020
		Urophycis brasiliensis		374	15	0.012
		Genidens sp.		324	13	0.010
		Conger orbignyanus		241	9.6	0.008
			Subtotal	2277		0.08
		Discard		221	16.3	0.007
			Total	2498		
BSB 06–07	11174	*Micropogonias furnieri*		369	23.9	0.033
(*N* = 44)		*Cynoscion guatucupa*		358	23.2	0.032
		Urophycis brasiliensis		270	17.5	0.024
		Genidens sp.		211	9.6	0.019
		Macrodon ancylodon		148	1.2	0.013
			Subtotal	1356		0.12
		Discard		186	12	0.017
			Total	1542		

FE, fishing effort; N, number of hauls.

while in bottom longlines were Skate (*Raja flavirostris*), Angel shark (*Squatina argentina*), and Brazilian flathead (*Percophis brasiliensis*). Although these three species are occasionally sold as low value species, there was no local market for them at the time the experiment was conducted, and as a result catches of these species were excluded from the economic analysis. Three species of gastropods, Black volute (*Adelomelon brasiliana*), Fine volute (*Zidona dufresneyi*), and Rapa whelk (*Rapana venosa*) were also occasionally caught in bottom longlines but not considered in the analysis.

The average commercial fish catch in gillnets, excluding discards, was 48.3 kg/haul (BSB 04–05), 46.4 kg/haul (CSA 04–05), 47.5 kg/haul (BSB 05–06), and 58.6 kg/haul (BSB 06–07). The average commercial fish catch in bottom longlines, excluding discards, was 23.3 kg/haul (BSB 04–05), 15.4 kg/haul (CSA 04–05), 20.3 kg/haul (BSB 05–06), and 30.8 kg/haul (BSB 06–07). For the scale of this experiment, the relationship of commercial fish catch between gillnets and bottom longlines was 2.0 (BSB 04–05), 3.4 (CSA 04–05), 2.4 (BSB 05–06) and 2.6 (BSB 06–07), indicating that the bottom longline fishing effort would be roughly increased by 2.5 times to have equivalent catches as gillnets. A conservative estimate would be to use a minimum of 200 hooks to catch similar amount of commercial fish than 100 m of gillnets at BSB. Based on the catch rates recorded, it is then expected that each fisherman operating any daily string combination of 400 m gillnets would catch around 15,200–18,752 kg assuming 80 workable days per fishing season at BSB. This result is based on a linear relationship between catch abundance and length of longline which may not necessarily always be the case given variable target fish distributions and potentially other factors.

Comparative dolphin, sea turtle, and sea lion interactions with each gear type are shown in **Table 5**. No interactions with seabirds were recorded for either gear during the trials. In total, 85 dolphins were bycaught in 452 gillnet hauls from 71 bycatch events compared to one dolphin bycaught from 415 longline hauls. The dolphin was hooked by the pectoral fin

TABLE 3 | Combined species occurrence index (%) showing species most commonly caught by both fishing gear types.

Targeted species			
Common name	**Scientific name**	**Gillnets (%)**	**Bottom longlines (%)**
Stripped weakfish	*Cynoscion guatucupa*	86.6	58.5
Whitemouth croaker	*Micropogonias furnieri*	72.6	56.5
Parona leatherjack	*Parona signata*	59.2	4.2
Brazilian codling	*Urophycis brasiliensis*	37.5	50.5
Mullet	*Mugil liza*	26.4	0
Catfish	*Genidens* sp.	15.2	53
Patagonian smooth-hound	*Mustelus schmitti*	8.8	30
Conger	*Conger orbygnianus*	0	36.3
King weakfish	*Macrodon ancylodon*	4.1	1.8

The species with ≥50% were classified as common species.

and entangled in the snood and mainline and necropsy results showed that this individual had drowned. Seventy-seven green turtles (*Chelonia mydas*), three loggerhead (*Caretta caretta*), and two leatherback turtles (*Dermochelys coriacea*) were incidentally entangled in gillnets. Seven green turtles were released alive, while the remaining 75 turtles were found dead. Sea turtle entanglements in bottom longlines involved three green turtles and one leatherback, with all individuals caught either in the main or anchoring line. Two of the three green sea turtles were found alive and subsequently released, while the leatherback was suspected to have died before the entanglement, which was later confirmed by necropsy. No direct interaction between turtles and hooks were recorded.

Economic Analysis of the Fishery

Information on the most relevant cost indicators and total assets for artisanal fishing operations in BSB is shown in **Table 5**. A preliminary analysis showed that the commercial fish catch rate in gillnets during this experiment was significantly lower than the historical commercial fish catch rate estimated by combination of interviews with fishermen and records from the Coast Guard (Mann–Whitney test, $P < 0.05$). The estimated fish catch rates and probabilities assigned for the economic analysis are shown in **Table 6**. The analysis of the economic indicators shows that the profit margin (*EPR*), as well as the return on investment (*ROI*), is higher when operating 400 m of gillnets than 800 hooks in bottom longlines (**Table 7**). When using 50% of gillnet fishing effort (200 m) in combination with bottom longlines (400 hooks) simultaneously, the *ROI* would be about 8–10% lower than using gillnets alone when considering the estimated catch rates during this experiment and the combined data from interviews, respectively. The IRP was calculated from a total of 21,875 combinations of the aleatory variables: fish catch, and price. These combinations give IRP values between 5% and 27.8%, and between −33% and 29.8% for 5- and 1-year scenarios, respectively (**Figure 7**). These values indicate acceptable perspectives for economic and financial return of the investment under a 5-year scenario in 100% of the cases. However, for a 1-year scenario the values are acceptable in 30.2% of the cases only.

DISCUSSION

In order to evaluate the effectiveness of the bottom longlines as a potential alternative fishing gear, we discuss its fishing performance, bycatch mitigation effectiveness and economic implications in contrast with gillnets.

Fishing Performance

The process of a fish encountering a fishing gear is different for gillnets and bottom longlines as it depends on random movements and swimming activity of fish and visual/olfactory stimuli from bait. The fishing gear's efficiency can vary with time, location, environmental factors and the presence of competitors, natural prey, or predators that influence fish behavior. Water temperature, light level, current velocity, turbidity, ambient prey density and intra- and inter-specific competition are likely to have the largest effects on fish catchability in the performance of baited fishing gear (Stoner, 2004). Daily activity rhythms, feeding motivation, sensory and locomotory abilities of fish also play an important role in the effectiveness of baited fishing gear.

Bait type is considered one of the most important factors that determine effectiveness in longlines. Bait choice involves trade-offs between quality, effectiveness and cost. In both study areas, Brazilian menhaden bait was effective at catching the two main targeted species, producing the highest catch diversity of commercial species and showed less bait loss rate. As it is usually caught as a discard species in gillnets, the use of Brazilian menhaden does not represent an additional cost for local fishermen. Although during the first trial there were bottom longline hauls baited with less effective bait as Argentine conger, no significant differences were observed in the CPUE throughout the second and third trials at BSB. Argentine conger presented a relative low catch rate and a higher bait loss rate which could be related to rapid deterioration over time and a greater probability of attacks by scavengers. The Argentine squid was a highly effective bait, especially for Whitemouth croaker, but the relative high cost limits its utility in a small-scale artisanal fishery.

Selectivity in bottom longlines can be affected by variables such as the gear design, type of hook, hook size, fishing operation and type and size of bait (Løkkeborg and Bjordal, 1992). Fish behavior and morphology will also directly affect the selectivity of fishing gears, if hook size is the sole factor determining selectivity then the selection curve should be sigmoidal in shape (Hovgard and Lassen, 2000). A more thorough analysis of the selectivity of different fishing gears is beyond the scope of this study. However, the analysis of the catch size frequency is useful in determining the potential efficacy of experimental bottom longlines to catch appropriate fish size. The catch size frequencies in bottom longlines were not described by a unimodal curve, except for the Brazilian codling, indicating that more than one factor was likely involved in selectivity during our experiment.

The CPUE recorded in relation to soak time in bottom longlines suggests that the most effective haul duration was

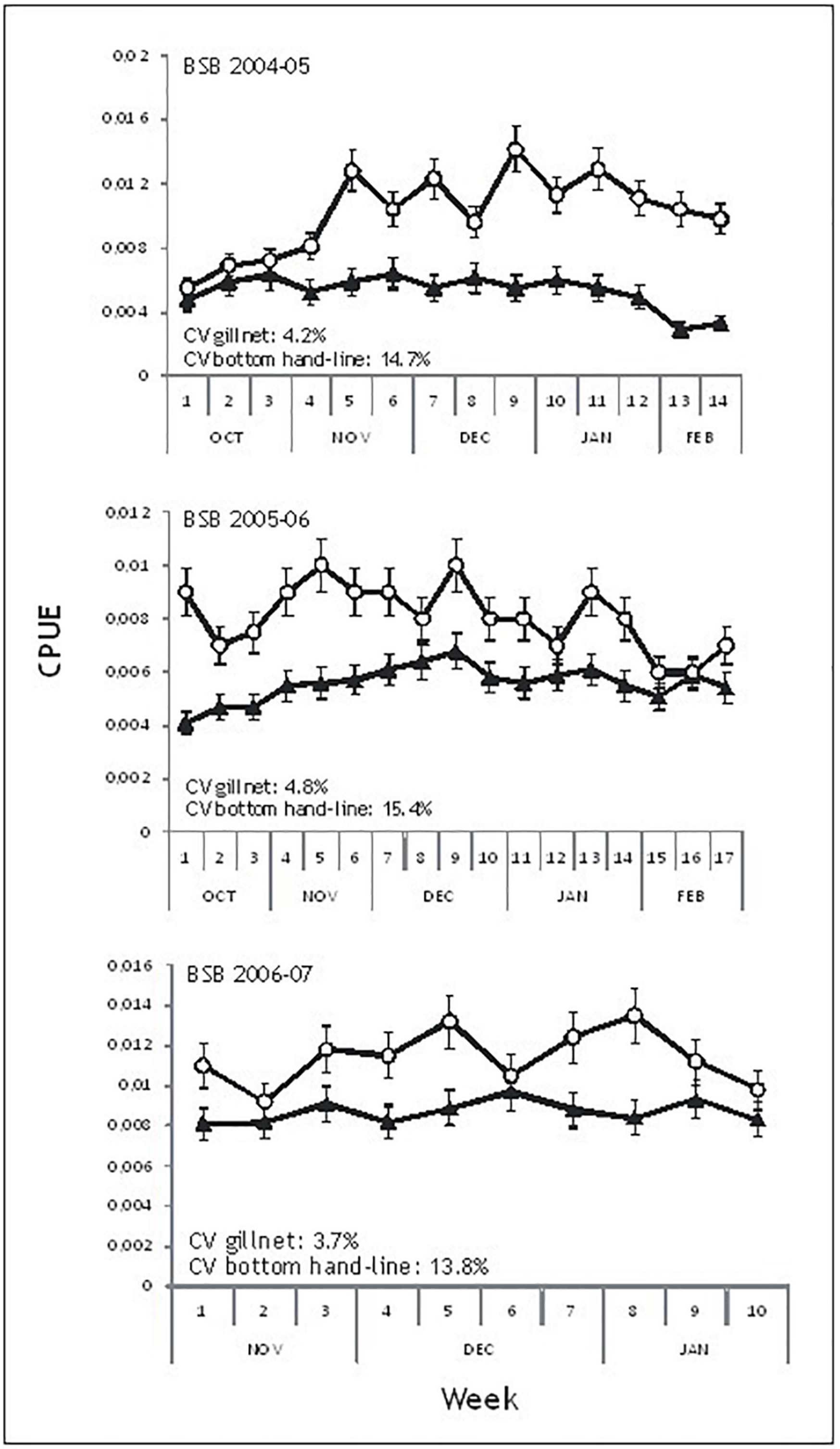

FIGURE 4 | Variability of CPUE and SE in gillnets (black triangle) and bottom longlines (white circle) in BSB. THE CPUE for bottom longlines is expressed as $CPUE^{-1}$.

around 3 h. The maximum number of fish that can be caught by a longline is directly related to the number of hooks available. Over time, as an increasing number of hooks are occupied the fishing power of the gear is reduced. This decrease in fishing power can be described by an exponential decay model (Rotschild, 1967), and may also decline over time if bait is

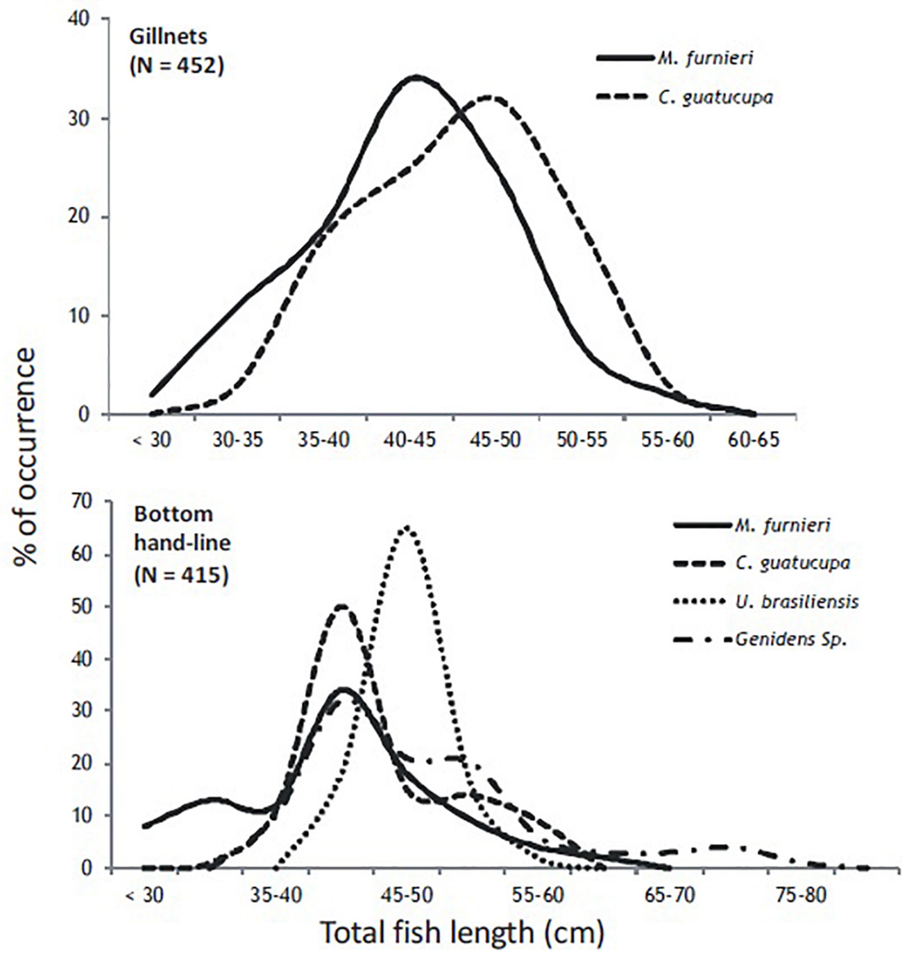

FIGURE 5 | Pooled fish catch size length (TL) distributions for commercial fish species caught with both fishing gears in BSB and CSA.

TABLE 4 | Summary of quality of commercial fish catch in relation to fishing gear.

	Gillnets		**Bottom longlines**	
Fish quality	**BSB kg (%)**	**CSA Kg (%)**	**BSB Kg (%)**	**CSA Kg (%)**
1	2794 (17.8)	1070 (16.7)	4158 (60.7)	809 (43.4)
2	1105 (70.4)	4685 (73.1)	2172 (31.7)	973 (52.2)
3	597 (3.8)	314 (4.9)	144 (2.1)	30 (1.6)
4	1256 (8)	340 (5.3)	377 (5.5)	52 (2.8)
Total	15699	6409	6851	1865

1 = perfect condition; 2 = good condition; 3 = some predation; 4 = gray gills or extensive predation. See text for more details.

removed from the hooks. Additionally, it has been found that the release rate of attractants from baits is initially high and declines rapidly, affecting effectiveness throughout time (Løkkeborg, 1990; Furevik and Løkkeborg, 1994). The durability in water and quality of the bait used may allow local fishermen to operate more than one haul a day, potentially increasing their catches with bottom longlines.

The experimental design of the longline gear could be further tested to see if the fish catch rate is improved. Although previous studies have shown that monofilament ground lines have many advantages (Bjordal, 1989; Sainsbury, 1996), the strong currents in this study area require a multifilament ground line for a better handling and fishing operation unless boats are equipped with winches. In many fisheries, circle hooks have proven to be more effective than traditional "J" shaped hooks (Quinn et al., 1985; Bjordal, 1989; Bolten and Bjordal, 2005; Kersteter and Graves, 2006), probably due to a lower escape rate of hooked fish. Circle hooks have also been shown to reduce the bycatch and increase the post-hook survival rates of sea turtles in some fisheries (Cooke and Suski, 2004; Watson et al., 2005; Minami et al., 2006: Read, 2007; Sales et al., 2010). However, their use might increase the catch of sharks and rays in pelagic and coastal fisheries (Afonso et al., 2011), including currently threatened species such as the Patagonian smooth-hound currently endangered (Massa et al., 2006). Increasing bottom longline effort by implementing 800 hooks to make it comparable to 400 m of gillnets does not seem to bring an issue for local fishermen as long as it results in an increased or comparable revenue to that from gillnets.

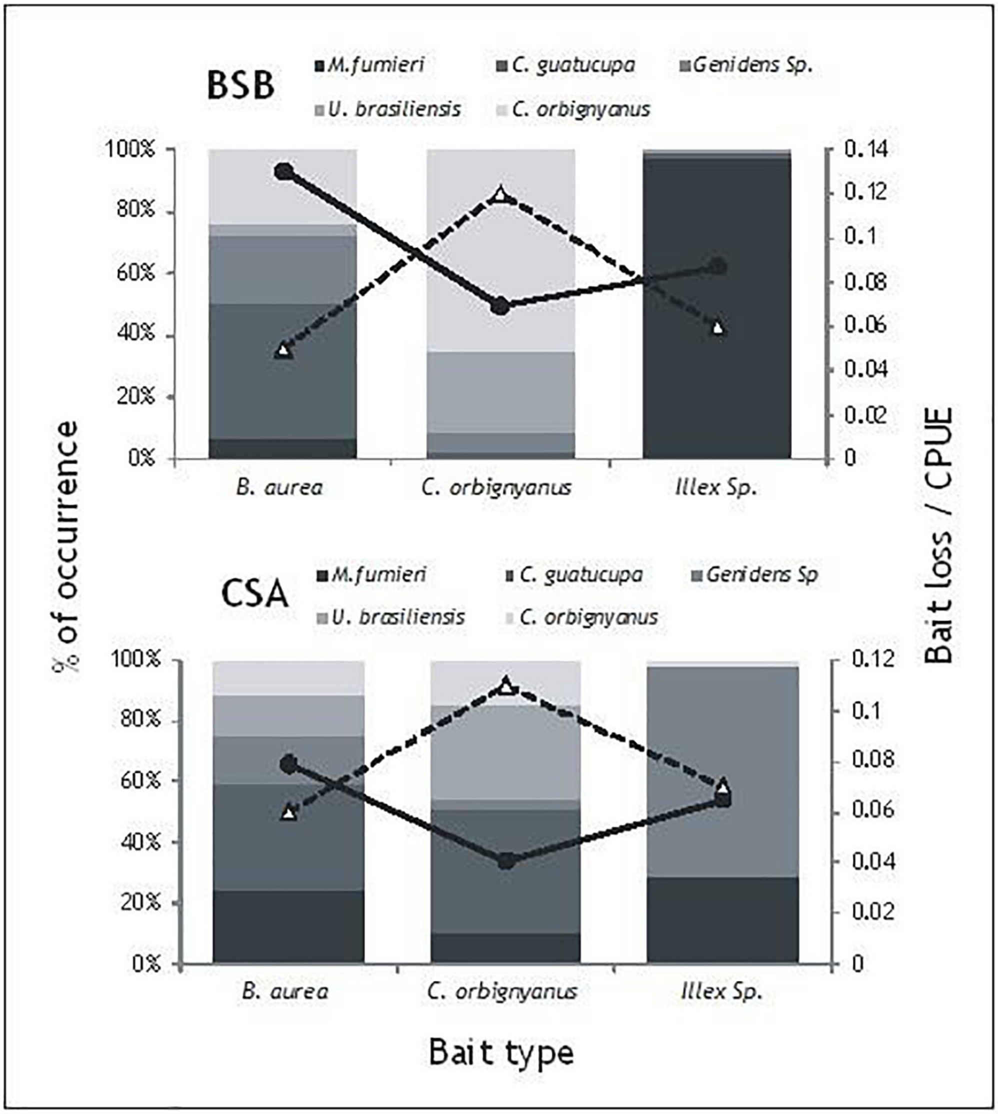

FIGURE 6 | Commercial fish catch in relation to bait type and bait loss with bottom longlines in BSB and CSA. CPUE (black circle, full line), Bait loss rate (white triangle, dotted line).

The analysis of fish catch rates showed no significant difference between individual fishermen, suggesting that although this was a new fishing method for them, fishermen were equally able to operate the experimental bottom longlines although they reported concerns about getting hooked.

Similar catch size frequencies for all targeted species were also recorded in both fishing gear types at BSB and CSA, indicating relative high size selectivity by both gears. The bimodal curve obtained with bottom longlines might be due to recruitment of different year classes into the population (Millar and Holst, 1997), influenced by fish behavior and catching process. Although baited gear has been reported to catch bigger fish than gillnets (Hovgard and Lassen, 2000; Santos et al., 2002; Stergiou and Erzini, 2002; Erzini et al., 2003), no significant differences were found in this study. However, bottom longlines were more effective at catching species with the highest local market prices such as Catfish and Brazilian codling. The catch size frequencies of the most common species recorded with both fishing gears showed that this coastal fishery mainly targets mature individuals (Macchi and Acha, 1998; Bezzi et al., 2000; Macchi et al., 2003). The most important commercial and heavily exploited fish in coastal Buenos Aires is Whitemouth croaker (Lasta and Acha, 1996). The proportion of juvenile Whitemouth croaker caught by gillnets (6%) or bottom longlines (3%) in this experiment was significantly lower than the average 33% reported for trawlers operating in BSB (Carozza and Lorenzo, 2011), indicating the high selectivity of gillnets and bottom longlines in areas where juveniles of several fish species are present (Lasta, 1995; Lasta and Acha, 1996; Acha et al., 1999; Acha and Macchi, 2000; Macchi et al., 2003; Militelli, 2007).

TABLE 5 | Summary of interactions with franciscana dolphins, sea turtles, sea lions in relation to gear type, number of hauls, catch, and fishing effort.

	Gillnets				Bottom longlines			
	BSB 04–05	CSA 04–05	BSB 05–06	BSB 06–07	BSB 04–05	CSA 04–05	BSB 05–06	BSB 06–07
Total number of hauls	135	138	118	61	138	121	112	44
Total FE (km^{-1} hr^{-1}) or (hooks hr^{-1})	1262.5	882.2	1001.2	412.1	32506	28687	30101	11174
Total commercial fish catch (kg)	6521	6409	5605	3573	3218	1865	2277	1356
Number of bycaught dolphins	35	17	22	11	1	0	0	0
BPUEdolphins (fishing effort)	0.028	0.019	0.022	0.027	3.1×10^{-7}	0	0	0
BPUEdolphins (kg of fish catch)	0.0054	0.0026	0.0041	0.0031	0.00031	0	0	0
number of bycaught sea turtles	35	3	31	13	4	0	0	0
BPUEsea turtles (fishing effort)	0.028	0.0034	0.031	0.031	0.00012	0	0	0
BPUEsea turtles (kg of fish catch)	0.0054	0.00047	0.0055	0.0036	0.0012	0	0	0
Number of sea lion attacks	172	108	81	54	47	42	34	14
Number of attacks/haul	1.3	0.8	0.7	0.9	0.3	0.3	0.3	0.3
% of hauls with attacks	31%	38%	33%	38%	19%	14%	17%	21%
APUE	0.026	0.017	0.014	0.015	0.015	0.022	0.015	0.010

FE, fishing effort; CPUE, catch per unit effort; APUE, number of attacks per unit effort.

Although the majority of fish caught by both gear types were of good quality, bottom longlines caught a higher proportion of top quality catch (perfect condition or alive), particularly of the two main targeted species. Bottom longlines seem to be more efficient in BSB than CSA as measured by the relative fishing performance recorded.

Bycatch Mitigation Effectiveness

Total biomass was also used to compare relative bycatch rates of dolphins and sea turtles as well as sea lion interaction rates with the two types gear.

Bycatch in artisanal fishing gillnets has been recognized as the main threat for the franciscanas (Reeves et al., 2012). Previous studies of mortality rates and abundance estimation have demonstrated that the level of bycatch is considered unsustainable for the FMAIV (southernmost franciscana population) (Perrin et al., 1994; Crespo et al., 2010; Negri et al., 2012). Information from genetic analysis (Mendez et al., 2008; Gariboldi et al., 2015) and movement patterns (Bordino et al., 2008; Wells and Bordino, 2013) indicate that franciscanas might be aggregated in subpopulations along the distribution of the species. In terms of conservation, each subpopulation is considered as a relevant evolutionary unit that should be protected.

Franciscanas with satellite linked tags of BSB have shown that they do not move down south to CSA (Bordino et al., 2008), suggesting that there is a subpopulation of franciscanas in BSB and another one in CSA, which correlates with the fact that one population inhabits an estuarine system whereas the other one is inhabits an open sea area. This leads to a difference distribution of fish relative abundance and thereby on the fishing practices. According to those differences, conservation measures should be adapted to a local level considering the range of the fishing community practices and the ecological implications for the species.

Conservation strategies are not based on a single method but on a compliance of different techniques involving technology applied to the local traditional fishing methods (e.g., pingers, led lights, reflective gillnets) and alternative fishing methods such as fishing pots or longlines (FAO, 2018a). Switching fishing gears implies a challenge for traditional fishermen that can only be attempt after evaluating the bycatch potential reduction of the alternative fishing gear.

In the present study, dolphins were bycaught in gillnets at both study areas. The BPUE*dolphins* in gillnets (km net^{-1} h^{-1}) obtained is consistent with previously estimated ranges for the same fisheries (Bordino et al., 2002, 2004, 2013; Bordino and Albareda, 2004), indicating that the incidental bycatch remains over sustainable levels when fishing with gillnets and suggesting that management actions for mitigating the incidental bycatch are required urgently.

In contrast, only one dolphin was bycaught in bottom longlines, accidentally hooked by a pectoral fin at CSA. Previous interviews with local artisanal fishermen in the study area did not report any interaction with franciscanas when using hooks (Weiskel et al., 2002). However, entanglement and death in unbaited and rolling hooks was documented for the Baiji (*Lipotes vexillifer*) (Zhou and Li, 1989; Zhou and Wang, 1994; Zhou et al., 1998; Reeves et al., 2000) and up to 15 odontocetes species have been recorded bycaught on pelagic longline hooks (Hamer et al., 2012). Therefore, the possibility of increased bycatch risk and/or depredation of bait by franciscana in this fishing gear should be monitored. Nevertheless, the present study has proven that longlines can be an effective alternative fishing gear for franciscana bycatch mitigation, although its implementation is not solely based on the mitigation potential but also on a benefit for the fishermen, from a cultural and economic point of view.

The use of fishing lines, especially pelagic longlines, has been widely reported to result in the bycatch of seabirds, sea turtles (e.g., Cherel et al., 1996; Barnes et al., 1997; Lewison et al., 2004b)

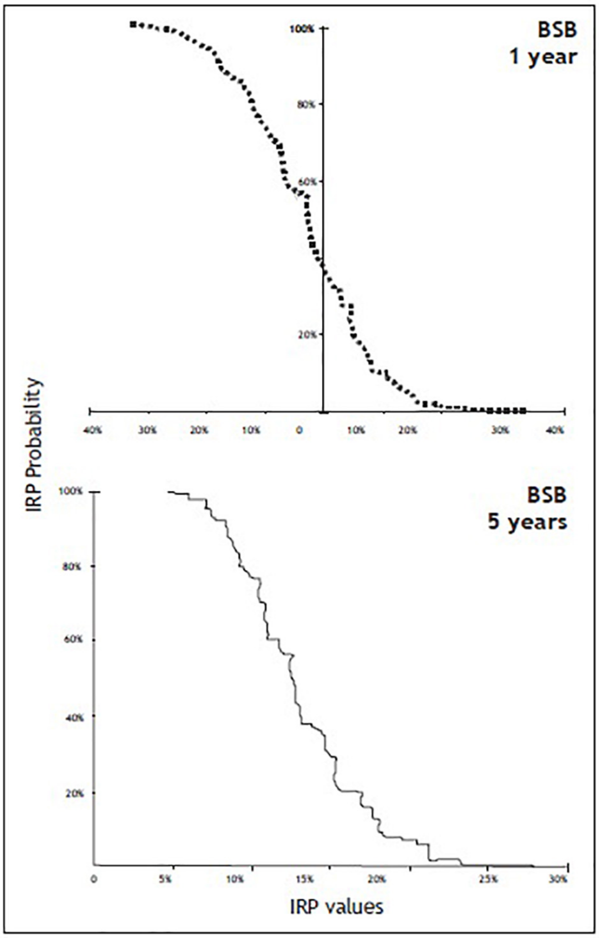

FIGURE 7 | Incremental Profitability Rate (*IRP*) profiles for BSB at 1- and 5-year scenarios.

TABLE 6 | Estimated unit cost (in kilograms of fish equivalent) for operational gillneting considering the average fishing effort, the catch rate for each gear during the experiment, and comparable fishing efforts between gears in BSB (400 m gillnets and 800 hooks).

Item	Average Cost (kg)	400 m Gillnet	800 hooks Longline
Used boat and fishing permit	6000	6000	6000
Used 60–90 outboard engine	5500	5500	5500
Gillnet unit (100 m)	200	800	0
Bottom longline units (80 hooks)	100	0	1000
Daily labor crew	40	3200	3200
Daily hooking + bait + cleaning	40	0	3200
Daily gas and oil	65	5200	5200
Annual boat maintenance	380	380	380
Annual taxes and insurance	550	550	550
Total assets (kg)		21630	25030

and marine mammals (Szteren and Páez, 2002; Donoghue et al., 2003; Hamer et al., 2012). However, no seabirds were caught in bottom longlines or even gillnets during this experiment, and have not previously been documented as bycatch in this artisanal fishery.

Sea turtle bycatch was relatively common in gillnets but rare in bottom longlines. Given the relatively low soak times of bottom longlines it is possible that incidentally bycaught sea turtles would be found alive and might of survived following release. In addition to the soak times the use of different hook types could further reduce the risk of serious injury or mortality on entangled sea turtles.

In this study, South American sea lions interacted with both bottom longlines and gillnets, and depredated fish without resulting in an entanglement. Such opportunistic behavior seems to be more evident in gillnets than bottom longlines as indicated for the comparative proportion of hauls with evidences of depredation and the APUE. Although the predatory behavior of pinnipeds in fisheries is a cause of concern due to the potential cost to fisheries (Marsch et al., 2003), during this experiment depredation represented less than 5% of the commercial fish catch. In addition to depredation of fish, sea lion interaction also resulted in some small losses of snoods and hooks, and holes in gillnets which can result in efficiency reduction. Damage to gear rather than catch loss is a primary concern for local fishermen. The intensity of sea lion interactions is likely to be influenced by a number of factors including the presence of entangled or hooked fish, the number and motivational state of sea lions around the gear, the gear location, environmental conditions and even the presence of predators such as sharks or killer whales (Harcourt, 1992; Hückstädt and Antezana, 2004; Vila et al., 2008). The depredation or damage to gear recorded during this study does not seem to be responsible for significant variations in fish catches.

Economic Implications

Economically, bottom longline gear was relatively cheaper because it lasted longer over several fishing seasons, had the same selectivity, caught species of higher quality and value in the local market and required shorter soak time than gillnets. However, the additional expense of bait and labor for hooking and cleaning the gear increase the operational cost, resulting in a lower revenue compared to the gillnets.

Caution must be taken when interpreting the results of the economic profitability analysis because the price of the fish in the local market is difficult to predict since it is usually driven by the productivity of the trawler fleet operating in the same areas affecting supply and demand. Demand is the main factor directly influencing fishing effort in this coastal fishery (Lagos, 2001). Incomplete knowledge of possible changes in the values of some of the variables used, typical in many developing countries, can also affect the analysis. The economic analysis performed only provides an overview of the profitability using data obtained from a relatively small-scale experiment.

In addition, this analysis did not consider the economic advantage of using bottom longlines from increased value of the fish caught, which yielded a better quality of catch and/or catches of commercial species with higher values than Whitemouth

TABLE 7 | Analysis of the economic indicators (in kg of fish equivalent), based in fish catch rates recorded during this experiment and combined from interviews and Coast Guard reports.

	BSB (this experiment)			BSB (combined)		
Economic indicator	**400 m Gillnet**	**800 hooks Longline**	**200 m gillnet +400 hooks**	**400 m Gillnet**	**800 hooks Longline**	**200 m gillnet +400 hooks**
Average total gross revenue (TGR)	16976	16976	16976	22953	22953	22953
Operational cost (OC)	6930	7130	7030	6930	7130	7030
Gross value added (GVA)	10046	10246	9946	16023	15823	15923
Labor cost (LC)	3200	6400	4800	3200	6400	4800
Gross cash flow (GCF)	6846	3846	5146	12823	9423	11123
Depreciation of gear (D)	800	250	525	800	250	525
Economic profit (EPR)	6046	3596	4621	12023	9173	10598
Profit margin (PRM)	35.6%	21.2%	27.2%	52.4%	39.9%	46.2%
Return on investment (ROI)	27.9%	14.4%	19.8%	55.6%	36.6%	45.4%

croaker or Stripped weakfish, or the possibility of more than one haul per day. As such, this analysis indicates a minimum expectation of profitability. The conservation status of the franciscana will be addressed only after the amount of fish and money coming in is stable and fishermen can be reassured that conservation measures will not jeopardize their ability to support household economy (Weiskel et al., 2002).

In general, gillnets and bottom longlines had a relatively similar efficiency/selectivity rate, but technical reasons, biological variables, environmental changes, or even cultural rather than economic aspects would also determine what gear might be used. As far as we know, the comparative advantages and disadvantages between gillnet and bottom longlines have not been examined for most coastal fisheries in Argentina. The present study shows strong potential for the use of bottom longlines as a viable alternative fishing gear for, at least, BSB. While a number of studies have compared fishing gear types in terms of catch composition, catch rates and selectivity, few studies have been based on the comparison of gear operating simultaneously in the same fishing grounds (e.g., Hovgard and Riguet, 1992; Engas et al., 1993; Huse et al., 1999; Hallyday, 2002; Santos et al., 2002; Stergiou and Erzini, 2002; Erzini et al., 2003; Eckert and Eckert, 2005). Bottom longlines in BSB seem to be favorable in terms of the sustainable use of living resources due to minimal capture of undersized fish, low discard and limited bycatch of non-target species.

Considering the necessary increase in bottom longline effort to catch the same amount of fish than gillnets established during this study, the net economic return remains favorable for this small-scale fishery. A hypothetical simulation where 50% of gillnet fishing effort is switched to bottom longlines resulted in only a profitability rate reduced by 6–8% when compared to only using gillnets in BSB. Accounting for costs and associated profitability, reducing gillnet effort by switching to bottom longlines appears to be a practical option involving a relatively low cost that could result in a significant reduction of current bycatch rates of franciscana in Argentina.

Even if bottom longline effort is increased by 2.5 times to catch an equivalent amount of fish as gillnets, the expected franciscana bycatch rates would still be 90% lower than the average recorded in gillnets in these fisheries. However, this estimate assumes a linear relationship between bycatch rates and bottom longline fishing effort. Subsequent studies should examine variability in catch as a function of soak time, longline length, and hook spacing.

Further work is recommended to increase the scale of the field trials before any attempts are made to promote the use of bottom longlines in this area. The effectiveness of any bycatch mitigation strategy may be reduced by a lack of compliance or inappropriate use of the gear among other factors. Moreover, a decrease in efficacy between experimental and fleet-implementation results may have serious consequences for the conservation of bycaught species, especially in the absence of an effective monitoring program (Cox et al., 2007). The evaluation of circumstances other than profitability is needed to determine the prospect of fishermen adopting this alternative gear. Such a change would similarly need to address how the cultural and social context bears on transitioning over to new fishing gear and can help shed light into how the interplay of these variables influence changes in artisanal fisheries.

DATA AVAILABILITY STATEMENT

The datasets generated for this study are available on request to the corresponding author.

ETHICS STATEMENT

Ethical review and approval was not required for the study on human participants, in accordance with the local legislation and institutional requirements.

AUTHOR CONTRIBUTIONS

LB was responsible of and data collection and analysis. PB was the project leader in charge of the project planning and data analysis. MG and MF were in charge of the economic analysis. AM and TW were fundamental for the data analysis and writing of this manuscript. All authors contributed to the article and approved the submitted version.

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Community Engagement: An Integral Component of a Multifaceted Conservation Approach for the Transboundary Western Pacific Leatherback

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Bycatch in fisheries is one of the greatest threats to marine megafauna such as sea turtles, and the Biodiversity Impact Mitigation Hierarchy (BIMH) has been proposed as an improved and holistic approach for integrating fisheries management with sea turtle conservation. The first three BIMH steps – avoid, minimize, and remediate – take place at sea where fishing activity is taking place. However, these at-sea measures are costly and difficult to effectively implement across the vast range of a highly migratory species. As such, some level of mortality continues, even when the first three steps of the BIMH are implemented as extensively as possible. These remaining negative impacts need to be addressed by compensatory conservation actions elsewhere, e.g., at sea turtle nesting beaches. As a case study, we use the critically endangered leatherback sea turtle nesting population in Papua Barat, Indonesia, to illustrate the opportunity for conservatory offsets to fisheries bycatch across the Pacific. We describe the community empowerment and nest protection programs that have been enhanced by the voluntary offsets from the tuna industry. While improved nest protection measures have helped optimize hatchling production, the engagement of the local communities, through activities that empower and enhance quality of life, has been a critical component to the successful increase in hatchlings. This momentum needs to be sustained and scaled-up to protect the majority of threatened nests over a consistent number of years to successfully provide the recruitment boost needed at the population level. These compensatory off-site conservation measures are also the most cost-effective means of achieving increases in leatherback populations, and perhaps one of the most critical components of the recovery strategy for Pacific leatherbacks.

Keywords: Papua Barat, Indonesia, leatherbacks, community engagement, hatchling production

INTRODUCTION

The conservation of critically endangered populations is a complex, multi-disciplinary and multi-faceted undertaking (Bennett et al., 2017). This requires consistent and reliable monitoring of the population, effective control of threats across all life history stages, long-term engagement with local communities and relevant government authorities, and the development of sustainable, creative, and flexible management strategies. The life history of sea turtles makes conservation of their populations particularly challenging. They spend most of their life at sea, often traversing the waters of many countries, while coming ashore only to lay eggs that incubate in the sand for approximately 2 months before the hatchlings emerge. Despite the need for a holistic conservation strategy, which addresses all sources of mortality across life history stages (Bellagio Blueprint for Action on Pacific Sea Turtles, 2011; Dutton and Squires, 2011; NOAA-NMFS, 2016), efforts to mitigate sea turtle bycatch continue to be enacted in a piecemeal manner in individual fisheries, typically under the assumption that nesting beach conservation is being effectively carried out. Furthermore, at-sea and nesting beach management tend to operate independently under different regulatory frameworks and funding initiatives. Therefore, to consolidate these various conservation efforts more effectively, the Biodiversity Impact Mitigation Hierarchy (BIMH) has recently been proposed as an improved, holistic and integrative approach for fisheries management and biodiversity conservation (Arlidge et al., 2018; Milner-Gulland et al., 2018; Squires and Garcia, 2018) because it considers a broader suite of actions to reduce anthropogenic mortality across a bycaught species' entire life cycle and range of habitat use (Squires et al., 2018). In the BIMH, the primary approach is mitigating bycatch at sea where fishing is taking place. However, at sea measures, such as gear modification, time-area closures, and best fishing practices, are costly and difficult to effectively implement in multiple countries across the vast range of a highly migratory species. Therefore negative impacts from fishing remain, despite these bycatch impact reduction measures. This mitigation deficit needs to be addressed by the last component of the BIMH, which are compensatory conservation measures at sea turtle nesting beaches (Squires et al., 2018).

As an example, we use the critically endangered leatherback turtle (*Dermochelys coriacea*) nesting population on the beaches of Jamursba-Medi and Wermon in the Bird's Head Abun region of Papua Barat, Indonesia, in the Western Pacific to illustrate the opportunity for conservatory offsets to fisheries bycatch within the BIMH framework. We describe the implementation of multi-level conservatory offsets within the BIMH strategy for this population through nesting beach protection and optimization of reproductive output, innovative approaches to engage local communities in the nesting beach conservation effort, identification of metrics for quantifying conservation gain, and review of elements necessary for success from three decades of conservation effort.

LEATHERBACK NESTING IN PAPUA BARAT, INDONESIA

Leatherback sea turtle populations are classified as critically endangered in the Pacific by the IUCN Red List (Tiwari et al., 2013), and the severe decline of over 90% is due to egg harvesting, exploitation for meat and fat, habitat destruction, and bycatch in artisanal and commercial fisheries. In the eastern Pacific, leatherback populations in Mexico, Nicaragua and Costa Rica have declined dramatically (Ábrego et al., 2020). In the western Pacific, the once large nesting population in Malaysia is now considered functionally extinct (Chan and Liew, 1996) and although the key leatherback populations remaining in the western Pacific, in Indonesia, Papua New Guinea, and the Solomon Islands are severely depleted, they offer the best prospects for recovery (Dutton et al., 2007). Approximately 75% of this nesting takes place along the north coast of the Bird's Head peninsula in the Abun region of Papua Barat, Indonesia (Dutton et al., 2007).

Two index beaches in the Abun region of Papua Barat, Indonesia – Jamursba-Medi (18 km) and Wermon (6 km) – support the last, largest leatherback nesting population remaining in the Pacific (**Figure 1**). Nesting occurs year round; April to September (boreal summer) and October to March (austral summer; Hitipeuw et al., 2007). Leatherbacks from this population forage widely – boreal summer nesters head to the South China Sea, western USA, the equatorial eastern Pacific, and the Kuroshio Extension region whereas boreal winter nesters travel toward southeastern Australia and Tasmania (Benson et al., 2011); fisheries bycatch on these migratory routes and foraging grounds is speculated to have considerably impeded the recovery of this population (Roe et al., 2015).

Despite the alarming decline observed in this nesting population since the 1980s (Tiwari, *unpublished data*; Tapilatu et al., 2013), it is considered the most robust nesting population remaining in the Pacific that has not yet collapsed to the point of being functionally extinct. Hope for the recovery of this population persists as long as holistic and effective conservation and management measures are consistently implemented throughout the population's range and critical life history stages (Dutton and Squires, 2011).

Within the BIMH framework, the at-sea conservation strategies for leatherbacks in the North Pacific include technology standards, time-area closures (e.g., in California/Oregon drift gillnet fishery), gear and effort restrictions such as hard caps on bycatch (e.g., the Hawaii-based longline fishery), and best practices that together comprise a strategy designed to reduce bycatch and decrease mortality of turtles. However, at-sea measures are challenging to implement across the entire migration range and do not address the large cumulative impact of artisanal fisheries in many developing countries that impact leatherbacks on their migratory routes and in coastal foraging areas on both sides of the Pacific (Arlidge et al., 2020). Given that preventing turtle bycatch completely is both challenging and costly, residual negative impacts on turtle populations continue to take place. Therefore, compensating for this, and achieving

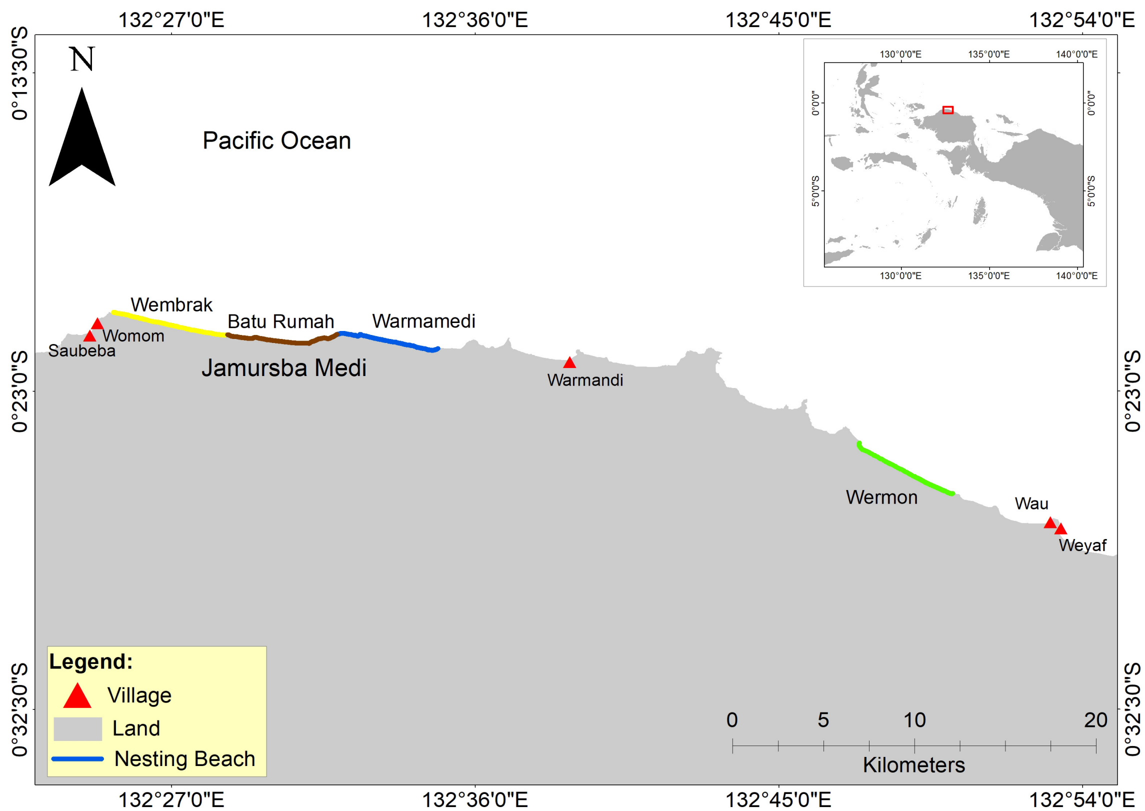

FIGURE 1 | Location of the nesting beaches (Jamursba-Medi and Wermon) and the adjacent villages (Saubeba, Womom, Warmandi, Wau, and Weyaf).

population recovery, depend on effective conservation of nesting populations. This conservatory offset to mitigating bycatch is perhaps the most critical component of the recovery strategy for Pacific leatherbacks. It also satisfies the criteria of the BIMH framework by addressing mortality at a different (and critically important) part of life cycle of the same population that is impacted by high seas and coastal fisheries. Conservatory offsets for sea turtles have yet to be formally incorporated under fisheries management and regulatory frameworks. However, Squires et al. (2018) describe a voluntary program where processors assess a tax each year on tuna landings from longline fisheries (known to impact sea turtles) to fund conservatory offsets at nesting sites, including the Abun Leatherback Project (ALP), through the International Seafood Sustainability Foundation (ISSF). We further describe how conservation outcomes at the Abun leatherback nesting beaches have been enhanced by the local community engagement.

Abun Nesting Beach Program

Since the 1980s, some level of population monitoring has been conducted at Jamursba-Medi and Wermon Beaches, but it was inconsistent and at best reduced harvest of females and eggs (Hitipeuw et al., 2007). However, we now know that nest destruction from feral pigs and dogs, tidal inundation, erosion, and high sand temperatures have resulted in low hatchling productivity (Tapilatu and Tiwari, 2007), and these threats began to be addressed only in this past decade with at best 35% of threatened nests being protected (Tiwari, *unpublished data*). Since 2017, the ALP has adopted a more effective and consistent protect-as-many-nests-as-possible strategy to optimize hatchling output. This overarching goal of maximizing hatchling production has been demonstrated elsewhere to be critical for population recovery (Tiwari et al., 2011). In Papua, ALP's community engagement program has played a very important role in increasing the percentage of nest protected (≥50%) using all strategies. In high risk areas, individual nest enclosures are built to protect nests *in situ* from pig, dog and monitor lizard predation, and nests are shaded individually *in situ* with palm fronds to lower lethal sand temperatures. Local community members are also hired to trap feral pigs in the forest behind the beach to further reduce the intensity of predation. Nests highly vulnerable to erosion and inundation are relocated to hatcheries or stable sections of beach.

However, while the conservation actions needed on the nesting beach are clear, the establishment of a long-term sustainable program in this remote area has been much more challenging. Jamursba-Medi and Wermon Beaches are owned by families in five adjacent villages. These landowners determine access to the beach and what conservation and management activities can be conducted. Therefore, in order for conservation actions to effectively achieve the desired goal of maximizing hatchling production, it is critical to have community support and involvement.

Abun Community Empowerment Program

Community-based conservation is a least-cost approach to turtle conservation especially in these remote areas with traditional land tenure and impoverished communities (Gjertsen et al., 2014). Local beach communities have strategic roles in conserving biodiversity and ecological functions, however, the historical approach of dealing with the local Abun communities within the conservation framework has been problematic. Community buy-in, welfare, and empowerment were overlooked because those trained in the biological-sciences focused more narrowly on more traditional community-based measures through obvious direct involvement in conservation (e.g., hiring villagers as patrollers, paying concession fees to landowners, and monetizing "eco-tourism"). In Papua, this sowed a sense of social/economic inequity among the village communities and a general discontent toward the leatherback project. A first social survey carried out in the villages adjacent to the beach in 2010 provided insights into local demographics, economy, education, health, and infrastructure available, but more importantly this survey revealed that community members believed that conservation projects functioning on their beaches since the 1980s had so far only prioritized the leatherbacks and a small group of community members, especially the landowners, with few benefits to the broader community (Gjertsen, 2011b). Therefore, integrating the local communities into the project was prioritized through carefully evaluated and targeted quality of life-enhancing activities (Pakiding et al., 2017) as described below, which benefit and empower the entire community (Waylen et al., 2010; Wright et al., 2015).

The community empowerment project was enhanced by the ISSF conservatory offset funding, and implemented through the placement of ALP staff or community workers, mostly fresh graduates from the State University of Papua, in the five villages for almost 10 months of the year since 2013. These community workers determine community needs by interviewing the local leaders every 6 months and guide the development of numerous projects desired by the community. The main livelihood of these communities is farming and hunting, therefore, ALP focused on the community's desire for increased skills and capacity in the agricultural and meat processing sector by introducing improved techniques, processing, and marketing of products. Over time some community members have adopted the introduced technology and were able to sell their products in nearby cities. For example, ALP's community worker introduced improved techniques to make coconut oil in 2013. This program grew from 5 households in 2017 to 38 households in 2019. In 2019, the community produced 2,200 liters of good quality coconut oil worth 4,250 USD. Local community members now make coconut oil during 3–4 months in a year when they cannot sell their agricultural products to nearby cities because of bad weather. Another livelihood-focused scheme introduced in 2018 was teaching the village women to knit traditional bags or "nokens." By 2019, 16 women were involved and had earned around 823 USD. Marketing of these products remains a big challenge, but an initiative by the Office of Cooperatives, Micro, Small, and Medium Enterprise of the Tambrauw District Government in 2019 to establish a marketing cooperative is expected to be one of the solutions.

Given the lack of public awareness about the importance of education, shortage of teaching staff, and limited school facilities and infrastructure, ALP has improved formal education opportunities at the elementary school level in the villages through government and private entities. An informal education program was also established for children not in school at ALP's learning houses. An important goal of the informal education program is to increase the children's awareness and respect for the wealth and importance of their natural resources. Therefore, ALP hosts a Turtle Camp every year and takes village children to the beach to see nesting leatherbacks, to release hatchlings, and to learn about turtle biology and conservation, while they also learn basic hygiene (brushing their teeth, showering, washing hands and feet), and the importance of garbage management. Even village adults are taught about proper garbage disposal and cleanliness in the village and homes. The educational programs, both formal and informal, have been successful and the community respects ALP's efforts. The Tambrauw District Education Office has even used ALP to assist with national examinations for elementary school students.

The lack of medical equipment and medical personnel is one of the biggest problems in these remote villages, and villagers rely on traditional medicines to cure their illnesses. ALP has collaborated with the Provincial and the Tambrauw Regency Health offices to conduct a health program for these communities. This collaboration is an opportunity for the Regional Government to evaluate the health of the community, provide health education, health checks, and free treatment. Meanwhile, medical team visits are organized by ALP to provide regular community health support. The local government is expected to follow up on these activities with better health programs for the communities.

An increasing number of community members are also hired for nesting beach work. Those who have customary rights to the beach work with ALP to monitor the beaches, identify additional community members to help ALP monitor the beaches, and prevent illegal activities. Community members without customary rights are hired by ALP for beach monitoring, nest relocation, hatchery construction, pig-trapping and other project-related activities with the approval of landowners. All community members benefit from the ALP community programs in the villages. Overall, ALP is bringing more income and improved living-conditions to the community.

Conservation Benefits

ALP has gained the trust and goodwill of community members largely because of the community empowerment project, which is showing promise toward enhancing the sense of "ownership" or the intangible "value" of leatherbacks' existence on their beaches. Now as a result of several years of ALP's presence in the community, their flexible and adaptive response to community feedback, needs and interests, and their consistent messaging that the ALP community and nesting beach team members belong to the same project, the local communities have started to understand and appreciate that they are benefiting because of the leatherbacks.

The benefits of this change in attitude are evident also on the nesting beach. Between 2013 and 2016, community members used to openly offer turtles eggs to ALP staff, but today this no longer occurs; local children report that turtle eggs are not served at home, and they are ashamed if caught consuming turtle eggs. Furthermore, since access to the nesting beaches is controlled by several families in Abun who also decide what activities can be undertaken, stabilizing beach access has been at the forefront of ALP's achievements. With secure access starting in 2017, ALP with community support has been able to protect a larger percentage of leatherback nests and increase hatchling production. It should be noted that the increase in hatchling production corresponds to the increased number of community members working with ALP's nesting beach staff.

Conservation Equivalency

Since the conservation benefit to the population of protecting (e.g., producing) a sea turtle hatchling is much lower than protecting a larger juvenile or reproductive female, it is important to account for the relative equivalency when evaluating conservatory offsets targeting different stages of life history. A better understanding of population dynamics (survival rates, age to maturity, sex ratios), as well as bycatch mortality, is needed to develop robust equivalency models, however for our purposes, we can consider some broad equivalencies with the available information for leatherbacks. Note that nest protection, and the resulting increase in hatchling production, is one component of a holistic conservation strategy that includes protection of nesting females on the beaches, the highest level of offset in terms of reproductive value to the population (Gjertsen et al., 2014).

With the initiation of nest protection measures, mean hatchling production was estimated at 21,996 between 2005 and 2013 at Jamursba-Medi during the boreal summer and at 9,490 at Wermon between 2005 and 2011 during the austral summer (Tapilatu, 2014); prior to this almost all the nests were destroyed (Hitipeuw et al., 2007). In the recent years of stable beach access and community engagement (2017–2019), hatchling production in Jamursba-Medi and Wermon increased to 32,000–50,000 hatchlings between April and September alone (Tiwari, *unpublished data*). Therefore, if the estimated reproductive value of 426 hatchlings = 1 adult reproductive female in Papua (Gjertsen, 2011a), then the hatchling production in Jamursba-Medi and Wermon during April to September results in 75–117 adult females. Lewison et al. (2015) estimated that between 1990 and 2011, 678 leatherbacks were taken by longlines and 93 in nets representing 771 leatherbacks taken in 21 years or on average 37 leatherbacks/year in the Western Pacific Regional Management Unit (RMU; Wallace et al., 2010). It appears that hatchlings produced in Jamursba Medi and Wermon in recent years (equivalent to 75–117 adult females a year) would offset the estimated 37 leatherbacks/year taken by these fisheries.

Peatman et al. (2018), however, estimated a median interaction with 24,006 leatherbacks between 2003 and 2017 in longline gear alone in the Western and Central Pacific Fisheries Commission's Convention Area suggesting that take and mortality levels would be much higher than those estimated by Lewison et al. (2015). Given the uncertainty in bycatch estimates and mortality rates across all fisheries in the western Pacific RMU (Wallace et al., 2013; Lewison et al., 2015; Peatman et al., 2018) and the serious declines observed in the nesting population, maximizing hatchling production will be critical for population recovery. A similar approach was emphasized for the eastern Pacific leatherback populations, whose situation is more dire and extirpation is expected in less than 60 years if urgent measures are not implemented to save 200–260 adult and subadult leatherbacks and produce 7,000–8,000 more hatchlings annually (Ábrego et al., 2020).

CONCLUSION

There is an increasing body of research on advancing conservation by influencing human behavior (e.g., Reddy et al., 2016) as well as calls for multi-stakeholder dialogues by the United Nations to build partnerships and identify solutions that are aligned with sustainability objectives (UNEA-4, 2019). Recognizing that local communities and their welfare are an integral component of the conservation equation is fundamental to the success of biodiversity conservation. Additionally, social and natural science professionals need to understand that communities have their own legitimate perspectives on conservation in order to be effective (Berkes, 2007; Bennett et al., 2017). In Jamursba-Medi and Wermon, prioritization and empowerment of the local communities are having a positive impact on the ability to protect leatherback nesters and produce increased numbers of hatchlings. This momentum needs to be sustained and scaled-up to protect the majority of threatened nests over a consistent number of years to successfully provide the recruitment boost needed at the population level. This effort will be greatly enhanced by the creation of the Jeen Womom Coastal Park at Jamursba-Medi and Wermon by the local Tambrauw government and its legalization by the Ministry of Fisheries and Marine Affairs in 2018. The newly formed Technical Implementation Unit of the government (UPTD), trained by ALP, will be responsible for all aspects of Park governance, protection, and sustainability.

Within the BIMH framework, the challenges of Pacific-wide at-sea bycatch mitigation necessitate dynamic and persistent conservation measures on the nesting beaches to optimize hatchling production. Further work is needed to develop mechanisms for determining the residual bycatch cost of specific fisheries, and to develop demographic models to quantify the conservation benefits. Meanwhile, compensatory off-site conservation measures remain the most cost-effective means of achieving increases in leatherback populations (Gjertsen et al., 2014), and a critical component in Pacific leatherback recovery.

DATA AVAILABILITY STATEMENT

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

ETHICS STATEMENT

The studies involving human participants were reviewed and approved, in accordance with local requirements in Papua Barat, Indonesia, by the Regent of the Tambrauw Regency; Abun District Head, Tambrauw Regency; and Department of Marine and Fisheries (DKP) of West Papua Province. The animal study was reviewed and approved, in accordance with local requirements in Papua Barat, Indonesia, by the Regent of the Tambrauw Regency; the West Papua Provincial Government Office of Maritime Affairs and Fisheries; and all the local landowners who decided what studies can be conducted on their beaches.

AUTHOR CONTRIBUTIONS

FP was the project leader. KZ, AA, and SK implemented the community components of the project. DL implemented the nesting beach component of the project. MT was the scientific and technical advisor to the leatherback project. MT, PD, FP, and DL were co-leads of manuscript preparation. All authors contributed to the article and approved the submitted version.

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Mitigating Seafood Waste Through a Bycatch Donation Program

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Discarding of prohibited, under-sized, or non-target finfish is a major problem globally. Many such unwanted or banned catches do not survive long enough to be released alive, creating complex ecological and policy issues for the fishing industry. In U.S. Federal waters, regulation requires bycatch to be avoided as practicable and bycatch of some finfish species is designated as prohibited species catch (PSC). By regulation, PSC cannot be retained or sold and it must be returned to the sea (dead or alive). Some PSC species have strict limits to further incentivize their avoidance and limit bycatch mortality and these limits can lead to fishery closures. Despite extensive efforts to avoid bycatch in the U.S. and elsewhere, unwanted catches still occur, creating the potential for substantial food waste. We present one rarely discussed approach to maximize the value of dead, unwanted or prohibited finfish catches. The Prohibited Species Donation (PSD) program utilizes trawl fishery PSC that would otherwise be discarded by instead donating it to hunger relief organizations. This program simultaneously provides food and reduces waste while avoiding inadvertent incentives for catching prohibited species. For 26 years, the non-profit organization, SeaShare, has worked with the Alaska seafood industry to distribute 2,660 t (~23.5 million servings) of prohibited species donations (salmon and halibut), high quality seafood that would have otherwise been discarded due to prohibition on retention. The PSD program provides an example that addresses food security and social value, an under-represented perspective in the global dialogue on unwanted catches.

Keywords: prohibited species, seafood waste, fisheries management, seafood industry, fishery discards

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INTRODUCTION

Discards account for nearly 10% of global fishery catches annually (Zeller et al., 2017), and this wasteful practice has been an increasing focus of management, research, and public concern. Finfish may be discarded for many reasons (e.g., regulations prohibit retention, fish are undersized, lack of market demand or value, quota overage). Some countries (e.g., Norway, Chile, Iceland), and more recently, the European Union have banned discarding (Karp et al., 2019). One goal of such bans is to incentivize more selective fishing, encouraging fishermen to fill their holds with valuable target species instead of unwanted (prohibited, under-sized, or non-target) catches that often get wasted (Borges et al., 2016). Despite such efforts to minimize discards, bycatch cannot be completely eliminated in most fisheries. From the food security perspective, better utilizing the spectrum of edible fish catches and thus, minimizing waste should be a priority (Borges et al., 2016; Van Putten et al., 2019).

Historically, the focus on waste reduction in fisheries has been on the supply-side of the issue, largely centered around efforts to avoid unwanted catches altogether. However, Van Putten et al. (2019) focused on complementary, demand-side mechanisms, exploring ways that small or non-target species might still add value. While their analysis strictly focused on economic value, we illustrate an additional demand-side mechanism that addresses food security concerns by taking a social value perspective. We describe one of the longest running bycatch donation programs of its kind in North American that might serve as a model, in particular for Europe, as it addresses the challenge of discarding finfish at sea via implementation of its new discard ban.

In federal waters off Alaska, Pacific halibut and salmon are occasionally caught incidentally using trawl gear, the only way to profitably target some groundfish species. Halibut and salmon are designated as prohibited species catches (PSC). These PSC are the targets of other fisheries and must be avoided while fishing for groundfish; groundfish fisheries are not allowed to retain or sell them and all PSC must be discarded and returned to the sea whether dead or alive (with minimum harm if alive), except when retention is required or authorized by other applicable law (North Pacific Fishery Management Council (NPFMC), 2019). Trawl catcher vessels targeting groundfish do not have the ability to sort their catches until the catches are offloaded at shoreside processing plants so there is little chance of PSC surviving long enough to be released alive from shoreside operations. Additionally, many of the PSC that are caught by catcher-processors or delivered to floating processors at sea do not survive (many halibut are released alive on bottom trawl catcher processors). Extensive observer coverage reduces the likelihood of unmonitored discarding and all PSC are counted by observers, with systematic sampling programs for biological data.

We examine the issue of PSC in the North Pacific in the context of the mitigation hierarchy, which seeks to minimize and offset the impacts from human activities (Arlidge et al., 2018). The North Pacific Fishery Management Council (NPFMC) and the Alaska seafood industry have a long history of cooperative efforts to reduce bycatch and to mitigate potentially deleterious effects on both ecosystems and other fisheries, while still meeting harvest goals. Such mitigation efforts apply operational modifications to fishing (e.g., bycatch limits, gear modifications, time-area closures) but after bycatch has been eliminated to the extent practicable, some PSC will inevitably remain. Historically, all PSC in Alaska was discarded at sea to avoid any incentive for trawlers to encounter such species. However, U.S. decision-makers agreed that some of these prohibited finfish need not be banned from human consumption altogether. In Alaska, trawl-caught salmon and halibut can contribute to the nation's food security by way of the Prohibited Species Donation (PSD) program, which allows for the donation of PSC through food banks. This donation of PSC provides a type of offsetting for the impacts of bycatch by minimizing the waste of the fish whose bycatch was unavoidable. We first describe the problem of salmon and halibut PSC and the efforts to avoid and minimize them. We then describe the PSD program as the last in a series of efforts to avoid the waste of seafood resources.

Salmon Prohibited Species Catch

Salmon play a vital economic, cultural, and dietary role for Alaska communities and concerns over salmon bycatch (e.g., Ianelli and Stram, 2015; Murphy et al., 2016) and declines of some salmon populations (e.g., Murphy et al., 2013; Schindler et al., 2013) have been pervasive for decades. Most salmon PSC is incidentally caught by fleets targeting walleye pollock using mid-water trawls. While PSC numbers are what count against bycatch limits, the ratio of bycatch to target catch in this fishery is quite low. In the past few decades, salmon PSC in Alaska has averaged about 200,000 fish annually from an average pollock catch of more than 1.3 million metric tons (on average, 0.15 and 0.18 salmon t^{-1} pollock for the Bering Sea and Gulf of Alaska, respectively, from 1991 to 2019). In the Bering Sea, Chinook (*Oncorhynchus tshawytscha*) and chum salmon (*O. keta*) accounted for approximately 10 and 90% of the salmon PSC, respectively from 2011 to 2019 (National Marine Fisheries Service (NMFS), 2020a,b). Recently in the Bering Sea, about one-quarter to one-half of Chinook salmon PSC has consisted of fish that originated from Bering Sea rivers (e.g., Guthrie et al., 2019b), while the majority of chum salmon PSC originated from Asian rivers or hatcheries (e.g., Whittle et al., 2018). In the Gulf of Alaska, fewer salmon are bycaught due to substantially smaller scale trawl fisheries; such catches are predominantly Chinook salmon originating from rivers and hatcheries in British Columbia and the U.S. West Coast (Guthrie et al., 2019a).

Salmon PSC has declined substantially in recent years, likely due to mitigation efforts and regulations that further limit bycatch. Efforts to mitigate salmon bycatch in the Bering Sea and Gulf of Alaska (Gisclair, 2009; Stram and Ianelli, 2015) have relied on cooperation and collaboration among agencies, the seafood industry, and Alaskan communities. These efforts include, but are not limited to gear modifications (e.g., salmon excluder devices, e.g., Gauvin et al., 2013), fixed time/area and rolling hotspot closures (Haflinger and Gruver, 2009; Little et al., 2014), and an extensive regulatory overhaul (National Marine Fisheries Service (NMFS), 2010; Stram and Ianelli, 2015). Particularly notable in the regulatory overhaul is a series of incentive plans that involve performance-based, tradable limits for salmon PSC with multi-year mechanisms to encourage long-term bycatch avoidance behavior (Sugihara et al., 2018) and a lower bycatch limit when western Alaska Chinook salmon returns are low. Thus, while salmon PSC still occurs in pollock trawl fisheries, the level of discards is likely much less than without extensive mitigation efforts.

Halibut Prohibited Species Catch

Pacific halibut (*Hippoglossus stenolepis*) in Alaska support a highly valuable multi-sector industry consisting of targeted commercial and recreational fisheries, charter fishing, and subsistence. The NMFS, NPFMC, and Pacific Fishery Management Council work with the International Pacific Halibut Commission (IPHC) to sustainably manage shared halibut stocks across west coast waters of the U.S. and Canada. The IPHC sets annual directed fishery catch limits while the NPFMC and other agencies (Fisheries and Oceans Canada, Pacific Fishery Management Council) set their own regional bycatch limits in fisheries targeting non-halibut species. Over the last few decades, concerns have focused on declining coast-wide halibut biomass (Stewart and Hicks, 2019) and the impacts of both discards in the target halibut fishery (e.g., under-size halibut) and bycatch from non-halibut fisheries (Martell et al., 2015). From 1992 to 2018, halibut bycatch across all IPHC management areas steadily declined from a high of 9,203 t to a low of 2,748 t. The estimated impacts of bycatch on the yield of targeted halibut catches have varied (Stewart et al., 2020) and simulations suggest substantial economic impacts to the halibut industry, though the actual value depends on fish prices and assumptions about fish movement (Martell et al., 2015).

The NPFMC and commercial fisheries industry have explored numerous mechanisms to mitigate halibut PSC. In 1999, bottom trawling was prohibited for targeted pollock fishing in the Bering Sea to mitigate halibut encounters (65 FR 31105). In 2011, Amendment 80 to the Bering Sea-Aleutian Islands groundfish management plan restructured and rationalized the mixed-species bottom trawl fishery, which primarily targets rock and yellowfin sole. In combination with the restructuring of target species quotas, the formation of cooperatives (which pre-dated Amendment 80), gear modifications, deck sorting (release of live fish which are deducted from bycatch tallies), intra-cooperative penalty structures, avoidance of night sets, and fixed and rolling hotspot closures, halibut PSC has remained below limits (See Abbott et al., 2015 and Holland, 2018 for broader discussions). In 2016, halibut PSC limits were reduced by as much as 25% for some fishery sectors (trawl and hook-and-line) in Alaska, bringing total PSC limits to 3,515 t in the Bering Sea and 1,972 t in the Gulf of Alaska (Amendment 111; 81 FR 24714). Meanwhile, the NPFMC and the NMFS are developing dynamic PSC limits that are based on halibut abundance to further mitigate impacts to stocks.

Prohibited Species Donation Program

The PSD program was developed by NMFS and the NPFMC to minimize the waste of valuable fish protein associated with bycatch by creating a regulatory framework through which unavoidable PSC can be donated to hunger relief organizations.

The PSD program authorized donations of salmon PSC in 1993, first as a pilot program, and in 1996 through Amendments 26 and 29 to the Fishery Management Plans for Groundfish in the North Pacific (61 FR 38358). In 1998, a pilot program included Pacific halibut donations and in 2000, the halibut program was reauthorized (Amendment 50; 63 FR 32144). Participation in the PSD program is voluntary and fishermen and seafood processors can enroll or leave the program at will. The industry must remove the head and guts from any donated fish (which are not tax-deductible) and store the fish in a manner that is fit for human consumption. Industry participants are allowed to process the head and guts for fish meal and oil, which can be subsequently sold (69 FR 52609), though such earnings are negligible. Any costs incurred by industry participants in handling the fish are born solely by them and are not reimbursed. Annual participation from the seafood industry varies slightly; in 2018, participants included 136 catcher boats, 11 shoreside processors, 38 at-sea processors, and two re-processors in Washington State (to inspect, trim, steak, and re-pack).

The PSD program provided the regulation to allow PSC donations to food banks, but it did not establish the mechanism (financial or logistical) for distribution. The program allows NMFS to authorize a distributor of PSC donations. Distributors must apply to NMFS and meet criteria for recordkeeping, reporting, food standards, storage, and distribution. Since inception of the PSD program, SeaShare has been the only applicant, and thus, the only authorized distributor of PSC.

SeaShare

SeaShare is a non-profit, donation-funded organization whose mission is to distribute valuable seafood protein to economically disadvantaged individuals across Alaska and the United States. During its 26 years, SeaShare has grown in scope and impact, and it distributes both PSC and target species (e.g., groundfish) seafood donations through Feeding America's (feedingamerica.org) national network of food banks. The logistical and financial burden of processing, transporting, certifying, and distributing seafood donations falls on SeaShare, which in turn relies on voluntary partnerships and financial support from fishermen and seafood processors who want to improve nutrition in Alaska and reduce waste. With the help of these partners, SeaShare has installed freezers in remote Alaska communities, enrolled freight donors, and qualified additional food banks to receive donated fish. Since 2004, SeaShare has distributed more than 2,386 finished metric tons of salmon and 276 finished metric tons of halibut (**Figure 1**). In total, these donations exceed 2,662 t of seafood, nearly 23.5 million servings (1.82 servings kg^{-1}).

Processing, shipping, storing, and distributing donations to Alaska's coastal and interior villages is complex, expensive, and requires extensive partnerships. SeaShare receives frozen donations primarily from fishermen and processors in Dutch Harbor, Akutan, and Kodiak, Alaska (**Figure 2**). The processors that donate salmon and halibut do not necessarily have the capacity to process these fish. Instead, much of the frozen donations are transported to Washington State, where they are processed (inspected, trimmed, steaked) for final distribution back to Alaska or in the contiguous United States. SeaShare facilitates this processing at a cost of approximately \$0.18–0.22 kg^{-1} using funds raised through grants and donations. Some of the seafood donations on Kodiak Island are processed and

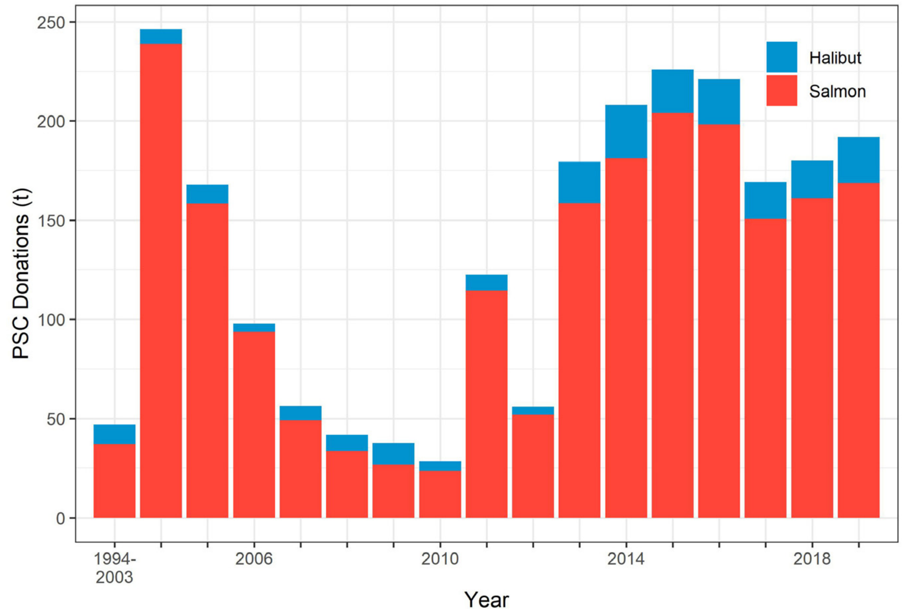

FIGURE 1 | Weights of annual salmon and halibut donations through the Prohibited Species Donation program. The 1994–2003 data are only available in aggregate form so these data are presented instead as an approximate annual average over this period (for halibut, from 1998 to 2003).

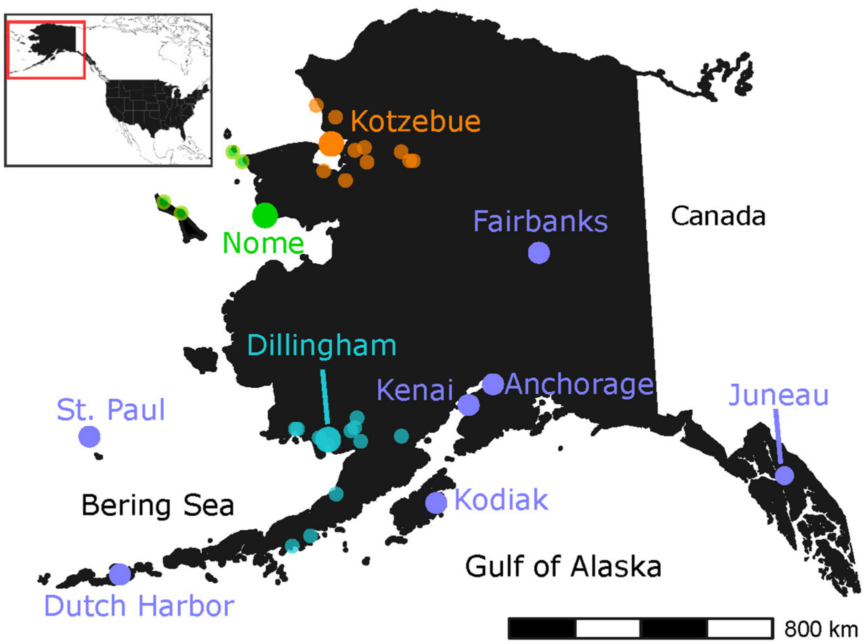

FIGURE 2 | Alaska food banks and food distribution center locations. Large purple circles are locations of food banks that receive donations directly from SeaShare. Smaller colored circles are food banks or communities that receive donations from the distribution locations denoted by the larger circles of the same color.

distributed locally without the extra step of transportation to the lower 48. In Alaska, there are several larger towns with food banks and/or the requisite infrastructure that receive donations directly from SeaShare or through SeaShare's partners (**Figure 2**). Meanwhile, other locations act as regional distribution hubs (e.g., Kotzebue, Nome, Dillingham) and SeaShare has invested in additional infrastructure (e.g., frozen storage) in several such locations. Leveraging a partnership with the United States Coast Guard, nearly 54.4 t of halibut have been flown to Kotzebue and Nome, two hub communities, since 2013. Much of this food is subsequently distributed to smaller towns and villages. This complex supply chain includes nearly 40 communities across

Alaska that receive seafood at no cost to them, in addition to some non-Alaska U.S. communities.

DISCUSSION

In Alaska, salmon and halibut have strict bycatch limits that can lead to fishery closures if exceeded. As the targets of other fisheries, any market-based distribution of these catches could create conflict with their respective target fisheries. However, because salmon and halibut are relatively expensive seafood, donation to food banks provides them to people who could not otherwise afford them. This avoids competition for the consumer market of these products, minimizes waste, and serves a population in need. This donation model may not work in every circumstance. For example, in the Pacific whiting fishery off the U.S. west coast, regulation allows donation of some salmon bycatch but logistics have prevented SeaShare from making donation programs cost-effective. However, regulation supports future creative efforts should the situation change.

No matter how creative fishermen become, some unwanted catches, and thus, some waste, will still exist (e.g., prohibited species are still discarded under the Landing Obligation). However, by allowing flexibility for bottom-up approaches, new opportunities to minimize waste may arise. The PSD program did not create a top-down bycatch distribution or donation strategy; rather, it removed a regulatory barrier that allowed creative solutions to improve social value and minimize waste. This idea of facilitating flexibility and creative solutions also lies at the foundation of the European Landing Obligation, in which a ban on discards is expected to drive fishers toward creative ways to make fishing more selective (Borges et al., 2016). Rochet et al. (2014) framed the policy as "an obligation for people to find solutions to reduce discards."

In the case of market discards, one driver of seafood waste is a lack of market demand or value (Van Putten et al., 2019). Thus, Iñarra et al. (2019) presented ways to add value to unwanted catches, framed around a conceptual model from the EU Directive on Waste (European Parliament Council, 2008). Iñarra et al. (2019) describe a hierarchy of potential fates for unwanted catches, ordered by decreasing economic value, with human consumption at the top followed by bio-products, animal feed, industrial uses, energy production, agronomy (compost), and finally, disposal. Their proposed decision tree for prioritizing potential fates of catches considers the environmental impacts (CO_2 emissions and water usage) associated with downstream production. Environmental impacts certainly have economic costs, but the framing of such considerations seems as much about social value as it does the economic value. In this context, the valorization of unwanted catches could more explicitly include social value. For example, what are the trade-offs between the economic value of catches that become compost relative to the social value of feeding people.

In 2015, the United Nations adopted a set of Sustainable Development Goals, including improvement of global food security and nutrition. Even in developed nations, an estimated 60 million people annually rely on food banks (Gentilini, 2013), making a strong case for programs that could simultaneously address food insecurity while also reducing amounts of food that are wasted because of regulatory or market barriers. SeaShare fills a critical nutritional need for protein, especially in Alaska, where more than 14% of the population and nearly 20% of children are food insecure, or lacking consistent access to safe, sufficient, and nutritious food (Feeding America, 2014). Most healthy adults require at least 50–70 g of protein per day (Institute of Medicine, 2005) and in a 2014 survey of food donation recipients, 54% of respondents listed protein (meat or seafood) among their most desired donated food items. Additionally, 81% said that a strategy for coping with food insecurity was to purchase less expensive and less healthy foods instead of healthier, yet more expensive protein (Feeding America, 2014). Thus, seafood donations meet a nutritional need that may be otherwise cost prohibitive for recipients.

In comparing global discard approaches, Karp et al. (2019) made a distinction between developed countries with generally low levels of catch utilization (high potential for waste) vs. developing countries with generally fewer discards and greater catch utilization (less waste). While higher levels of utilization may not necessarily yield direct conservation benefits for captured fish, the idea can still be framed around the mitigation hierarchy (Arlidge et al., 2018). The first three steps of the hierarchy seek to *avoid*, *minimize*, *and restore* human impacts at the location of potentially harmful activities, as do many of the bycatch and discard efforts described here (e.g., time-area closures, gear modifications, bycatch limits). Meanwhile, the fourth step of *offsetting* impacts typically occurs offsite and does not necessarily benefit bycaught stocks directly. Distribution of would-be discards has a direct benefit on human health while also offsetting the demand for protein (or less healthy alternatives) that might require additional production. Thus, donation of seafood discards creates a type of social offsetting by enabling food banks and the seafood industry to provide more healthy seafood protein with a greater environmental and cost efficiency.

DATA AVAILABILITY STATEMENT

Publicly available datasets were analyzed in this study. These data can be found here: https://github.com/jordanwatson/Prohibited_Species_Donation_Program.

AUTHOR CONTRIBUTIONS

JW designed study. All authors contributed to writing of the manuscript.

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Mitigating Bycatch: Novel Insights to Multidisciplinary Approaches

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Fisheries bycatch conservation and management can be analyzed and implemented through the biodiversity mitigation hierarchy using one of four basic approaches: (1) private solutions, including voluntary, moral suasion, and intrinsic motivation; (2) direct or "command-and-control" regulation starting from the fishery management authority down to the vessel; (3) incentive- or market-based to alter producer and consumer behavior and decision-making; and (4) hybrid of direct and incentive-based regulation through liability laws. Lessons can be learned from terrestrial and energy conservation, water management, forestry, and atmospheric pollution measures, such as the use of offsets, tradeable rights to externalities, and liability considerations. General bycatch conservation and management principles emerge based on a multidisciplinary approach and a wide array of private and public measures for incentivizing bycatch mitigation.

Keywords: bycatch, biodiversity mitigation hierarchy, inter-disciplinary, conservation, regulation

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CONTEXT: THE BYCATCH ISSUE

Bycatch refers most often to those species incidentally taken in fishing operations aimed at other (target) species. Bycatch in this paper refers to species accidentally caught other than the target species, brought on board, dead or alive, and that can therefore be either released alive, discarded dead, or landed. Bycatch can be other finfish (including undersized target species), protected species (fishes, sea turtles, marine mammals, and seabirds), live corals, or sponge reefs. We include habitat impact (Holland and Schnier, 2006; Driscoll et al., 2017) with bycatch (hereafter simply bycatch). Central to this paper is the fact that bycatch species and living habitats include vulnerable, threatened, endangered, protected or otherwise emblematic species for which the take should be minimized. Bycatch in this paper is extended to include habitat impact.

Economists classify bycatch into two types. The first type is non-target species that are commercially harvested and receive a market price, but harvest is not at the ecologically-economically optimum level due to size, age, or contribution to ecosystems. The market price does not capture the full costs of foregone biodiversity and ecosystem services, including impacts on population growth and food webs, because the bycatch is underpriced. Examples could include a commercially landed fish species caught before reaching sexual maturity or a plankton foraging fish contributing to the food web. The second type of bycatch is threatened, endangered or protected species that are prohibited for retention. Examples include seabirds and sea turtles ensnared in pelagic longline gear and marine mammals caught in drift gillnets. Because these species do not have a formal market, the bycatch is 'unpriced,' even though the species do have non-market value through their contribution to biodiversity, the ecosystem, and existence.

This paper, part of a special issue on bycatch and its mitigation, develops a broad-based conservation framework and suite of policy instruments to address bycatch, drawn from marine and terrestrial biodiversity conservation, pollution, and climate mitigation examples. While the ecosystem approach to fisheries has largely focused on harvest strategies, the future of fisheries management will also benefit from the conservation and regulatory framework described in this paper. Most, if not all, conclusions on bycatch mitigation that have been based on harvest strategies can be broadly applied to environmental protection of living habitats and to marine biodiversity conservation in general, and therefore, to complete implementation of the ecosystem approach to fisheries. The focus of this paper and the associated case studies pertains to commercial fisheries, but when possible, artisanal fisheries are also considered.

The paper develops this broad-based conservation framework through three specific objectives. First, using the conservation biology mitigation hierarchy as a broad framework, the paper provides new insights into the suitability of the no net loss objective. Second, the paper extends the previous bycatch management literature by developing four regulatory categories for fisheries bycatch mitigation: private, direct regulation (top-down and command-and-control), incentive (market-based), and hybrid. The previous literature, reviewed below, largely focused upon reviewing different incentive-based approaches, overlooked private and hybrid approaches, and did not establish a systematic, broad approach across all four categories.

Third, the paper breaks new ground by developing the broader context for the four regulatory categories that impact the choice, design, and effectiveness of each regulatory approach. That is, the choice of bycatch mitigation approach and specific policy instruments within each approach does not occur within a vacuum, but rather within a specific fishery context. The type of fishery – gear, target and bycatch species, fleet and vessel characteristics, scale of production, markets, monitoring and enforcement capacity, quantity and quality of available information, potential for bycatch-reducing technology, transboundary species, regulatory and management structure, and other factors affect the approach to bycatch management and appropriate policy instruments.

The paper is organized as follows. Section "Mitigation Hierarchy As a Framework" presents the mitigation hierarchy framework, including discussions on no net loss as an objective and bycatch mitigation impacts that account for impact equity. Section "Basic Regulatory Approaches to Bycatch Mitigation" presents four basic bycatch mitigation approaches. Section "Incentive-Based Policy and Policy Instruments" addresses the incentive-based approach, including pricing bycatch and direct and indirect incentives. Section "Direct and Incentive-Based Regulation" compares direct and incentive-based regulation. Section "Interactions Between Private Solutions and Incentive-Based Policies" develops the potential interactions between private solutions and incentive-based regulation. Section "Broader Context Shaping Choice and Performance of Policy Approach" develops the broader context shaping the choice and performance of the different policy approaches. Section "Equity and Fairness" raises the issues of equity and fairness in bycatch mitigation.

MITIGATION HIERARCHY AS A FRAMEWORK

Bycatch mitigation and the ecosystem-based approach to fisheries management are more than biological and technical issues. They also include modifying behavior and decisions made by producers (vessel owners/operators and crew, processors, distributors, retailers) and consumers to account for the biodiversity and ecological impacts of their decisions.

The mitigation hierarchy (MH) (Business and Biodiversity Offsets Programme [BBOP], 2012) provides an overarching conservation framework for marine biodiversity conservation in general, and is particularly useful for addressing bycatch (Lent and Squires, 2017; Milner-Gulland et al., 2018; Squires and Garcia, 2018; Squires et al., 2018; Booth et al., 2019a; Arlidge et al., 2020b). The MH provides a framework for defining measurable goals and structuring available knowledge of potential management measures to achieve these goals (Milner-Gulland et al., 2018; Booth et al., 2019a). The MH in the bycatch context is as follows: (1) Avoid bycatch; (2) Minimize bycatch when it cannot be avoided; (3) Restore or rehabilitate bycatch on-site when it cannot be minimized; and (4) Implement biodiversity offsets for the same species and stock (or habitat) as a last resort to address the residual from the first three steps.

The first three steps in the MH, generally implemented in that order, constitute what could be called conventional *conservatory mitigation* (Squires and Garcia, 2018; Squires et al., 2018). They aim to restore the biodiversity to its pre-disturbance baseline or No Net Loss in biodiversity (NNL) or any other agreed "healthy state." (Here we abstract from definition, measurement, and the actual baseline chosen, including the issue of "shifting baselines," Kahn and Friedman, 1995; Pauly, 1995; Papworth et al., 2009). Avoidance entails measures to reduce the probability of encounter between potentially harmful gear and potential bycatch by separating fishing activity from individuals or stocks of concern (Arlidge et al., 2020b). Examples include no-fishing areas, deployment restrictions on fishing gear, and dynamic or static time-area closures (Hobday et al., 2013; Kaplan et al., 2014; Little et al., 2015; Maxwell et al., 2015; Lent and Squires, 2017; Milner-Gulland et al., 2018; Squires and Garcia, 2018). Fisheries subject to avoidance through time-area closures include the Hawaii shallow-set pelagic longline (Curtis and Hicks, 2000; Chakravorty and Nemoto, 2001) and California drift gillnet swordfish fishery (Janisse et al., 2010; Gjertsen et al., 2014), the tropical tuna purse seine fishery of the Eastern Pacific Ocean (through "El Coralito") (Hall and Roman, 2013), and the southern and eastern scalefish and shark fishery (Australian Fisheries Management Authority [AFMA], 2009).

Step 2 in the MH, minimization, reduces bycatch through, for example, reductions in effort, technology standards, or bycatch reducing technological change that alters selectivity. Examples include use of circle rather than J-hooks with shallow-set pelagic longlines targeting swordfish to minimize sea turtle bycatch (Watson et al., 2005), bycatch reduction devices for demersal trawl gear (Wakefield et al., 2016), nylon leaders for shark bycatch in pelagic longline fisheries (Ward et al., 2008; Booth et al., 2019b), hook depth (Shiode et al., 2005), type of hook and position of hook in water column for elasmobranch species in pelagic and coastal fisheries (Afonso et al., 2011), Tori lines for pelagic longline fisheries seabird bycatch (Gilman et al., 2005, 2019), the design of fish aggregating devices (FADs) to reduce bycatch of pelagic sharks (Dagorn et al., 2012b; Restrepo et al., 2017), or targeting of bigger tropical tuna schools (Dagorn et al., 2012a). Other examples include use of narrower nets in demersal gillnet fisheries to reduce sea turtle bycatch rates (Price and Van Salisbury, 2007; Gilman et al., 2010) and longlines raised off the bottom for sharks and rays (Favaro and Côté, 2015). Modification to gill net size and tension can increase selectivity of certain species and life-history stages with meshing and entanglement (Thorpe and Frierson, 2009; Harry et al., 2011). Turtle Excluder Devices (TEDs) minimize sea turtle bycatch by shrimp trawls (Crowder et al., 1994).

In the third step in the MH, restoration or remediation strategies facilitate live release of individuals, their safe return to the sea, and their post-release survival (Booth et al., 2019a). For example, line cutters to cut hooked sea turtles in pelagic longline fisheries reduce post-hooking mortality (Gilman et al., 2006; Gjertsen et al., 2010; Dutton et al., 2011). On-board handling of caught elasmobranchs before return to sea (Booth et al., 2019b) and at-sea release of dolphins while retrieving nets with Eastern Pacific tuna purse seine vessels (Hall and Roman, 2013; Hall et al., 2017; Gilman et al., 2019).

The last step of the MH aims to compensate for part or all of the residual impact that remains after implementing the first three steps by addressing impacts to the same bycatch species and stock, either in the same ecosystem or in the global ecosystem. Early discussion of fisheries bycatch and offsets are found in Worldfish Center (2004), Wilcox and Donlan (2007), Dutton and Squires (2008, 2011), Janisse et al. (2010), Pascoe et al. (2011), Gjertsen et al. (2014), and Quigley and Harper (2006) discuss salmon habitat offsets. Van Dover et al. (2014) discuss deep-sea environments. Examples include protection of sea turtle nesting sites (Dutton and Squires, 2008, 2011; Janisse et al., 2010; Dutton et al., 2011; Gjertsen et al., 2014) and sea bird rookery sites (Wilcox and Donlan, 2007; Pascoe et al., 2011) or purchases of gear by outside interests for fishers to minimize bycatch in fisheries from fleets outside the fishery subject to regulation (Gjertsen et al., 2010).

Is the No Net Loss (NNL) Objective Appropriate?

The MH aims for NNL or even a net gain in biodiversity, for the stock or population of bycatch species. NNL is sometimes Maximum Sustainable Yield (MSY), which is often required by law (Wolf et al., 2015; Squires and Garcia, 2018). In other instances, some predetermined baseline serves as the NNL.

No net loss in the absence of a required MSY, however, does not necessarily generate the optimum level of producer and consumer benefits (hereafter social welfare) when biodiversity, or more specifically bycatch and offsets, are considered as a public good (Kotchen, 1999; Vicary, 2000). Just to be clear, bycatch reduction is the public good that is provided by fishery operators, and similarly, offsets are a public good that are provided by fishery operators or other parties. A public good is one available to all society without diminishing the amount available and is typically privately underprovided since not all benefits can be captured by the provider.

No net loss equates to an *a priori* level of social welfare, or baseline, that only coincidentally gives NNL. Bycatch reduction, or more generally offsets and biodiversity, is not only a public good, but it is only one contributor to social welfare (which accounts for all market and non-market costs and benefits impacting society). For example, the NNL objective can lower social welfare by not conserving sufficient bycatch if the bycatch stock requires rebuilding. The NNL objective can also lower social welfare if it restricts target catch to such an amount that the foregone social benefits out-weigh the social benefits of bycatch mitigation to NNL. NNL can also lower social welfare when producer benefits are diminished due to lower commercial landings. Whether net consumer welfare decreases (from less consumption) or increases (from conservation) depends upon society's valuation of the public good component. NNL can further lower social welfare if it constrains offsets to a sub-optimal level (Kotchen, 1999). Both social welfare and bycatch can decline when the NNL objective requires unilateral bycatch reduction of a transboundary bycatch stock, lowering producer and consumer benefits in the country of regulation and shifting production of the target and bycatch species to other, unregulated countries with higher bycatch rates (Helvey et al., 2017). Such conservation, production, and consumption "leakages" occurred in the Hawaiian shallow-set pelagic longline fishery for swordfish when complete avoidance (closure) transferred production to unregulated high-seas swordfish fleets with higher bycatch rates (Sarmiento, 2006; Rausser et al., 2009). The increased production was imported back into Hawaii as an inferior frozen rather than fresh product, thereby lowering consumer welfare, and the reduced Hawaiian catch and processing lowered producer welfare in Hawaii.

The NNL objective and its implementation also implicitly have distributive impacts that can disproportionately impact local and lower-income fishers (Booth et al., 2019a; Griffiths et al., 2019a,b). Avoidance, for example, can disproportionately lower such fishers' production, contributing to an inequitable impact and lower social welfare, particularly if society accords such fishers greater social importance. Compliance can then suffer, and without sufficient monitoring and enforcement, bycatch can even increase, or effectiveness of offsets could decline say through lower local community monitoring and protection. NNL can be subject to the requirement that people – especially project-impacted – are no worse off, and preferably better off, i.e., the Pareto Principle (Griffiths et al., 2019a,b). The overall bycatch objective can account for fishers, especially small-scale and lower-income, more dependent for income and food security, to have a net negative impact on the bycatch stock (satisfying no worse off), provided the gains and losses across all fisheries combine to achieve the bycatch objective (Booth et al., 2019a). An analogous issue arises with MSY compared to Optimum Yield that also accounts for the greatest overall benefit to the national economy and considers qualification by relevant economic, social, or ecological factors (OECD, 2006).

In lieu of NNL, the bycatch reduction objective can be defined in net social welfare gain, population stability, population recovery, sustainability or simply catch minimization, depending on what is practical given budgetary and operational constraints (Booth et al., 2019a). Much like Optimum Yield, the goal can be further broadened to account for ecological factors, overall social welfare, and distributional impacts on producers and consumers. For example, a target or even bycatch species may constitute the major protein source for low-income consumers, and the impact of lower fish consumption can adversely and significantly impact their diet, potentially require substitution to higher-cost protein sources (if at all), thereby lowering consumer welfare. More broadly, food security and poverty reduction (Béné et al., 2016; Booth et al., 2019a) require incorporation into the bycatch reduction goal, particularly in low-income countries and with artisanal fisheries. An unanswered question, analogous to single-species management, is whether a sustainability constraint of NNL pertains to each species, in which case weak stock-species management issues arise for multispecies (target and bycatch) species.

In sum, NNL may or may not be the appropriate objective for society, and each case under consideration requires weighing the pros and cons for a bespoke approach. Moreover, the issue is not just NNL "level" but the conservation method by which it is achieved and the equity and fairness of the process and distributive impacts.

Least-Cost Bycatch Reduction

Least cost or cost-effective bycatch reduction minimizes the cost of a given level of bycatch reduction (Dutton and Squires, 2008; Gjertsen et al., 2010, 2014; Pascoe et al., 2010, 2011; Gjertsen, 2011; Innes et al., 2015; Lent and Squires, 2017; Squires and Garcia, 2018; Squires et al., 2018; Booth et al., 2019a; Arlidge et al., 2020b). It can be extended to the entire MH (Squires and Garcia, 2018; Squires et al., 2018; Booth et al., 2019a; Arlidge et al., 2020a,b). The least-cost MH minimizes costs across and within MH steps and bycatch reduction channels. The least-cost MH allows the maximum possible bycatch reduction for a given conservation budget and mitigates fisheries bycatch consistent with given targets, guidance in the Law of the Sea, Regional Fisheries Management Organizations, national fishery management authorities, and measures under the Convention on Biological Diversity. Through least-cost implementation, the MH also enables socioeconomic trade-offs to be explicitly incorporated into decisions (Squires and Garcia, 2018; Booth et al., 2019a).

Equitable Bycatch Reduction

Bycatch reduction can be extended to account for equity through the use of social distribution (welfare) weights ω_i (Little and Mirrlees, 1974). In principle, the relevant parties are not just those regulated but all parties that are impacted, such as low-income consumers deprived of a food source and food security. Impacted parties also include fishers outside of the regulated fishery that are impacted by spillovers from the regulated fishery, such as regulated fishers impacted by avoidance that now fish in another fishery. In practice, the parties of concern may be limited to those of the regulated fishery.

Social distribution weights for Party i, ω_i, and equity-adjusted bycatch mitigation and equity-adjusted least-cost bycatch mitigation are defined as follows. Let $\omega_i = \left[\frac{\bar{Y}}{Y_i}\right]^{\eta}$ for Party i, where Y_i denotes per capita income for i, $\bar{Y}$ denotes mean per capita income of all parties N, and η denotes a progressivity parameter (elasticity of social marginal utility of income). $\eta > (<)\ 1$ makes ω_i more (less) progressive. Let B_j/C_j denote the least-cost ratio for bycatch B_j in MH step j divided by the cost of bycatch mitigation in step j C_j, where the ratio is equalized across steps for least-cost bycatch mitigation: $B_j/C_j = B_k/C_k$ $j, k = 1, 2, 3, 4,\ j \neq k$ (Squires and Garcia, 2018). Then, $\omega_i \left[B_j/C_j\right]$ gives greater weight to least-cost bycatch reduction for income groups in step j in which $Y_i < \bar{Y}$. Equalizing across all four steps $j, k,\ j \neq k$, gives: $\omega_i \left[B_j/C_j\right] = \omega_i \left[B_k/C_k\right]$ Without least-cost consideration, bycatch mitigation with equity consideration for lower-income or close proximity groups, as in Griffiths et al. (2019a,b) is $\omega_i B$, where here B denotes biodiversity impacts in general.

Offsets

Offsets, when applied prior to the final residual step of the MH, are a public good (Kotchen, 1999) that become an incentive-based policy substitute for avoidance and minimization (Lent and Squires, 2017; Squires and Garcia, 2018; Squires et al., 2018; Milner-Gulland et al., 2018). For eligible species, offsets can be used as an incentive-based policy instrument earlier in the mitigation hierarchy to compensate for any negative impacts through off-site conservation actions that improve the status of the affected bycatch (same species and stock) elsewhere. Bycatch mitigation costs should fall, freeing up scarce conservation budgets for other needs, or benefitting fishers as the foregone target catch and revenues should fall as less avoidance may be required.

BASIC REGULATORY APPROACHES TO BYCATCH MITIGATION

The fundamental regulatory approach to bycatch mitigation, and biodiversity conservation in general, through the MH may be binned into four general categories:

(1) Private solutions;
(2) Direct regulation;
(3) Incentive-based measures;
(4) Hybrid solutions.

Each of these approaches will be explored in turn in the sections below.

Private Solutions

Private solutions include voluntary and private negotiation between producers incurring bycatch and other private parties, intrinsic motivation, and moral suasion. Intrinsic motivation refers to behavior/activity coming from within the person for its own sake rather than the desire for some external reward. Intrinsic motivation contrasts with extrinsic motivation, which is engaging in a behavior in order to earn external rewards or avoid punishments (Gneezy et al., 2011; Young, 2015). Intrinsic motivation includes social and personal norms of conservation and altruism.

Moral suasion can be an important instrument to align individual and public interests (Romans, 1966). Monetary costs of moral suasion are typically small, they are quickly implemented, and can complement economic incentives or direct regulation (Bos et al., 2020). By affecting social norms or adherence to them, moral suasion is expected to contribute to bycatch reduction and to increase compliance.

Examples include negotiations and voluntary agreements such as the United States west coast groundfish fishery case discussed below. [See Gneezy et al. (2011), Bowles and Polanía-Reyes (2012), Kotchen (2013), Segerson (2010, 2013), Young (2015), and Farrow et al. (2017) for general discussions of voluntary and private environmental regulation]. Credible threats, such as formal public regulation, can incentivize voluntary bycatch reduction (Segerson, 2013). Credible threats of embargoes and trade measures can also be effective, such as with the tuna–dolphin and shrimp–sea turtle issues (Joyner and Tyler, 2000).

Another example is the voluntary program reducing bycatch of river herring, blueback, herring, and American shad in the northwest Atlantic mid-water trawl fishery targeting Atlantic herring and Atlantic mackerel through an industry, state government, and university partnership (Bethoney et al., 2017). Potential public regulation provided the credible threat that motivated the voluntary program to stave off such regulation. In the United States, a voluntary program facilitates the donation to foodbanks of salmon bycatch in the Bering Sea–Aleutian Island trawl fishery (Clucas, 1997; Watson et al., 2020). The Alaskan groundfish fishery employs a voluntary program to reduce halibut bycatch (Fina, 2017).

Bycatch policy can also be influenced by intrinsic motivation that is inherent in societies, communities, and individuals. Intrinsic motivation may be particularly important in economies that are less market-centric and are more characterized by non-economic relationships. Intrinsic motivation may be especially important for bycatch in small-scale and artisanal fisheries. Social networks can impact intrinsic motivation (Alexander et al., 2020; Arlidge et al., 2020a,b). Social norms can be classified as folkways, mores, taboos and laws (Young, 2008, 2015). For example, customary taboos that temporarily close coral reef areas to fishing have long been practiced in Solomon Islands (Foale and Manele, 2004; Foale et al., 2011). There is disagreement on whether fishing taboos (and customary marine tenure) are primarily intended for management between social groups or to sustain food security from fisheries (Johannes, 1978). Taboo on a clan reef may be declared as a mark of respect for the death of a prominent clan member, to protect sacred sites, or to prepare for a feast by allowing the short-term replenishment of fish (Abernethy et al., 2014).

Social norms may impact compliance to bycatch regulation, whether direct or incentive-based.[1] Bycatch-regulated vessels may be motivated to comply with regulations, and even to go beyond literal compliance, not only fear of legal sanctions but also by social pressures and norms. (See Thornton et al., 2009 for trucking). Along similar lines, voluntary bycatch reduction may arise due to credible regulatory threats, but additional compliance may arise due to social norms. Larger and more commercialized seafood firms may respond more positively to bycatch regulation than small firms, since they are more visible and more closely scrutinized by regulators, consumers, and advocacy groups and are more concerned with brand and social reputation.

Direct Regulation

Direct regulation, also called top-down or command-and-control, focuses on mandating specific behavior through standards on technology, process, and performance which address in particular the avoidance and minimization steps of the MH. A standard is a limitation on behavior on a producer, such as a performance standard on the outcome of production as with a catch quota or limit (Helfand, 2013). Bycatch can be tackled through top-down, direct regulation by a fishery management authority, government, or Regional Fisheries Management Organization. Direct regulation can be accomplished through combination of technology, process, and performance standards.

A technology standard specifies bycatch reduction technologies or production processes that producers must implement for avoidance or minimization. Examples include the prohibition of sundown sets to reduce dolphin mortality when setting on dolphins to capture large yellowfin tunas in the Eastern Pacific Ocean (Gjertsen et al., 2010; Hall et al., 2017), discarding offal on the opposite side of the vessel from which gear is released and required use of Tori lines on longline vessels (seabirds) (Gilman et al., 2014, 2016), selectivity requirements for gear such as mesh size, use of circle hooks with mackerel-type bait rather than J-hooks with squid bait (sea turtles) (Kerstetter and Graves, 2006; Reinhardt et al., 2017), bycatch reduction devices on trawl nets (Melli et al., 2020), and the use of pingers on drift gillnets (marine mammals, sea turtles) (Gilman et al., 2010). Technology standards may change as technology changes. One critical issue with technology standards is that they tend to remove incentives for fishery operators to find other ways to reduce bycatch and they can also freeze bycatch-reducing technology in place.

A process standard requires that vessels satisfy limits or conditions on the process of fishing to achieve avoidance or minimization. Examples include bycatch avoidance

[1] An reviewer suggested that the role of social license to operate can be included in the discussion on intrinsic motivation and especially social norms. Bycatch is often associated with reduced social license, which in turn affects social norms. Social license (to operate) can be defined as existing when a project has the ongoing approval within the local community and other stakeholders, ongoing approval or broad social acceptance and, most frequently, as ongoing acceptance (Prno and Slocombe, 2012). Social license thus constrains vessels and supply chain firms to meet the expectations of society and to avoid activities that societies deem unacceptable rather than compliance with legal requirements (Thornton et al., 2009). The relationship between social license to operate and social norms is complex and beyond the scope of this paper.

through time-area closures, including Marine Protected Areas (Di Lorenzo et al., 2020), such as the time-area closure to reduce bycatch of harbor porpoises in the Gulf of Maine sink gillnet fishery (Murray et al., 2000) and another that is aimed to reduce bycatch of Atlantic cod and yellowtail flounder in the Georges Bank scallop fishery (Keith et al., 2020). Limits on vessel size or trip length or frequency are another process standard. Dynamic ocean management, such as the Hawaii Turtle Watch program (Howell et al., 2008), is another form of process standard (Hobday et al., 2013; Little et al., 2015; Maxwell et al., 2015).

A performance standard requires vessels meet a standard to minimize bycatch, such as a bycatch quota, while allowing the vessels to choose any appropriate method to meet that standard subject to inherent legal limits such as in the Pacific coast United States groundfish trawl fishery (Holland and Martin, 2019). New Zealand manages bycatch of Hooker's sea lions in the arrow squid trawl fishery through quotas on bycatch (Bache, 2003). Performance-based permits are a related policy option, involving issuing permits only to those who meet bycatch standards (Gjertsen et al., 2010).

Standards can be uniform across all producers and consumers or differentiated by types of producers and consumers. Performance standards on bycatch, such as bycatch quotas, may be differentiated by vessel size class. For example, in the tropical tuna FAD fishery the major determinant of species and size caught is vessel's capacity (size and use of satellite buoys, echo-sounders, and supply vessels) (Guillotreau et al., 2011). Performance standards through avoidance can impact a broad range of fleets, generating diverse responses through fishing effort redistribution, such as with a closure of the United Kingdom Exclusive Economic Zone for a diverse group of French vessels (Dépalle et al., 2020). Another example is sea turtle bycatch quotas for Hawaii pelagic longline shallow sets (swordfish) but not deep sets (bigeye tuna). Differentiated standards can be designed to more closely match producers' ability to reduce bycatch, and thereby could lower costs of compliance.

Standards can be absolute or relative (Helfand, 2013). Standards are defined, either implicitly or explicitly, as a rate, such as per unit of time, area, effort, or catch. If the measure in the denominator is completely exogenous to the process, the standard is absolute, such as an absolute limit on bycatch or effort per unit of time (time is exogenous). An absolute standard limits total quantity of bycatch. A relative standard, also known as rate-based or intensity standard, is a standard per unit of catch, bycatch, effort, habitat, or other measure over which the regulated entity has some control. Thus, if the denominator can be controlled such as bycatch per unit of target catch or effort, the standard is relative and does not have a limiting total quantity. A relative performance standard example is bycatch per unit of input (effort), per unit of gear (e.g., per thousand hooks), or per unit of target species catch. If a relative standard applies equally to vessels of different sizes or for example bycatch rates – a uniform standard – a relative standard will require proportionately equal bycatch reduction from all vessels. A differentiated relative standard distinguishes the standard by vessel size class or some other distinguishing feature. The ratio defining the relative standard can be uniform or differentiated and can be adjusted over time.

In sum, regulatory policy instruments can be based upon the state of technology, performance or outcomes of the bycatch mitigation, or on the process of production, transportation, processing, and distribution. Technology standards tend to be easily understood and often are readily accepted. Process-centered policy instruments affect the choice and state of technology and the choice and use of inputs in production. Performance-based approaches generally provide producers greater flexibility in meeting bycatch reduction goals than those based upon process. Performance-based approaches tend to create stronger and more direct economic incentives because they directly address the desired policy outcome. Process-centered incentives are more indirect because only some of the inputs and practices are regulated, and the relationship between the regulated inputs and the expected outputs can be indirect and more uncertain. Process-centered incentives are consequently weaker because they are more indirect. Nonetheless, performance-centered policy instruments may be more difficult and costly to monitor and enforce than process-centered approaches, especially due to at-sea production.

Incentive- or Market-Based Measures

Incentive-based (also called market-based) measures place a price bycatch and thereby give residual bycatch a cost. This bycatch cost in turn is incorporated into the price of the catch of target species and any bycatch that already may be sold (and hence has an existing price). Pricing bycatch then increases the cost of production, which in turn incentivizes changes in producer and consumer behavior and decision-making to reduce the scale and mix of bycatch (Goulder and Parry, 2008).

Economic incentives, increasingly used to address pollution, climate change, terrestrial conservation, water, and energy efficiency, have potential for greater application to bycatch reduction and marine biodiversity conservation (Hall, 1996; Dutton and Squires, 2008, 2011; Gjertsen et al., 2010, 2014; Pascoe et al., 2010, 2011; Dutton et al., 2011; Segerson, 2011; Innes et al., 2015; Walmo et al., 2016; Lent and Squires, 2017; Milner-Gulland et al., 2018; Squires and Garcia, 2018; Squires et al., 2018; Booth et al., 2019a,b; Arlidge et al., 2020b). As this paper emphasizes, however, economic incentive-based bycatch mitigation is context-specific in its design, use and effectiveness. Incentive-based bycatch reduction can apply to any step of the MH, including offsets to address residual bycatch after the first three steps of the MH (Dutton and Squires, 2008; Pascoe et al., 2011; Lent and Squires, 2017; Milner-Gulland et al., 2018; Squires and Garcia, 2018; Squires et al., 2018; Booth et al., 2019a; Arlidge et al., 2020b).

Incentives can be positive or negative. Positive incentives reward producers for reducing bycatch, such as through an ecolabel, or subsidies which can be cash or in-kind and direct or indirect. An in-kind example is credits for days fishing or access to areas closed to fishing if certain bycatch reduction measures are implemented. Negative incentives penalize producers for bycatch, such as direct taxes (Wilcox and Donlan, 2007; Dutton and Squires, 2008, 2011; Gjertsen et al., 2010; Pascoe et al., 2010; Dutton et al., 2011; Segerson, 2011; Innes et al., 2015; Booth et al., in press) or in-kind and indirect, such as loss of access to a fishing ground or fewer days to fish. Full retention of bycatch forms an implicit tax, because it displaces target catch and revenue (Chan et al., 2014). Tax receipts applied to further bycatch reduction form a "double-dividend" bycatch tax (Dutton and Squires, 2008; Gjertsen et al., 2010; Pascoe et al., 2010; Booth et al., In press). As an example, the Federation of Seafood Harvesters (FISH), the industry association of the California drift gillnet fishery for swordfish, voluntarily initiated payments in the Fall of 2004 to the Asociacion Sudcaliforniana de Proteccion al Medio Ambiente y la Tortuga Marina (ASUPMATOMA), a Mexican conservation group, to aid their Pacific leatherback turtle (*Dermochelys coriacea*) recovery efforts (Janisse et al., 2010). FISH financed the payments for offsets by a voluntary tax. The three major United States tuna processors, operating through the International Seafood Sustainability Foundation (ISSF), voluntarily tax themselves US$1/ton of landed longline-caught tuna to finance offsets for sea turtle bycatch (Squires et al., 2018; Pakiding et al., 2020).

Individual transferable quotas (ITQs) for bycatch (Boyce, 1996; Bisack and Sutinen, 2003; Gjertsen et al., 2010; Hannesson, 2010; Singh and Weninger, 2014; Hall et al., 2017; Miller and Deacon, 2017) are another incentive-based policy instrument, where bycatch reduction and selling or otherwise transferring ITQs confers a benefit and positive incentive through revenues earned and buying ITQs to cover bycatch not covered by quota creates a cost and negative incentive. Trade of bycatch ITQs may be tailored and when applied to weak stock species each fisher may be required to hold sufficient rights to cover any bycatch (Miller and Deacon, 2017). An advantage to ITQs is that they counter the "race-to-fish" associated with an overall bycatch TAC.

Incentive-based policy instruments can be a combination of penalty-and-reward measures, a two-part policy instrument (Fullerton and Wolverton, 1999; Segerson, 2011; Kotchen and Segerson, 2019). A penalty (indirect tax), such as fewer fishing opportunities for not avoiding or minimizing, can be coupled with a reward (indirect subsidy), such as extra fishing opportunities for avoiding or minimizing (Segerson, 2011; Kotchen and Segerson, 2019). The penalty and reward do not have to be equal (Fullerton and Wolverton, 1999; Segerson, 2011; Kotchen and Segerson, 2019). On average over time if the penalties and rewards are correctly set, the quota should just be met and penalties should just match rewards (Fullerton and Wolverton, 1999; Segerson, 2011; Kotchen and Segerson, 2019). The Scottish credit scheme is designed to reduce cod bycatch through a penalty-and-reward system of days (Scottish Government, 2011). The Bering Sea pollock mothership fleet employs a voluntary credit scheme to reduce salmon bycatch (Mize, 2014).

A deposit-refund system for fishing gear and equipment, which can reduce "ghost fishing" is another example of these two-part policy instruments, which can also address the uncertainty about the bycatch reduction (Jensen et al., 2017).

Incentive-based policy can follow either the Polluter Pays Principle (PPP) or the User (Beneficiary) Pays Principle (UPP). Biodiversity offsets provide an example of PPP, since producers, who create the bycatch, must pay for the offsets. Conservatory offsets, applied earlier in the MH than compensation, are incentive-based (Squires and Garcia, 2018; Squires et al., 2018). Payments for ecosystem services provide an example where entities concerned about bycatch may be willing to pay those who cause the bycatch to reduce their bycatch (Bladon et al., 2016). The BPP tends to hold in international fisheries, since there is no global institution that requires countries to reduce bycatch (although a fleet's government could require bycatch reduction due to that country's own, internal reasons).

Hybrid Solutions: Bycatch Liability

Bycatch liability is a performance-based hybrid of direct and incentive-based regulation in which parties are held liable for bycatch exceeding some baseline. Bycatch liability is typically triggered in some period by an event or condition (Lodge et al., 2019). Liability requires determining the damages that have occurred, whereas most incentive-based policy instruments raise the cost of bycatch by enough to incentivize vessel operator decision-making and behavior to reduce bycatch.

Several key challenges arise. One is defining, measuring, and monitoring the triggers with relative ease and low costs. Another concern is the transactions and information costs of reaching and enforcing agreements. A final issue is the burden of proof that involves damage to a person, community organization, etc., establishing a direct causal link between their losses and the resource user's activities.

There are two general forms of liability: strict and negligence based. A producer under strict liability is responsible for bycatch regardless of the amount of care taken to avoid bycatch (in the more legal sense of damage). A producer under a negligence rule is not held responsible for bycatch unless the producer is negligent in conducting its operations. Full compliance with existing regulations can be considered *de facto* evidence of non-negligence and thereby absolve the producer of responsibility for any residual bycatch. The incentive, cost, and risk implications of imposing liability for bycatch depend upon the form of liability used (Lodge et al., 2019).

Strict liability, applying the PPP, holds producers liable for actual rather than expected costs of bycatch (Lodge et al., 2019). Under negligence liability, non-negligent producers are not liable for any residual bycatch. This implied property right, and the associated allocation of costs between society and producers, contrasts with strict liability and implies only partial implementation of the PPP. The scale and cost of production and price to target catch for non-negligent production do not incorporate the full social cost of production, including the bycatch, which is similar to the outcome under direct regulation.

INCENTIVE-BASED POLICY AND POLICY INSTRUMENTS

Incentive-based bycatch reduction policy and policy instruments create economic incentives to change the behavior and decision-making of producers to optimally reduce bycatch.

Bycatch is reduced through altering both the bycatch-target species catch ratio (substitution effect, mix) and scale of production that reduces both bycatch and target species catch (scale effect). Incentive-based approaches similarly alter the behavior and decision-making of producers in the supply chain and consumers in terms of the mix and scale of species to process, distribute, and consume. Over a longer period, such incentive-based approaches also generate "dynamic" incentives to create, diffuse, and adopt technology that lowers bycatch relative to target catch, i.e., to increase the selectivity of fishing.

Incentive-based approaches offer a number of other advantages (Gjertsen et al., 2010; Pascoe et al., 2010; Dutton and Squires, 2011; Segerson, 2011; Innes et al., 2015; Walmo et al., 2016; Lent and Squires, 2017; Miller and Deacon, 2017; Milner-Gulland et al., 2018; Squires and Garcia, 2018; Squires et al., 2018; Booth et al., 2019a; Lodge et al., 2019). They allow vessels greater flexibility to devise solutions that creatively and cost-effectively reduce bycatch. They also allow vessels to flexibly respond to changes in market conditions, the environment, technology, and resource conditions. They allow vessels to use decentralized, privately held information that is unavailable to the management authority. For example, vessels can use their knowledge about the time of day and location of the bycatch to adjust when and where they fish for the target species. Bycatch ITQs in the United States West Coast groundfish fishery led to fishers altering the deployment of trawl gear to either gain more precise information on the location of bycatch or to exploit the differential movements of bycatch species (Miller and Deacon, 2017). Trawl fishers shifted toward night fishing, a time when bycatch species migrate up from the sea floor and thereby become less vulnerable to trawl gear, while key target species remain near the sea floor. Trawl fishers also shifted toward shorter tows to obtain higher frequency information on bycatch, enabling avoidance through a shift in location when a bycatch stock concentration is encountered. In contrast, direct regulation tends to "bind up" or constrain vessels, and thereby restrict vessels' ability to respond to these changes. This inflexibility tends to raise production costs, by imposing uniform bycatch reduction regulations on a fleet with many differences among vessels, captain and crew skills, and in how they fish.

Individuals operating in groups of vessels sufficiently small to devise and self-manage their own bycatch reduction scheme may be more able to pool risk creating insurance programs which are proven to be effective for voluntary approaches for addressing rare and stochastic bycatch (Segerson, 2011; Deacon, 2012; Holland and Jannot, 2012; Holland and Martin, 2019; Kotchen and Segerson, 2019). These small groups have greater incentives for vessels to work collectively on activities such as real-time information sharing. Disadvantages include the potential for free riding on bycatch reduction of others or they can induce the "race to fish" as bycatch limits are met.

Pricing Bycatch

When seafood prices fail to contain information about the unpriced or underpriced costs of bycatch, producers (vessels, firms in the supply chain) and consumers do not have the full information about bycatch necessary to make decisions that lead to the optimal level and mix of bycatch. These costs are unaccounted for by producers and consumers, and more generally markets and their prices. Without seafood prices incorporating these bycatch costs, both target species and bycatch are overharvested with excessive bycatch relative to target catch.

Incentive-based policy reduces underpriced bycatch by creating a market price for the bycatch that accounts for the otherwise un- or underpriced residual bycatch jointly harvested with the target catch. Fully pricing the residual bycatch associated with the target catch increases the market price for the target catch. This higher target catch price conveys information to ex-vessel markets transmitted in full or in part through the supply chain to consumer markets.

For example, a coastal community in San Jose, northern Peru, has partnered with a local not-for-profit charity to address problematic sea turtle bycatch through a trial community management cooperative that includes pricing bycatch (Arlidge et al., 2020b). The initiative intends to create direct incentives for bycatch reduction by giving price premiums to fish caught by vessels that follow best-practice bycatch reduction guidelines.

Direct and Indirect Incentive-Based Approaches

Incentive-based approaches to bycatch reduction can be direct or indirect. Direct incentives tie penalties or rewards directly to, and conditional upon, verifiable conservation outcomes (called conditionality) that otherwise would not have occurred (called additionality). Examples of direct incentive approaches include payments for ecosystem services (Bladon et al., 2016), conservation easements (Deacon and Parker, 2009), taxes and subsidies, biodiversity offsets, credits, tournaments and prizes to incentivize bycatch-reducing technological change, and property rights such as bycatch ITQs.

Indirect incentives are incentives that are only indirectly linked to conservation in general and bycatch in particular (Gjertsen and Stevenson, 2011). Indirect incentive measures change the relative costs and benefits of specific activities in an indirect way (Convention on Biological Diversity [CBD], 2020). Conservation *per se* is not directly tied to an economic incentive. Instead, conservation occurs as a result of the incentive. Individuals are not directly rewarded for pursuing conservation activities or achieving a conservation performance/outcome, nor are they directly penalized for degrading activities or failure to achieve conservation performance. Indirect conservation uses development initiatives and changes in business models, product markets, employment, and income opportunities to encourage local resource users to change their behavior in ways that lead to greater conservation.

The two general categories of indirect incentive-based approaches to bycatch reduction are: alternative livelihoods (integrated development projects) and community-based conservation (community resource management and community conservation). Both may be especially suitable for small-scale fisheries (Allison and Ellis, 2001). Indirect approaches may not always specifically target bycatch but instead both target and bycatch species.

Alternative livelihoods, such as ecotourism, direct people away from the environmentally damaging activity and toward an alternative with lower impact livelihood activities providing at least equivalent monetary and non-monetary benefits (Wright et al., 2015). As an example, community members may receive wages as patrollers or rangers of sea turtle nesting beaches to protect sites, where the wages provide on-going incentives with conditionality and additionality (Gjertsen and Stevenson, 2011; Marcovaldi, 2011; Pakiding et al., 2020). In some instances, people may simply add the alternative livelihood to their existing activities (Torell et al., 2010), precluding bycatch reduction. Incentives are not necessarily created for the community as a whole. A related approach provides capital or infrastructure to the community as a whole, such as a school. An important issue is whether incentives are on-going or one-shot and whether there is conditionality and additionality. Wright et al. (2015) review additional issues.

Eco-tourism, another type of alternative livelihood, provides benefits to all or part of the community for preserving a population, such as sea turtles. For example, the communities of Kubulau district in southwestern Vanua Levu, Fiji, created a network of 13 protected areas to address poaching threats. Together with Moody's Namena Resort, the Kubulau communities enforce no-take areas against poaching to protect important dive sites, using a surveillance system involving community fish wardens. The system is financed through dive-tag fees from dive-tourism operations, and the funds are used for community-developed, tertiary scholarships, and operational costs such as patrolling. Such broad-based programs also provide protection to sea turtles and elasmobranchs that can be subject to bycatch in other areas (Niesten and Gjertsen, 2010). The Misool Eco-Resort in Papua, Indonesia entered into a 25-year lease with the customary owners of uninhabited Batbitim island to establish a no-take zone that protects coral reefs, sea turtles, elasmobranchs, and fish, protecting populations of potential bycatch over at least part of their life history (Niesten and Gjertsen, 2010).

Community-based conservation is based upon simultaneous achieving successful conservation and development (Berkes, 2004, 2006). Community-based conservation is decentralized and entails meaningful community participation in conservation (Agrawal and Gibson, 1994). Thus, community-based conservation 'includes natural resources or biodiversity protection by, for, and with the local community' (Western and Wright, 1994). Community-based management can take many forms and involve many existing institutions. It is also contextual and influenced by social norms, customs, and culture. For example, community-based management in the Pacific Islands tends to involve traditional institutions, especially taboos, to implement spatial management (Abernethy et al., 2014). Customary marine tenure, an institution, can have flexible boundaries that can impact, for example, spatial management (Foale and Manele, 2004). Community-based conservation is an important component to leatherback sea turtle nesting conservation in Papua Barat, Indonesia and supports offsets (Pakiding et al., 2020). Improved nest protection has helped optimize hatchling production, but local community engagement, through activities that empower and enhance quality of life, has been to the successful increases in hatchlings.

Community-based conservation sometimes is effective and in other instances is not (Ostrom, 1990; Baland and Platteau, 1997, 2000; Berkes, 2004, 2006). For example, a widespread community approach to mariculture and fishing prohibition failed to prevent poaching within the mariculture ranch, because sanctions were ineffective (Hair et al., 2020). A network of community-based MPAs was established in the early 2000s to conserve declining populations of bumphead parrotfish and other locally valuable fish (Hamilton et al., 2019). The populations did not decline due to sustained fishing pressure, poor enforcement of community-based management measures, and loss of fish nursery habitats due to logging.

Indirect incentive policies are potentially more sustainable than direct approaches since they do not require on-going financing, may be more consistent with social norms, may be more useful when the bycatch problem is not well defined or property rights are less clearly defined and enforced. However, indirect incentives may be 'one-shot' in nature due to front-loaded benefits without conditionality or additionality, may be in addition to rather than substitute for detrimental activity, and have unintended consequences. Communities are heterogeneous, and thus benefits and costs are not necessarily incurred in a fashion that would foster incentives for all relevant community members. This underscores the importance of front-loading community engagement in the design and implementation of alternative economic activities.

DIRECT AND INCENTIVE-BASED REGULATION

Regulation can play a critical role in enacting incentive-based approaches. Direct bycatch regulation, such as avoidance through closed areas, or minimization through bycatch limits or gear requirements, has a number of advantages. These include the known impact on producer behavior if the producer is compliant, low levels of risk for producers when the bycatch management requirements are well defined and established, and relatively low administrative costs if compliance can be easily monitored and enforced.

Direct bycatch regulation as the sole regulatory approach has a number of disadvantages that reflect information, cost, and incentive compatibility issues faced by fishery management authorities and producers. Direct regulation does not use all of the information that can potentially engage bycatch reduction options across and within steps of the MH. Direct regulation largely uses the information on bycatch mitigation held by the fishery management authority when in fact, producers hold information, sometimes quite subtle and producer-specific, that the fishery management authority does not typically know and use. This information grows in importance as producers gain experience, learn and adapt. Bycatch mitigation can entail multiple, ongoing adjustments in fishing that are taken individually and may have varying and even small impacts at the vessel-level but collectively can have a significant impact.

By not pricing residual bycatch, direct regulation does not add the cost of the remaining bycatch to the cost and price of the target catch (although the target catch production costs reflect the higher costs due to direct regulation). The target-catch price and costs will typically be lower than under bycatch pricing, so that vessels are not incentivized to sufficiently reduce the scale of production and bycatch-target catch ratio. However, sufficiently impactful direct regulation raises costs by enough to reduce the scale of production to the desired level (or even below), although not in a cost-effective manner, i.e., direct costs are unnecessarily incurred by producers to meet bycatch goals. Thus, direct regulation does not impose responsibility upon producers for any bycatch that might occur despite compliance with those regulations. It implies only partial implementation of the Polluter Pays Principle, since producers do not pay the cost of residual bycatch occurring despite compliance. Direct regulation shares bycatch reduction costs between producers (for avoidance, minimization, and restoration) and those other stakeholders who also suffer from the residual loss of bycatch. By failing to engage all bycatch reduction channels across and within steps of the mitigation hierarchy and across all producers and fishing areas, bycatch will not be reduced in a fully socioeconomically and ecologically optimal way.

Direct bycatch regulation does not generate any funds for compensation of either anticipated or unanticipated bycatch. Society instead bears the full cost and the full risk of any resulting significant bycatch despite compliance. Vessel operators who stay in the fleet are making sufficient profits to remain in business, however, and consumers who can pay the higher prices are still able to consume the product.

When costs of bycatch mitigation vary across producers, uniform or "one-size-fits-all" direct regulation is not cost-effective, because it does not create economic incentives to meet bycatch mitigation targets in a least-cost way. The cost-minimizing mitigation approaches can vary by producer, who can use producer-specific knowledge and methods. Direct bycatch regulation that is differentiated by some criteria, such as different limits according to vessel size class, could reduce the regulatory costs to vessel operators.

Direct bycatch regulation can incentivize non-compliance due to higher costs, with vessel operators engaging in actions to circumvent these regulations and other actions that ultimately hinder bycatch reduction and create economic waste. Finally, direct bycatch regulation fails to incentivize producers to exceed their regulatory requirements set through technology, process, or performance standards.

For all these reasons, reliance solely upon direct regulation to reduce bycatch is likely to fall short on several of the criteria that a fishery management authority might use in evaluating alternative policy approaches. Nonetheless, impactful direct regulation can sometimes induce more bycatch reduction than incentive-based approaches, particularly in time-critical situations.

There are also limits to incentive-based approaches due to the assumption of purely rational behavior. Thus, "Individuals may have bounded rationality, limited by cognitive resources, and employ a variety of heuristic procedures to achieve outcomes that are 'good enough' rather than truly optimal" (Conlisk, 1996). Further, a range of emotional, social, cultural and cognitive biases shape people's decisions (Cinner, 2018). Another limitation is the interaction of extrinsic motivation – economic incentives – with intrinsic motivation, leading to crowding out, as discussed below. Bycatch reduction is also shaped by social networks, trust and social capital, local leadership and role models, governance and institutional structures, social norms and peer pressure, perceived legitimacy of regulations, perceived effectiveness of proposed measures, and even the skill, experience and motivation of individual fishers and captains (Booth et al., 2019b).

Economic incentive-based approaches may not always have a substantial cost advantage over direct regulation if there is little heterogeneity in costs among vessels. If incentive-based instruments have only a small impact upon target catch prices, then the failure to optimally exploit target catch reduction channels (that therefore reduce bycatch) under direct regulation may have little impact in practice. Instead, direct regulation that is tailored to the heterogeneity, such as vessel size class or area and time fished or gear type, may be superior. The relative conservation and management costs of direct and incentive-based regulation can also tip the balance one way or the other.

Policy instruments based upon market incentives more typically, although not always, create stronger bycatch reduction incentives. In some instances, direct regulation can create even stronger incentives but at a higher economic cost.

INTERACTIONS BETWEEN PRIVATE SOLUTIONS AND INCENTIVE-BASED POLICIES

Economic incentives can interact with private solutions and prosocial behavior, notably intrinsic motivation such as social norms or altruism, in positive ways, called crowding in, and in negative ways, called crowding out (Deci, 1971, 1975; Bowles, 2008; Gneezy et al., 2011; Bowles and Polanía-Reyes, 2012; Rode et al., 2015; Young, 2015; Nyborg et al., 2016; Farrow et al., 2017; Booth et al., 2019a). Economic incentives can have two effects: the standard relative price effect that makes the incentivized behavior more attractive, and an indirect psychological effect associated with intrinsic motivation. The total effect on behavioral intentions is thus comprised of two effects either reinforcing each other through crowding in or offsetting each other through crowding out.

Resumption of dolphin hunting in the Solomon Islands after a conservation agreement between local communities and a conservation group previously providing financial support to develop alternative activities may be due to crowding out. Villagers explained that stopping the hunt had brought much tension in the village, and that resuming hunting brought peace back among community members (Innes et al., 2015; Oremus et al., 2015). Overfishing by rural communities in Columbia regulated by a weakly enforced quota with a fine as evidenced by a common property resource game with the local population (Rodriíguez-Sickert et al., 2008; Velez et al., 2010) and a comparable field experiment by rural Cambodian villagers (Travers et al., 2011).

A producer's intrinsic motivation can change in response to a change in external intervention, or a change in the perceived nature of the task or in the producer's self-perception. In some instances, a large psychological effect can crowd out the economically incentivized behavior.

Crowding out can operate through several principal channels. One channel is information. The fishery management authority (regulator) may be better informed about conservation goals than the producer (e.g., owner and/or operator of the vessel, principal). The fishery management authority, when better informed than the producer, may choose a reward level signaling the difficulty of the task and the producer's ability to complete the task satisfactorily, which could require additional economic rewards to complete the conservation. The fishery management authority may alternatively signal lack of trust in the producer's ability or willingness to reach a satisfactory conservation goal, which in turn can lower intrinsic motivation.

A second channel for crowding out occurs when extrinsic economic incentives reduce other intrinsic motives to conserve. One example is the higher personal benefit to an individual producer associated with a higher level of prosocial behavior, thereby impacting the reputational value attributed to a producer's intrinsic and extrinsic motivation. Decreasing the signal about a producer's prosocial preferences and increasing the signal about a producer's self-interest may result in lower image motivation and impaired self-esteem. (Image motivation pertains to the desire to be liked and well regarded by others and therefore depends on behavior visible to others). Offering higher material rewards may cause the indirect psychological effect to crowd out the standard price effect, depending on the extent to which the signals are public.

In sum, intrinsic motivation interacts with economic incentives in complex ways that change over time. In some instances, intrinsic motivation is more effective at reducing bycatch than economic incentives. Carefully tailored incentives can build off intrinsic motivation that enhances rather than inhibits bycatch reduction. Incentive-based policy can reinforce and shift intrinsic motivation by causing producers to first change their behavior, then shift their beliefs to conform to that behavior. The effects of incentives depend upon how they are designed, the form in which they are given (monetary and non-monetary), how they interact with intrinsic motivation, and what happens after the incentives are withdrawn. Although admittedly complex, the true cost of economic incentives should include the adverse effects of any motivation crowding out.

BROADER CONTEXT SHAPING CHOICE AND PERFORMANCE OF POLICY APPROACH

Transboundary Bycatch

Some bycatch populations are transboundary such that bycatch reduction requires multilateral cooperation or coordination across multiple parties and even multiple fishery management authorities (Barrett, 2003, 2010, 2016). This broader bycatch context is also holistic, extending beyond fishing interactions with the bycatch species to its life cycle and geographic range. Unilateral bycatch mitigation can be subject to a conservation "leakage," whereby the same bycatch population faces higher mortality in another jurisdiction due to bycatch in a different fleet (Mukherjee, 2015; Chan and Pan, 2016; Helvey et al., 2017). These leakages may be accompanied by a trade leakage, whereby the target catch is imported into the country unilaterally reducing bycatch in order to fill the consumption gap of the target species, possibly leading to increased bycatch mortality.

Implications of Industry Size and Organization

In commercial fisheries with reasonably strong and effective fishery management, bycatch reduction depends upon coordination and agency problems. Agency problems refer to attaining compliance when the producers (agents) have more information about bycatch reduction than the management authority (regulator and principal) (Vestergaard, 2010; Jensen et al., 2017). Coordination problems refer to coordinating the actions, behavior, and decision-making of producers among themselves and with the fishery management authority.

The effectiveness of bycatch reduction depends upon the number of producers, the scale of the individual businesses, and industry organization. Consider the different context of a fishery such as the large-scale tuna purse seine and longline vessels or Pollock or groundfish vessels in the United States North Pacific or North Atlantic, and most coastal fishing fleets. Bycatch reduction then also depends upon a comparatively strong and effective management authority with the capability for at-sea monitoring, control, and surveillance of catch or effort and effective enforcement and with the capability of organizing and coordinating producer behavior.

When there is a limited number of large-scale producers, bycatch reduction is simplified due to the strong coordination and lower transactions and information costs among and within large companies or other strong cooperative arrangements such as formal cooperatives or a mother ship and catcher vessels (Deacon, 2012; Kotchen and Segerson, 2019; Aceves-Bueno et al., 2020). This can lead to comparatively low transactions and information costs within and between organizational units and also with the fishery management authority. Lower costs in turn allow bycatch limits to be allocated to the large organizational units, which then self-resolve these issues. With the resulting regulatory interdependency among group members, when any member of a group contributes to improved group performance, it generates benefits for all other group members through penalty avoidance. The United States Alaskan large-scale groundfish trawl fleet, comprised a limited number of multi-vessel companies, voluntarily reduces halibut bycatch through a co-management scheme with the North Pacific Fishery Management Council (Fina, 2017). Group approaches can also promote information sharing, pooling of risks within a group, and reduce uncertainty

(Holland and Jannot, 2012; Holland and Martin, 2019). Such regulation has the potential to be voluntary, typically under a strategic threat of formal regulation (Kotchen, 2013; Segerson, 2010, 2013; Kotchen and Segerson, 2019).

In such a decentralized system, control of catch, monitoring, compliance, and enforcement is critical, such as through a basic incentive structure. Such an incentive structure could include a penalty of a lower bycatch limit for failure to meet the objective or a reward such as a larger target catch or longer season or carry-forward of bycatch limits from 1 year to the next (Segerson, 2011; Kotchen and Segerson, 2019). The comparatively limited number of organizational units lowers the costs of coordination and reduces the agency (asymmetric information) problem between the fishery management authority and the organizational units. The larger-sized firms or organizations and fewer vessel numbers means that these units can at least partially, if not fully, self-organize and coordinate bycatch reduction actions internally rather than relying upon markets and the fishery management authority to fulfill these functions (Coase, 1937). Such actions reduce the problem of differing quality and quantity of information – moral hazard – that otherwise occurs when producers only partially rather than fully fulfill the fishery management authority's intentions. Each party in a "contract" has an incentive and opportunity to gain from acting contrary to the principles laid out by the agreement. Here, producers may circumvent bycatch reduction regulations by taking advantage of loopholes or because the producers' actions are not observed or enforced.

When there are a large number of limited-scale producers, collaborative bycatch reduction becomes more complicated and expensive. There is also a greater potential for producers to fail to fully reduce bycatch according to intent of the regulations or the regulations do not completely cover all relevant bycatch possibilities (i.e., moral hazard) (Vestergaard, 2010; Jensen et al., 2017). The larger number of individual companies and vessels increases the information and transactions costs of coordinating the vessels' actions among themselves and in relation to the management authority. The vessels (which include multi-vessel companies), to the extent they can self-organize through industry associations and cooperatives, lower the costs and difficulties of coordination among themselves and with the management authority, i.e., facilitate co-management. Nonetheless, the management authority and the regulatory structure must provide more of the coordination and absorb more of the costs of coordination, monitoring, and enforcement.

With more numerous small-scale vessels, the fishery management authority must organize more formal, intricate, and costly monitoring, control, surveillance, and enforcement compared to the first type, due to the larger number and likely greater heterogeneity of the producers. The asymmetry of information between vessels and the management authority can be sizable given the large number of spatially dispersed vessels. Markets are used rather than transactions internal to firms in the case of larger and fewer firms due to the sizable information and transactions costs of coordination among the many vessels (Coase, 1937). Non-governmental organizations (NGOs), by absorbing and lowering the costs of information and transactions of bycatch reduction, can also contribute to coordinating the different vessels to meet bycatch reduction. However, NGOs can also raise costs by introducing additional and heterogeneous perspectives into the bycatch management process. In this second case, there is a well-developed tool kit of policy instruments that can be applied, many of them incentive-based, as discussed in the next section.

In sum, the choice of regulatory approach and policy instruments depends, in part, upon the number of vessels or firms, their scale of production, and how the industry is organized (including cooperatives and centralized industry groups). The fishery management authority can often set a bycatch limit and a penalty for non-compliance (a negative incentive) or even an in-kind reward for compliance, and allow the vessels to self-organize, as long as there is sufficient monitoring, control, and surveillance. Thus, private solutions, incentivized through a strategic threat, may suffice.

Information Between Producers and Consumers

Information about bycatch is not equally available throughout the supply chain. The quality and level of bycatch information decreases from the producer through the processors, distributors and on to the consumers. Adverse selection arises when any of these involved economic agents has more bycatch information than others but for its own benefit does not reveal it to the other parties when entering into a bycatch reduction agreement, such as unrevealed greater bycatch rates than other parties and participating in a certification scheme. Moral hazard arises when any involved agents do not fully comply with say bycatch reduction requirements when it does not bear the full costs of that risky behavior. This asymmetric information does not allow socially efficient behavior and decision-making for optimal conservation for parties farther from producers (Vestergaard, 2010; Kotchen, 2013; Segerson, 2013; Jensen et al., 2017). Thus, for example, consumers were initially unaware of the dolphin bycatch associated with harvest of large yellowfin tunas in the Eastern Pacific Ocean, and as a consequence, continued to purchase canned tuna unaware of the dolphin bycatch (Ballance et al., 2021). Eco-labels, standards, certification, and information programs all intend to rectify this issue by signaling information about bycatch, although the information quality (e.g., distortions) can readily deteriorate and may not even reach producers from consumer markets or higher in the supply chain.[2]

Bycatch Reducing Technological Change

Bycatch-reducing technological change lowers the ratio between bycatch and target catch to better achieve the avoidance and especially minimization steps of the MH. Examples include

[2] An anonymous referee noted that certification can provide a price incentive to producers, but also provides a signal to consumers, affecting the social license and thereby affecting the social norms in the fishery. In theory, with better information between producers and consumers, certification may not even be necessary to achieve this.

turtle excluder devices for shrimp trawlers (Crowder et al., 1994), sorting grids for groundfish trawlers or purse seiners (Broadhurst, 2000; Misund and Beltestad, 2000), dyeing pelagic longline bait blue, side-setting, and using Tori lines and weighted branch lines that sink faster to reduce seabird bycatch (Melvin et al., 2014; Gilman et al., 2016, 2020; Hall et al., 2017), the Hawaii Turtle Watch program that provides information to pelagic longline vessels on areas with sea turtle concentration (Howell et al., 2008), circle hooks rather than J hooks to reduce sea turtle bycatch encounter and post-hooking mortality with pelagic longliners (Andraka et al., 2013), non-entangling and biodegradable designs of [FADs used in tuna purse seine fisheries reduce the entanglement of sharks, sea turtles and other organisms (Moreno et al., 2016), and illuminating gill nets with chemical or battery-operated lightsticks to reduce bycatch of sea turtles, seabirds and marine mammals (Werner et al., 2006; Wang et al., 2013)].

Producers, acting in their private capacity, tend to provide bycatch-reducing technology at a level below what is optimal to society (Romer, 1986, 1990; Squires and Vestergaard, 2013b). This technology provision is subject to free riding – other parties benefit from the technology without contributing to the costs of its development and provision. This lowers the incentives to supply, through research and development, the socially optimal level of the new technology, can slow the rate of adoption and even limit the diffusion of the new technology. Governments therefore often fill the gap (Jaffee et al., 2005). Technology policy seeks to induce and finance research and development and account for the under-provision and free riding of bycatch-reducing technological change (Squires and Vestergaard, 2013b).

Nonetheless, direct involvement by industry in research and development and learning by doing does occur for bycatch reducing technological change (Hall et al., 2000; Gilman and Lundin, 2010). For example, bird-scaring Tori lines for longlining, and the backdown procedure after dolphins are captured, the Medina dolphin safety panel, deploying at least one rescuer during backdown, and carrying specified dolphin safety/rescue equipment for tuna purse seine vessels to reduce dolphin mortality in the Eastern Pacific Ocean (Hall and Roman, 2013).

Bycatch-reducing technological change (as a form of "directed technical change") can arise due to several factors (Acemoglu, 2002). The price effect on the catch side arises when the harvesting technology has lower bycatch and the target catch consequently commands a price premium over target catch with higher bycatch. An example is pole-and-line caught tuna compared to tuna caught using FADs, where the former receives a price premium for little or no bycatch. There was a concerted effort to ensure that price premium by informing consumers and retailers about the difference between pole and line vs. FAD-caught tunas. The price effect also arises on the input side when there are relatively scarce inputs, and correspondingly high input prices incentivize reduced use of this input to reduce production costs. An example is scarce, protected Chinook salmon that are bycatch to Alaskan pollock (Mize, 2014). The regulatory limits on Chinook bycatch constrain the amount of pollock that can be harvested. Protecting Chinook reduces the pollock catch, creating a high implicit price and cost (of foregone pollock catch) to the Chinook population, an input. The high implicit Chinook cost incentivizes innovation to reduce Chinook salmon bycatch.

Another factor that incentivizes research and development for bycatch-reducing technological change is the market size effect, which occurs when new technologies have a large market and more abundant inputs. An example is innovation to reduce dolphin mortality in the Eastern Pacific Ocean, since the target species, yellowfin tuna, entered into the large North American and European markets for canned tuna and there were abundant inputs of large yellowfin tuna (Gjertsen et al., 2010; Hall and Roman, 2013; Hall et al., 2017; Ballance et al., 2021). Continued access to these markets and their large volume required innovations to reduce dolphin bycatch. Alaskan Pollock also has an important market size effect given the volume and value of pollock production.

Social network effects (value of product or service increases according to number of others using it and social interactions), ideas (accumulated and new technology) and knowledge spillovers (new technology adopted by one fisher demonstrates benefits to others), and social learning (individuals are influenced by actions taken by others when information is dispersed) can be important to the adoption of new technology (Arrow, 1962; Katz and Shapiro, 1986; Romer, 1986, 1990; Jones and Romer, 2010; Squires and Vestergaard, 2013a,b, 2018; Mobius and Rosenblat, 2014; Sorenson, 2018; Alexander et al., 2020; Arlidge et al., 2020a,b). Technology can be embodied in the physical capital stock, such as a modified or new gear type (e.g., sorting grids for trawl nets), or new methods of fishing (learning by doing), such as fishing at different times of the day when there is less bycatch. Social interactions, social norms, and the number of users directly impact the type of new technology adopted and its rate of diffusion and adoption. Network scale economies and social learning create dynamic incentives that accelerate the rate of bycatch-reducing technological change (Arthur, 2009; Arlidge et al., 2020a,b).

Diffusion and adoption of new technology may be enhanced through subsidized gear or preferential access to markets (Squires and Vestergaard, 2013b; Eigaard et al., 2014). Governments or producers in one fishery can subsidize new bycatch reducing gear for producers in another fishery with bycatch on the same species and population, thereby creating an offset. A commercial fleet could finance the adoption of bycatch reducing gear by a small-scale fleet with bycatch of the same species, thereby reducing the avoidance step of the MH for the commercial fleet and allowing fishing that otherwise would not have occurred (Gjertsen et al., 2014).

EQUITY AND FAIRNESS

Different policies have different impacts upon equity and fairness in both process and distribution (consequences), which in turn can impact monitoring, compliance, and enforcement. Some policies can more inequitably and unfairly impact all or select producers and consumers than others.

The bycatch reduction policy process can be consistent with principles of equity and fairness. Bycatch reduction policy has distributional consequences, leaving some groups relatively better or worse off. Policy impacts can be assessed for consequences by equity metrics (Cowell, 2016).

The impact of differently designed policies upon equity and fairness in process and distribution can affect monitoring, compliance, and enforcement. Some impacts can be perceived as inequitable and unfair if the foregone catch and revenue from avoidance and even minimization disproportionately fall upon lower-income or otherwise disadvantaged producers and even some consumers. Moreover, benefits, whenever they are non-market and diffuse across a broad population or whenever there are higher prices from eco-labeling, may not be transmitted to these fishers. Low-income fishers then bear the costs but do not enjoy the benefits of bycatch mitigation, which can aggravate any inequitable impacts. The NNL objective may also disproportionately impact local and lower-income fishers and thereby contribute to an inequitable impact and lower social welfare (Griffiths et al., 2019a,b). Policies crafted using normative principles of equity and fairness and social distribution weights can ensure equity and fairness in process and distribution (consequences) (Young, 1994).

CONCLUSION

Bycatch reduction is not only a technical issue of harvesting technology and biology, but also a human issue involving behavior and decision-making by producers and consumers. Bycatch reduction also occurs within the context of different industrial and regulatory structures of fisheries, which in turn can impact the choice of basic regulatory approach – private solutions, direct regulation, incentive- (market-) based, and hybrid – and then choice of policy instruments.

There is no single "best" approach to bycatch reduction. The "best" approach almost invariably differs by the type of fishery – the species and its life history, geographical distribution (including transboundary stocks), and population status, gear, vessel numbers and ownership structure, domestic or international fishery, commercial or artisanal fishery, the fishery management authority and its governance, the importance of markets, geographical location, and legal structure of the State or Regional Fisheries Management Organization. Nonetheless, for commercial fisheries some very basic and broad conclusions can be drawn. In contrast, bycatch reduction in artisanal and small-scale fisheries remains a challenging issue, in part due to its conflation with economic development, and likely includes elements of community-based and alternative livelihoods.

The mitigation hierarchy provides an analytical framework by which to evaluate policies to mitigate bycatch. The least-cost mitigation hierarchy potentially gives greater bycatch reduction, especially when there are limited bycatch mitigation budgets. The equitable least-cost mitigation hierarchy can explicitly address distributional consequences of bycatch mitigation policies. No single policy approach or instrument is superior across all possible criteria, and the choice(s) depend upon each individual case. The "best" approach on paper may not be feasible, and a "second-best" approach that is practicable may then be preferred.

Bycatch reduction is comprised of multiple components requiring specific regulations or policies. When combining instruments, however, the fishery management authority should consider whether the different approaches being combined form substitutes or complements. Policy instruments that are substitutes can create redundancies without any bycatch reduction, which also raises costs and can even be counterproductive. Combining approaches that are complementary can lead to a better overall outcome than use of a single approach in isolation.

Uncertainty and timing are typically underappreciated but often critical to bycatch reduction. Uncertainty arises in determining who are the winners and losers, which can delay implementation of bycatch policy instruments until the results are more clearly sorted out (Libecap, 2014). Benefits are often more uncertain and enjoyed further in the future than more immediate and certain costs. Uncertainty can lead to waiting until more information is available before adopting incentive-based policy instruments or bycatch-reducing technology. Compounding uncertainty over the size and distribution of net benefits is the time of response for the bycatch population. A slowly rebounding population can delay compensation or rewards to fishers adopting bycatch reducing policy instruments that have serious costs in foregone target catch and revenue or have sizable costs in adaptation (such as fishing in another area). Fishers facing immediate costs, such as vessel loan payments, may not be able to readily adapt.

Designing and implementing workable solutions to bycatch clearly presents a challenge for fishery managers and stakeholders. Nevertheless, this work is critically important as bycatch is a loss to society and in some cases, can cause extinction. A multidisciplinary approach conducted in collaboration with the fishing community can provide the widest possible array of options for mitigating bycatch whilst maintaining a viable fishery.

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

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Credit Systems for Bycatch and Biodiversity Conservation

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Credit systems for mitigation of bycatch and habitat impact, incentive-based approaches, incentivize changes in fishery operator behavior and decision-making and allow flexibility in a least-cost method. Three types of credit systems, originally developed to address environmental pollution, are presented and evaluated as currently underutilized incentive-based approaches. The first, a cap-and-trade approach, evolved out of direct regulation through restricted limits with flexibility through the creation of tradeable unused portion of the limit, called credits. The second, a penalty-reward system, incentivizes bycatch- and habit-impact- reducing vessel behavior through rewards for positive behavior, and penalties for negative behavior. The third is a hybrid of the first two. All three systems can be used in the context of both absolute (fixed) and relative (rate-based or proportional) credits. Transferable habitat impact credit systems are developed for area management. The cap-and-trade credit system is directly compared to a comparable property rights system in terms of characteristics, strengths, weakness, and applicability. The Scottish Conservation Scheme and halibut bycatch reduction in the Alaskan multispecies groundfish fishery provide real-world examples of success with credit systems. The strengths, weaknesses, and applicability of credit systems are summarized, along with a set of recommendations. Cap-and-trade credit systems provide an important alternative to property rights, such as when rights are not feasible, and for this reason should prove useful for international fisheries. Penalty-reward and hybrid credit systems can substitute for cap-and-trade credit systems or property rights or complement them by addressing a related but otherwise unaddressed issue.

Keywords: credits, bycatch, habitat impact, conservation, economic incentives, credit systems, property rights

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INTRODUCTION

Credit systems for bycatch and habitat impact conservation provide incentive-based approaches to reduce bycatch. Credit systems incentivize changes in vessel operator behavior and decision-making that allow them to flexibly reduce bycatch in a their own, least-cost way. Credit systems can be voluntary, as in the Alaskan pollock fishery (Bersch, 2013; Mize, 2014; Fina, 2017), or mandatory, as they were in the now superceded Scottish credit scheme (Curtis, 2017).

Three basic types of credit systems are defined in this paper, based on systems that were all originally developed to manage environmental pollution (Fischer, 2001, 2003; Montero, 2002; Boom and Dijkstra, 2009; Sovacool, 2011; Nentjes and Woerdman, 2012; Goulder et al., 2019). The first is direct regulation made flexible through a cap on Total Allowable

Catch/Bycatch/Effort/Habitat Impact, shares of which are allocated to[1] vessels to create vessel-level limits or quotas.[2] The vessel's unused portion of the limit – the credit – can compensate the bycatch of a vessel beyond its limit, with the bycatch credit of another source, here the unused bycatch limit of another vessel. This trade in credits can be between vessels in a multi-vessel firm (internal source) or between independently owned vessels (external source). Credit exchange, whether in-kind or for monetary payment, creates a price in the "credit market," which can be implicit if formed through exchange between internal sources or explicit if formed in the secondary "credit market." Credits might also be banked for use in the next management year, depending upon the features of the program. This transferable credit (TC) system is a basically a cap-and-trade system.

Transferable credit management can be combined with price ceilings and floors in the credit market to form a two-part policy instrument, also called a hybrid policy instrument (Roberts and Spence, 1976; Pizer, 2002; Hepburn, 2006).[3] The first part of the policy instrument is the TC and second part of the policy instrument is the credit price ceiling and floor.[4] TC bycatch management was first proposed by Sugihara et al. (2009), and further discussed by Pascoe et al. (2010); Bersch (2013); Mize (2014); Van Riel et al. (2015); Lent and Squires (2017); Squires and Garcia (2018); Squires et al. (2018).

The second type of credit system is a two-part policy instrument that is a penalty-and-reward credit (PWC) system, also called an indirect tax-subsidy (Roberts and Spence, 1976; Fullerton and Wolverton, 1999, 2000, 2003; Segerson, 2011; Kotchen and Segerson, 2019). The first part is the penalty (indirect tax) and the second part is the reward (indirect subsidy). Deposit-refund programs are a familiar example, in which the deposit is the penalty and the reward is the refund (Bohm, 1981; Fullerton and Kinnaman, 1995; Fullerton and Wolverton, 1999, 2000, 2003). PWC has been discussed for bycatch by Segerson (2011); Van Riel et al. (2015), – who call it "behavioral credits, Kotchen and Segerson (2019) – who call it "behavioral credits," Lent and Squires (2017); Squires and Garcia (2018).

The third type of credit system is a three-part policy instrument, also a hybrid policy instrument. TC that are part of a cap-and-trade system are supplemented with penalty-reward credits (Weitzman, 1974; Roberts and Spence, 1976; Pizer, 2002). Adding penalty-reward credits to TCs creates additional flexibility and the ability to tackle additional issues that cannot be fully addressed by TC. For example, TC can address the overall level of bycatch but only with great difficulty and imprecision can TC by itself address juveniles or areas.

A limited amount of work has been done in fisheries on credit systems for bycatch (Pascoe et al., 2010; Segerson, 2011; Van Riel et al., 2015; Lent and Squires, 2017). Nonetheless, because credit systems have largely been developed in the literature on environmental pollution and to a lesser extent the economics of regulating industries (as noted in the references provided in the discussion), further insights can be drawn from environmental economics of pollution and industry. The PWC Scottish Conservation Scheme and the TC halibut bycatch reduction in the Alaska multispecies groundfish trawl fishery provide real-world examples of the effectiveness of credit systems in mitigating bycatch.

Section "Bycatch Credits" further develops the three types of credit systems for bycatch and habitat impacts. Section "Absolute and Relative Bycatch Credits" introduces absolute and relative credit systems, which are variations that can be applied to each of the three credit systems. Section "Habitat Impact Credits" introduces credits for transferable habitat impacts, which are separately discussed due to their distinct features as a separate type of bycatch. Section "Credits vs. Property Rights" discusses the difference between rights-based and credit management for TC. Section "Fishery Examples" presents two case studies. Section "Concluding Remarks" concludes this study.

BYCATCH CREDITS

This section develops TC, PWC, and the combined three-part approach, drawing from the environmental economics literature. Both TC and PWC can be implemented at the individual vessel, multi-vessel firm, broad industry level, where firms (single and multi-vessel) self-organize, or the individual State in an international fishery. Incentives are stronger the more directly the credit system is applied to bycatch. For example, a stronger incentive is created when applied to bycatch rather than target catch or effort. Incentives are created by establishing standards that limit the behavior of a producer, with a performance standard on the outcome of production, or a process standard on the process of production (see Helfand, 2013 for environmental pollution).

The three types of credit systems, since they derive from process or performance standards which are made flexible, can be uniform or differentiated by type of source, season, or even time of day (see Helfand, 2013 for environmental pollution). Thus, credit systems can be uniformly applied to all vessels or differentiated according to some criteria, such as vessel size class, bycatch species, gear type, area, season, habitat, Flag State, etc. If a single uniform standard is applied to all vessels, cost-effectiveness of each producer and by extension society as a whole will be undermined and therefore result in a less optimal outcome, given

[1]Bycatch can be broadly defined as unwanted species or individuals caught during fishing operations (Hall, 1996; Squires et al., 2021). Bycatch, from an economics perspective, can be classified as either a: (1) commercially exploited species with contributions to biodiversity and the ecosystem and its services that are not incorporated into market prices (i.e., incomplete market prices), or (2) protected species, also with contributions to biodiversity and the ecosystem and its services, but not commercially exploited or with a market price (i.e., without a market price – unpriced). A related problem is habitat impact, especially with groundfish bottom trawls, and more generally biodiversity. Hereafter, unless otherwise specifically noted, bycatch includes habitat impact whenever relevant, and the general points apply to the broad issue of biodiversity conservation through credit systems (e.g., habitat or water credits).

[2]In international fisheries, the Total Allowable Catch/Effort/Habitat Impact shares are invariably first allocated to States and then to the vessel (Grafton et al., 2010).

[3]Hybrid policy instruments differ from multiple policy instruments (Hepburn, 2006) by combining multiple policy instruments into a single instrument.

[4]A policy instrument is an individual economic tool which can be used to vary an economic parameter in order to achieve an economic objective. Hybrid instruments should be distinguished from the use of multiple instruments for the problem.

that vessels face different costs in meeting the uniform bycatch standard. As a consequence, bycatch reduction is more costly for each vessel, managers, and society as a whole.

Differentiated standards, which are more closely tailored to different classes of producers (e.g., vessel size classes, gear types), better fit the capability of heterogeneous producers (e.g., vessels of different sizes, gears, Flag States) to reduce bycatch at the lowest possible cost for that class of vessel. Differentiated standards thus tend to be more cost-effective than uniform standards, although transactions, information, and administrative costs (including enforcement) for the regulator may increase with the level of complexity. Differentiated standards are also less regressive in their distributional impacts, since standards are more directly tailored to individual producers. Direct regulation through differentiated standards can in principle achieve the same cost-effective result of a TC but this would require that different standards be set for each pollution source, and, consequently, that policy makers obtain detailed information about the compliance costs each firm faces. Such information is rarely available to government. By contrast, market-based instruments provide for a cost-effective allocation of the pollution control burden among sources without requiring the government to have this information (see Stavins, 2001 for environmental pollution).

The TC has a Total Allowable Catch, Total Allowable Effort, or Total Allocable Habitat Impact which can be a hard cap or 'soft' cap. The goal could also be a vessel-level benchmark, based on a technological, scientific, or industry-specific bycatch-target catch ratio rather than historical ratio (see Weishaar, 2007; Gerigk et al., 2015; and Goulder et al., 2019 for environmental pollution) or "yardstick" management, in which the bycatch of comparable vessels is used to infer a vessel's attainable bycatch level (see Shleifer, 1985 for environmental pollution and industry regulation). These approaches have largely been developed for and applied to environmental pollution, and are the basis of consideration as an incentives-based approach to bycatch mitigation. Under this approach, the vessel operator must account for every unit of bycatch in excess of the benchmark or "yardstick" and pay a penalty (monetary or in-kind) if the operator cannot attain credits via the trading system or benefit from its own past or future credit savings.

Transferable Credits

Under TC, a regulation obliges vessels to not exceed a limit, to have these limits accredited in some manner (including by third parties), and to report them to the regulator. While the regulator creates the rules and takes an active role in monitoring compliance, the regulator does not directly participate in the credit system. Vessels can buy new credits from, and sell their own credits, to other market participants. In a voluntary credit system, the regulator can be expected to have far less involvement, and may not even set an overall limit in a relative TC (although it would set the relative ratio). TC, in contrast to property rights, are not created by the regulator and distributed to vessels, although the flexibility to transfer the credits is facilitated by the regulatory framework and therefore a secondary market emerges for credits.

The TC price sets the basis for incentive-based management. The credit price raises the cost of production and the prices of target catch and bycatch that are landed and sold in commercial markets. Bycatch now has a market and a price, thereby creating a cost for bycatch that was formerly excluded from the cost of production. This includes bycatch of non-market bycatch, such as protected species, which now is also a cost that is absorbed by the target catch. The higher cost of production for both types of bycatch reduces the overall amount of bycatch and target catch produced (scale effect) and reduces the bycatch-target catch ratio (substitution effect). In the language of economics, the credit price internalizes the external cost of bycatch and thereby leads to a more socially optimal scale (volume) and scope (bycatch-target catch ratio) of production.

Transferable credit and PWC can incentivize real-time spatial management (Hobday and Hartmann, 2006; Little et al., 2015) and modified gear deployment to reduce the bycatch-target catch ratio. Real-time spatial management can be organized through a formal third party, such as Sea State in Alaska (Mize, 2014; Little et al., 2015), or internally within a firm through cooperation and communication (Fina, 2017). Credits can also create "dynamic" incentives to induce the creation and adoption of technological change that reduces the cost of production and the bycatch/target catch ratio – increasing "selectivity," i.e., bycatch reducing technological change (see Lent and Squires, 2017; Squires and Garcia, 2018; Milner-Gulland et al., 2018; Squires et al., 2018; for fisheries, and Montero, 2002 for environmental pollution).

Economic theory predicts that under TC, bycatch control measures will be concentrated in vessels that can do so at lowest cost (see Goulder and Parry, 2008 and Nentjes and Woerdman, 2012 for environmental pollution). Such vessels will earn credits and sell them at a profit. Vessels with high bycatch control costs then reduce costs by buying credits instead of controlling cost past some point whilst the costs of bycatch are fully reflected in their production costs and further down the marketing chain. Transferability creates gains from trade and the economic efficiency through the ensuing cost-effectiveness.

Credit markets can range from bilateral transactions between vessels, where vessels can be separately owned or owned by the same multi-vessel company, to a formal market between independent vessels. Credit market volume can also vary, depending upon fishery size and characteristics and the vessel's derived demand for credits and the vessel's supply of credits. A larger volume of credits is expected to lead to more stable prices, since individual transactions are smoothed out and each one is less influential. Greater market activity (i.e., credit formation and exchange) is expected during the end of the production period when bycatch limits begin to constrain production. Moreover, credit exchanges entail transactions and information costs, which can inhibit credit creation, exchange, and price signals that accurately reflect the value of a credit.

Penalty-Reward Credits

This is a two-part policy instrument, the first part of which is a penalty (indirect tax) on one market or non-market transaction such as catch, bycatch, or input (gear, equipment, days). The penalty can be in monetary units or in kind, such as units of catch, effort, or habitat impact. The penalty has a similar impact to a requirement that a vessel operator purchase additional quota from the regulatory body to cover any excess from the original quota. The penalty can be fixed in amount if the bycatch limit is exceeded or proportional to the amount by which the bycatch limit is exceeded (Segerson, 2011; Kotchen and Segerson, 2019).

The second part of this two-part policy instrument is a reward (indirect subsidy), either monetary or in-kind and fixed or proportional, on a different market or non-market transaction that is an alternative to bycatch with less adverse impact or an input or activity that reduces bycatch (Fullerton and Wolverton, 1997, Fullerton and Wolverton, 2000, 2003; and Kotchen and Segerson, 2019 for environmental pollution; Segerson, 2011 and Kotchen and Segerson, 2019 for bycatch). The tax and subsidy do not have to apply at the same rate, to the same species or input, or even to the same economic agent (vessel, firm). This type of credit system is a relative standard (process or performance standard), discussed in Section "Absolute and Relative Bycatch Credits" below, in which the indirect subsidy applies to vessels whose performance meets the standard or limit, whilst the indirect tax applies to vessels with performance or process exceeds the standard or limit. A deposit-refund system for Fish Aggregating Devices (FADs), penalty-reward bycatch credits, bycatch cap with fixed or proportional penalty/reward, or transferable credit markets with price ceiling and floor provide fisheries examples. A deemed value system for rights-based management is another indirect penalty-reward (tax-subsidy) (Squires et al., 1995).

This penalty-reward credits system can avoid the challenge of monitoring, enforcing, or measuring a direct tax (whether Pigouvian or "green") on bycatch (see Fullerton and Wolverton, 1997; Fullerton and Wolverton, 2000, 2003; and Kotchen and Segerson, 2019 for environmental pollution; see Segerson, 2011 and Kotchen and Segerson, 2019 for bycatch). This approach applies the penalty (tax) to observable market transactions, such as the purchase of target catch by a processor or even a consumer, and simultaneously to reward (indirectly subsidize) other market transactions, such as the purchase of "clean" inputs (e.g., subsidy to purchase bycatch friendly gear) or observable expenditures on non-market transactions such as operating expenditures for fishing in areas with lower bycatch. Even when it is possible to monitor and measure bycatch, enforcement may not be feasible. For example, a tax on bycatch may create a powerful incentive to reduce bycatch but it may also induce illegal discarding. The penalty reduces production and consumption of both bycatch and target catch (scale effect) and reduces the bycatch-target catch ratio (substitution effect).

The penalty is equivalent to a tax at the same rate on all inputs to production, such as vessel, gear, equipment, crew, fuel, bait (see Fullerton and Wolverton, 1997; and Fullerton and Wolverton, 2000, 2003 for environmental pollution; see Segerson, 2011 and Kotchen and Segerson, 2019 for bycatch). The penalty renders all bycatch-generating inputs relatively more expensive, and thereby reduces the bycatch-target catch ratio. The reward subsidizes all non-bycatching generating inputs, such as desired gear or resource stock in a fishing area. The first part is a tax that is imposed upon the presumption that all production uses a "dirty" technology. The second part is an environmental subsidy that is provided only to the extent that production uses "clean" technology.

The penalty does not have to equal the reward in order to effectively address bycatch (Fullerton and Wolverton, 1997, Fullerton and Wolverton, 2000, 2003 and Kotchen and Segerson, 2019 for environmental pollution; Segerson, 2011 and Kotchen and Segerson, 2019 for bycatch). A vessel operator could sometimes receive a reward and sometimes incur a penalty. On average and over time, if the penalties and rewards are correctly set and accounting for how the regulated fishery operators adjust to the policy, the quota should just be met and penalties should just match rewards (Fullerton and Wolverton, 1997; Segerson, 2011; Kotchen and Segerson, 2019). This matching of penalties and rewards implies that the policy would neither generate revenue nor require the regulator to raise funds to finance the rewards if denominated in monetary units. In the language of economics, this is revenue neutrality, and there are no net costs to vessels on average. Similarly, this matching implies that the regulator would not expend net in-kind credits or penalties on average over time.

Under certain bycatch situations, the magnitude of a fixed penalty can be set at any level high enough to ensure that the vessel has higher profits by complying with the target than not complying (Segerson, 2011). Under uncertain bycatch, such as rare-event bycatch, the fishery operator cannot avoid the penalty with certainty, and must instead weigh the marginal cost of bycatch reduction against the marginal expected benefit, which reflects not only the magnitude of the avoid penalty but the effect that additional bycatch reduction has on the likelihood of exceeding the target and hence imposition of the penalty. Thus, under uncertainty, the penalty must be set more carefully to ensure that this balancing leads to efficient bycatch reduction rather than too much or too little bycatch reduction. The combined penalty-reward approach treats randomly occurring outcomes symmetrically, imposing a penalty for exceeding the limit and a reward for being below it. Hence, despite the uncertainty about whether the allowable limit will be met given the firms' decisions, the limit itself does not affect producer incentives (Kotchen and Segerson, 2019). Such symmetry does not apply to the only a penalty or reward (based on a given threshold) but not both. Thus, combining the penalty with the reward ensures that even with uncertainty, private incentives align with social incentives, regardless of where the threshold is set. Segerson (2011) discusses the case of a proportional penalty under certain and uncertain bycatch.

Policy makers face many challenges in determining optimal penalties and rewards and in deciding whether to use in-kind or monetary units. A key factor is asymmetric information; policy makers hold less information about bycatch in contrast with the vessel operators. Uncertainty about the ratio between bycatch and target catch and economically optimal scale of production and how vessels respond to penalties and rewards further compound the difficulty in setting these penalties and rewards. In other cases, it may be difficult to determine the biologically optimal level of bycatch due to uncertainty around population estimates and 'rare event' bycatch species. The closer the penalties and rewards are calibrated to bycatch caught, the stronger and more accurate the incentive, however the more information is required. Whether the bycatch species is the limiting species to the catch of other bycatch and target species also impacts the difficulty in setting the penalty and reward.

Deposit-refund systems are most likely to be appropriate when: (1) the objective is one of reducing illegal or uncontrolled disposal, such as drifting FADs, as opposed to such objectives as general reductions in the level of bycatch or number of FADs and (2) there is a significant asymmetry between *ex ante* and *ex post* (post-deployment) clean-up costs (see Bohm, 1981; Bohm and Russell, 1985; Stavins, 2001 for environmental economics). For these reasons, deposit-refund systems may be among the best policy options to address disposal problems associated with gear and "ghost fishing."

An additional two-part policy instrument combines TC management with price ceilings and floors in the credit market (see Weitzman, 1974; Roberts and Spence, 1976; McKibbin and Wilcoxen, 2002; and Pizer, 2002 for environmental economics). Together, the credit price ceiling and price floor constrain the credit market to price positions within minimum and maximum bounds. This creates credit price stability. If the credit price stays at the floor, then the credit market becomes equivalent to a tax. Another two-part policy instrument entails an initial distribution of limits, accompanied by credit trading, combined with additional limits available from the management authority at a specified "trigger" price (see Roberts and Spence, 1976; and Weitzman, 1978; Pizer, 2002 for environmental economics). This approach combines quantity and price controls.

Combined: Three-Part Policy Instrument

Transferable credits that are part of a cap-and-trade credit system supplemented with penalty-reward credits form a three-part policy instrument (see Roberts and Spence, 1976; Weitzman, 1978; and McKibbin and Wilcoxen, 2002 for environmental economics). TC systems alone incentivize bycatch reduction, and the bycatch-reduction incentives and uncertainty are reduced by adding explicit penalties, such as loss of bycatch or effort quota, or rewards, including additional bycatch or effort quota drawn from an explicit pool set aside for this purpose, or a reward for catch below quota/baseline.

This combined approach addresses multiple issues (externalities) in a way that is not possible with a single approach. For example, TC can address the overall level of bycatch and when combined with a bycatch cap increases certainty. Adding penalty-reward credits enhances the ability to address additional issues such as juvenile target species bycatch or areas that are "bycatch hotspots." The combined approach can also incentivize desired adoption of bycatch reducing gear, gear deployment, and equipment. In principle, this three-part policy instrument combines the advantages of price-based (e.g., taxes) and quantity-based (e.g., quotas or limits) policy instruments and compensates for their deficiencies. A reward for adopting a bycatch reducing technology addresses the additional issue (externality) of sub-optimal technology. Three-part policy instruments do however incur greater monitoring and administrative costs.

A three-part policy instrument can also support formal or informal credit price floors and ceilings and thereby lower vessel risk and uncertainty. The penalty protects against unexpectedly high credit prices if true (marginal) bycatch reduction costs are higher than anticipated and therefore a pure relative credit trading system would result in bycatch reduction at a non-optimal level. Rewards may stimulate further bycatch or habitat impact reduction if the marginal costs of bycatch reduction are lower than expected and a pure relative credit system would lead to too little conservation. The regulator pays vessels for every unit of bycatch that falls below their allowance (payments are typically in-kind, such as additional days). The reward sets a floor for the credit price, since any vessel with bycatch would rather collect this payment than sell credits on the market at a lower price. Within this price range, the credit program provides for satisfying the bycatch reduction target determined *ex ante*.

Comparison of the Three Credit Systems

The **Table 1** compares the salient features, strengths, and weaknesses of the three credit systems.

ABSOLUTE AND RELATIVE BYCATCH CREDITS

Credits of any of the three types can be either absolute (fixed) or relative (rate-based, ratio, proportional) (see Helfand, 2013 for environmental pollution standards). Credit systems are implicitly or explicitly defined as a rate, such as per unit of time, per unit of input such as a FAD, per unit of area, or per ton of target catch. If the measure in the denominator of either the limit in a TC system (e.g., bycatch limit per time period) or the penalty-reward credit are completely exogenous to the process, any credit system can be considered an absolute credit system. An example is a bycatch limit of a specified number of animals per year. When a vessel has some control over the denominator, such as target catch in a bycatch-target catch ratio or input such as bycatch per day or per gear such as a FAD or number of bycatch animals per ton of target catch, the credit system is a relative credit system. Each compliance period, the regulator multiplies the total allowable bycatch by each vessel's bycatch ratio to obtain the allocation to each vessel (such quotas or limits in the denominator in relative systems are sometimes called intensity targets or rate-based standards in the environmental pollution literature: Weishaar, 2007; Nentjes and Woerdman, 2012; Helfand, 2013; Goulder et al., 2019).

Absolute credits entail an exogenous total and per vessel bycatch limit within each compliance period, but the total bycatch and the vessel allocations are endogenous with relative credits (see Goulder and Parry, 2008; Goulder et al., 2019; and Kotchen and Segerson, 2019 for environmental pollution). Unlike absolute credits, the regulator does not know the total bycatch and each vessel's bycatch until the end of the compliance period, after which vessel operators' production decisions over the period have been made.

The input for bycatch credits can be either directly related to bycatch reduction, such as gear, or a more general input such as days fished, with alternative impacts upon incentives and bycatch reduction. The input can be a stock variable, such as vessel size, or a flow input directly related to production, such

TABLE 1 | Features, strengths, weaknesses of three credit systems.

Characteristics	Strengths	Weaknesses
Transferable credits with cap-and-trade		
Direct regulation made flexible	Creates incentives to lower bycatch	Residual bycatch not priced or costed
Allocate limits to vessels	Substitute for property rights when rights are not suitable	Creates implicit output subsidy (no price or cost for residual bycatch)
Credit is unused limit	Fewer allocation issues than property rights	Vessels do not incorporate residual bycatch cost into decision-making
Transferable	More acceptable in many international fisheries than property rights	Weaker incentive to reduce bycatch and lower efficiency than property right
Bycatch credits are priced to create cost and economic incentive	Flexibility to respond to changes in markets, environment, resource conditions	Weaker incentives for bycatch-reducing technical change than rights-based management due to lower costs and implicit output subsidy
Aggregate supply of credits not fixed but limit is fixed	Economic efficiency, cost-effectiveness	Requires careful monitoring
Can be relative or absolute bycatch credits	Management authority controls aggregate bycatch limit	High-grading and discards
	Minimal information requirements about vessels	Setting overall bycatch level
	Dynamic incentives for bycatch-reducing technological change	
Penalty-reward (two-part policy instrument)		
First part: penalty (indirect tax)	Creates incentives to lower bycatch	Rewards may not equal penalties in short run (not revenue-neutral)
Second part: reward (indirect subsidy)	Avoids monitoring and enforcement problems of direct tax by applying tax to observable market transactions	Management authority does not directly control aggregate bycatch limit and mortality
Penalty and reward in money or in kind (e.g., days)	Flexibility for regulator	High-grading and discards
Penalty lowers bycatch by conferring a cost to bycatch	Economic efficiency, cost-effectiveness	Residual bycatch not priced or costed creating (implicit output subsidy)
Reward lowers bycatch by conferring a benefit to bycatch reduction	Welfare increasing (Pigovian) indirect tax and indirect subsidy	Can reduce need to monitor and enforcement and thereby the associated costs
Long-run revenue neutral	Dynamic incentives for bycatch-reducing technological change	Relating optimal penalty and rewards to (optimal) bycatch fishing mortality complex
Deposit-refund	Deposit-refund systems reduce lost gear, subsequent "ghost fishing," and overcapacity due to increasing gear productivity	Setting optimal penalties and rewards for vessels requires vessel-specific information, creating uncertainty. Monitoring and enforcement needs (e.g., gear marking)
Transferable credits with cap-and-trade combined with price ceilings and floors	Deposit-refund systems suitable when significant asymmetry between *ex ante* (legal) and *ex post* (illegal or post-deployment) retrieval or clean-up costs	Setting overall bycatch level
Can be relative or absolute	Complements cap-and-trade property right or credit systems aimed at overall catch and overcapacity	Deposit-refund systems less suited for overcapacity and overfishing
Combined transferable credit and penalty-and-reward (three-part policy instrument)		
Price controls combined with quantity controls	Creates incentives to lower bycatch	Additional complexity and costs
	Allows addressing additional bycatch issues (externalities), e.g., age, area, timing	Potential transactions and information costs, including asymmetric information between vessels and management authority
	Combines advantages of both price- and quantity-based policy instruments	Residual bycatch not priced or costed (implicit output subsidy)
	Can lower vessel risk and uncertainty by price ceiling and floor (form of insurance)	
	Dynamic incentives for bycatch-reducing technological change	
	Economic efficiency, cost-effectiveness	

as fishing time, with stronger incentives for bycatch reduction with flow inputs.[5] Relative credits can also be defined in terms of a performance benchmark, such as a target bycatch reduction per vessel (cf. Weishaar, 2007; Gerigk et al., 2015; Goulder et al., 2019; and Kotchen and Segerson, 2019 in environmental economics) or a "yardstick" (cf. Shleifer, 1985 in the economics of regulation literature).

Reducing the relative credit's required ratio of bycatch-target catch or bycatch-input reduces bycatch. Similarly, tightening the relative credit's regulation ratio of bycatch per unit of input reduces catch. Over time, the regulatory body can adjust this ratio according to conditions in the environment, stock abundance, markets, experience, etc. The relative bycatch credit system, by which compliance requires avoiding exceeding a given ratio of bycatch to output or input, contrasts with the absolute credit system, by which compliance requires avoiding a given level of bycatch.

Absolute credits are likely to induce different bycatch conservation than relative credits because of the potential to change the denominator in relative credits.[6] For instance, under absolute bycatch credits (such as a limit on the number of sea turtles caught), the vessel must reduce bycatch, the numerator. Under relative bycatch credits (such as the number of sea turtles per day/hook/or metric ton of swordfish caught), for example, the vessel can adjust the numerator by reducing bycatch or it can increase the denominator by increasing days, or more use of a gear, or increasing target catch (whichever is the denominator). An absolute bycatch credit program is unambiguous: bycatch must decline, but a relative bycatch credit program can potentially lead to ambiguous results – for example, as target catch increases, bycatch increases. The dependence of relative bycatch credits on within-period denominator decisions has important implications for incentives and associated system performance, affecting harvest levels, overall bycatch reduction, and the levels and distribution of costs (cf. Goulder et al., 2019 for environmental pollution).

Even if the denominator cannot be manipulated, the two approaches can differ in their effects if the denominator can fluctuate over time (Helfand, 2013 for environmental pollution). For example, bycatch per unit of target catch or some input (e.g., number of sea turtles per mt of swordfish or per hook) could fluctuate due to vessel breakdowns, spikes in fuel costs or plunges in target catch prices, bycatch species population levels, or even weather. In contrast, an absolute cap on bycatch (e.g., number of sea turtles) would not respond to such fluctuations.

The incentives generated by a relative credit defined as bycatch per unit of input (e.g., days or number of hooks) differ compared to bycatch per unit of target catch. Both reduce the observed bycatch-target catch ratio, but the prescribed bycatch-input catch ratio generates weaker incentives to reduce bycatch because the impact is less direct.

[5] A stock is measured at one specific time and represents a quantity existing at that point in time, which may have accumulated. A flow is measured over an interval of time.

[6] This paragraph extends Helfand (2013) discussion of (non-tradable, direct regulation) standards to credit systems.

Credit systems, either absolute or relative, with differentiated standards can help meet distribution objectives, since less stringent standards can be assigned to vessels that otherwise would face especially high compliance costs (Goulder et al., 2019). Multiple standards with absolute credits affect the distribution of policy costs but do not reduce cost-effectiveness.[7] Multiple standards with relative credits increase the economic costs, thereby lowering cost-effectiveness, because they alter the relative magnitudes of the "implicit output subsidy" from unpriced residual bycatch across vessels and thereby distort the relative target catches of these vessels. Target catch levels may also be higher under relative credit systems, since the denominator (target catch or input level) in the bycatch rate is unconstrained. Hence, unit costs and revenues due to the scale of catch may differ between absolute and relative credit systems.

Another penalty-reward system with a relative bycatch standard arises when the (indirect) subsidy applies to vessels with performance better than (below) the standard, and the (indirect) tax applies to vessels with relative bycatch rates above the standard (see Parry and Krupnick, 2011; Goulder et al., 2019 in environmental pollution).[8] In contrast with a relative transferable credit system, in which both the (indirect) tax and (indirect) subsidy apply to all covered vessels, such a system involves no "implicit output subsidy" from the unpriced residual bycatch to vessels that fail to meet the standard, and no tax on vessels that exceed the standard.

Due to the exigencies of sustainability and thereby absolute limits (performance or process standards), such as bycatch and target catch Total Allowable Catches, fisheries bycatch credit systems are invariably absolute rather than relative (rate-based). Nonetheless, relative credit systems could be applied when absolute limits are unavailable or unnecessary but the intent remains to reduce bycatch.

HABITAT IMPACT CREDITS

Transferable habitat impact quotas, first proposed as property rights (Holland and Schnier, 2006), can be extended to credit systems that can be applied to benthic habitat such as deep-water coral and sponge communities. Habitat impact in these cases is seen as one facet of the bycatch issue, but typically impacting a special type of species (e.g., cold-water corals and sponges) or the seafloor itself. Both of these unique features have sufficiently unique features to require separate and distinct discussion. The same basic economic principles of bycatch mitigation developed for bycatch reduction are applicable and developed here.

Transferable habitat impact quotas as a credit system directly address spatial management in a cost-effective manner. They can be combined with property rights or credit systems for catch (target, bycatch) or effort and technology standards

[7] Because an absolute TC program does not include the "implicit output subsidy" from the unpriced residual bycatch, the extent of standard variation across vessels (holding the total number of allocated limits fixed) does not affect a vessel operator's decisions at the margin (and thereby economic efficiency), and has only distribution consequences (see Goulder et al., 2019 for pollution).

[8] The environmental economics literature calls this approach a feebate.

such as prescribed gear and its operation.[9] They can be especially appropriate for gear such as groundfish trawls and scallop dredges that adversely impact the benthic habitat. They incentivize the use of skipper and industry-wide knowledge, unknown to the regulator, on a tow-by-tow basis. It can serve as an alternative to Territorial Use Rights for Fisheries (TURFs, Christy, 1982) – a form of spatial property rights, and provide a cost-effective alternative to permanent area closures such as Marine Protected Areas, or allow smaller and more tailored permanent area closures.

A transferable habitat impact quota credit system can be implemented at the industry or group level due to the complexity and cost of defining, observing, and enforcing habitat impact units allocated to individual vessels or even smaller groups of vessels, and indicators of group performance might be more easily monitored (Kotchen and Segerson, 2019). An industry-level program also circumvents the issue of fractional units and rare events for individual vessels and allows self-organization and regulation. For example, one endangered species of coral has such a small population that the bycatch limit is less than one coral per vessel over the course of a year. Vessel Monitoring Systems can facilitate time/area enforcement by continually monitoring vessels' location and rate of movement thereby reducing costs of implementation. Group or industry approaches, in the face of limited information and uncertainty about the impact of individual vessel actions on habitat, can promote information sharing (Abbott and Wilen, 2010) and the pooling of risks across the individuals comprising the group (Holland, 2018; Holland and Martin, 2019; Kotchen and Segerson, 2019). Group or industry approaches can also allow the group to collectively devise and implement solutions, using lower information and transactions costs than individual approaches, for multiple issues.[10]

Transferable credit can be applied in a cap-and-trade system with an aggregate cap – Transferable Habitat Impact, with the credit comprised of the unused portion of each vessel's or group's Transferable habitat impact quotas. Habitat impact quota could be held in reserve by the regulator. The world's first transferable habitat impact program was implemented as rights-based management in British Columbia (Wallace et al., 2015; Driscoll et al., 2017). Such a program could be implemented as a TC program.

CREDITS vs. PROPERTY RIGHTS

Management by transferable property rights or absolute TCs, when both are cap-and-trade, ostensibly appear to be the same. Both allow maximum bycatch (right or limit) subject to the overall fishery cap (e.g., Total Allowable Catch/Bycatch/Effort/Habitat Impact with the option to buy or sell allowances (rights or credits). Such trade confers production flexibility and lower costs through reallocation of bycatch reduction activity, leading to vessels that can reduce bycatch at lower cost. Thus, vessels that more readily reduce bycatch and at lower costs can be expected to create and then sell credits to vessels with greater difficulty and higher costs of reducing bycatch. Both property rights and credits result in a market price for bycatch and therefore a reduction in bycatch as its cost is incorporated into production decisions. Due to the "implicit output subsidy," by which residual bycatch does not receive a cost so that remaining bycatch is not fully costed, TCs are less cost-effective for each vessel and society as a whole than rights-based management (Fischer, 2001, 2003); this is more fully discussed below.

Rights-based and absolute TC bycatch management differ along several dimensions. (This discussion is based upon Goulder and Parry, 2008; Boom and Dijkstra, 2009; Sovacool, 2011; Nentjes and Woerdman, 2012; Helfand, 2013; Goulder et al., 2019 for environmental pollution.) These dimensions are reflected as follows: (1) absolute TCs are direct regulation made flexible through credits that are not rights or entitlements but limits made flexible through exchange of unused portions (credits); (2) rights pertain to the entire limit or quota whereas credits refer only to the unused portion; (3) absolute credits are created *gratis* by producers but rights are created by the regulator or society; (4) under absolute TCs, residual bycatch is not priced and hence is free of explicit cost to the producer (giving the "implicit output subsidy"), whereas the residual bycatch under rights is always priced and hence given a cost to the producer (because the residual is property) that has an opportunity cost (value of next best alternative) because the unused right can always be sold; (5) the two systems differ with respect to economic efficiency, distributional impacts, and incentives for bycatch-reducing technological change[11]; and (6) aggregate supply of rights in any given year when a fishery cap is fixed but not for credits, although credit supply is limited by the annual fishery cap.

The property right in its entirety is both owned and transferrable, whether actually used in total or in part (for environment, see Goulder and Parry, 2008; Boom and Dijkstra, 2009; Sovacool, 2011; Nentjes and Woerdman, 2012;

[9] In the language of economics, multiple externalities each require their own policy instrument unless the externalities are tightly linked. For example, overfishing of target species may be subject to individual transferable quotas and deep-water habitat may be subject to transferable habitat impact quotas.

[10] In the language of economics, a group approach can internalize multiple externalities, including the biodiversity and ecosystem service one of interest and vessel congestion. Group approaches can face moral hazard ("hidden action" and adverse selection ("hidden information") problems (Holland, 2018; Kotchen and Segerson, 2019). Successful groups may require homogeneous membership with well-aligned interests and/or formal contracts with monitoring and enforcement provisions and/or how the policy is designed (i.e., the rewards and penalties established by a specific policy) as well as the internal operating rules of the group itself. Without these conditions, non-compliance and free-riding may occur, in which one firm contributes more than its efficient level while the other firm contributes less (thereby free riding on the efforts of the other firm), and which contributes to economic rent and profit dissipation (Deacon, 2012; Kotchen and Segerson, 2019).

[11] Under relative (rate-based) TCs, since compliance depends on a ratio (e.g., sea turtle bycatch per mt of swordfish or per thousand hooks), vessels can influence their allowance allocations by changing their output swordfish) or input (hooks) levels during the compliance period (see for environment, Helfand, 2013; Goulder et al., 2019). In contrast, under rights-based management or absolute TCs, a vessel's allocation of rights or limits (respectively) is not influenced by within-period production changes. The dependence under relative TCs of the limit allocation on within-period production decisions has important implications for incentives and associated system performance. It significantly affects production levels, overall bycatch reduction, and the levels and distribution of costs.

Goulder et al., 2019). A credit only pertains to the unused portion of the limit, and does not entail a property right or entitlement to the entire limit, including the residual after the credit has been exchanged (Nentjes and Woerdman, 2012). TCs are complementary to direct regulation of bycatch and not a substitute, whereas rights-based management is a clear-cut substitute to direct regulation of bycatch.

Under TCs, both vessels entering the fishery or existing vessels receive mandated limits *gratis*, whereas vessels that exit or reduce bycatch lose the limits (see Nentjes and Woerdman, 2012 for environmental pollution). Under property rights, vessels typically receive rights *gratis* (although they can be auctioned or purchased from other vessels), and when exiting the industry or reducing bycatch, rights can be traded since they are property. Rights must typically be purchased when entering the fishery. Additional rights must also be purchased. The rights can be transferred and valued at the price from the rights market or as agreed upon between the vessels involved in the transaction. Both transferred and residual rights are valued at this price. The value of the residual rights forms an economic opportunity cost that the economically rational operator will incorporate into any profit-maximizing decisions (since it may be more profitable to lower the scale and scope of production and sell rights). Vessel exit does not diminish the quantity of rights held by multi-vessel firms, allowing greater reduction of fixed costs than under TCs and long-term consolidation of rights among fewer vessels.

Transferable credits do not directly place a price on limits, and therefore do not explicitly place an economic cost upon the residual bycatch that would be borne by the fisher (see from environmental pollution Goulder and Parry, 2008; Boom and Dijkstra, 2009; Sovacool, 2011; Nentjes and Woerdman, 2012; Goulder and Parry, 2008). TCs only prices the credit. The residual bycatch under the limit is neither a property right nor tradable.[12] Because the residual bycatch remains unpriced, the impact upon vessel decision-making and behavior is weaker than property rights (Goulder and Parry, 2008). This is also a major reason why TC is less efficient than rights-based management, since as noted the latter prices and thereby confers a cost upon the residual bycatch (because of the ownership conferred upon the residual bycatch). This non-priced and non-costed residual is called an "implicit output subsidy" in the environment literature (Fischer, 2001, 2003).[13]

Under rights-based management, the residual impact cost that is added to the other costs of producing bycatch raises the average and marginal cost and price of bycatch (if it has a price) (Boom and Dijkstra, 2009; Nentjes and Woerdman, 2012; Goulder et al., 2019). The price of the target catch and bycatch incorporates this higher cost (due to the residual bycatch). The higher cost and price incentivize vessels to fish at the optimum scale for both target catch and bycatch and scope (bycatch-target catch ratio) of production. The vessel operator has internalized the external cost of bycatch. In contrast under TCs, although the price of the bycatch will reflect the variable costs of production, this price will not include the cost of the unpriced bycatch residual.[14]

In sum, average and marginal production costs under absolute (and relative) TCs should be lower than under rights-based management due to the "implicit output subsidy," i.e., the unpriced and uncosted bycatch residual, with only the credit receiving a price and cost (developed in the pollution literature, see Fischer, 2001, 2003; Boom and Dijkstra, 2009; Nentjes and Woerdman, 2012; Goulder et al., 2019). The uncertainty due to the expected shorter duration and lower security of the vessel's limit under credits, compared to rights, also raises average and marginal costs. Lower average and marginal costs of target and priced bycatch hamper the ability to be cost-effective and result in economically sub-optimal economies of scale – product volume – and scope – bycatch-target catch ratio.

The "implicit output subsidy" of credits reduces the gains from trade, and hence lower costs, from credit trading (see Goulder et al., 2019 for the environment). Absolute TCs and rights-based management minimize vessels' costs by trading credits or rights until their marginal bycatch reduction costs equal the common credit or rights price. This maximizes the cost savings from trade, since in principle this trade equalizes the common credit or rights price across vessels.[15] In sum, fleet-wide costs of a program with the same stringency and scope are lowest with rights-based management, followed by absolute credit systems and with highest costs relative credit systems due to the "implicit output subsidy," i.e., unpriced and uncosted residual bycatch.

Rights-based management creates stronger incentives for bycatch-reducing technological change than TC, PWC, or their three-part policy instrument combination. This is again due to the "implicit output subsidy" from not pricing residual bycatch and therefore not conferring a cost to the residual bycatch. In addition, the limit and hence credit is of more limited duration than rights-based management. Hence, with lower cost of bycatch and greater uncertainty over the duration, incentives to innovate and adopt bycatch-reducing technological change are weaker compared to rights-based management.

Table 2 summarizes the main points of the above discussion for absolute TC and transferable property rights.

[12]Technically, the residual bycatch becomes private property from an unpriced common resource stock after capture. The absence of property here refers to an unpriced common resource that is not part of an allocated right to an amount of bycatch.

[13]A second type of implicit output subsidy arises with a relative standard rather than absolute performance or process standard (see Fowlie et al., 2016 and Goulder et al., 2019 for environmental economics). The relative standard, such as bycatch per unit of effort or target catch multiplied by the Total Allowable Effort or target Total Allowable Catch, increases the limits a vessel receives from the management authority and allows higher target bycatch.

[14]The difference in cost-effectiveness reflects the "implicit subsidy" to bycatch under TCs, which causes vessels' target catch output levels to exceed the levels consistent with minimizing the costs of achieving a given bycatch limit at would be achieved under rights-based management (see Goulder et al., 2019 for environmental pollution). Moreover, TCs are not expected to equalize marginal bycatch reduction costs across vessels even when credit trading is fully functional. This failure to satisfy the "equi-marginal principle" limits aggregate cost reductions from credit trades, because some vessels remain with higher costs than others, where trade would allow higher cost vessels to sell credits to lower cost vessels.

[15]Under relative TCs, a vessel minimizes its costs by trading credits until its marginal bycatch reduction costs equal the credit price (see Goulder et al., 2019 for the environment). This price generally differs across vessels, because it depends on bycatch reduction technologies that may be vessel-specific. This variation in technologies in turn prevents trading from equalizing marginal bycatch reduction costs across vessels, attenuating the gains from trade.

TABLE 2 | Cap-and-trade absolute transferable credits versus transferable property rights.

Feature	Transferable credits	Transferable property rights
Initial allocation	Gratis, created by vessel, firm.	Typically, gratis, but can be directly purchased by vessel or firm from management authority or by auction.
Entire versus residual bycatch	Pertain to only used portion of bycatch limit but not residual bycatch. Residual bycatch is not property right, tradable, or opportunity cost of foregone profits.	Pertain to both used and residual bycatch. Residual bycatch forms opportunity cost of foregone profits.
Divisibility	Freely divisible. No impediment.	Freely divisible. No impediment
Duration	Shorter, limited to one production period.	Longer than one production period, often into perpetuity.
Transferability	Pertains only to credit (unused portion of limit) but not to residual bycatch	Pertains to entire right, i.e., to entire amount of bycatch. Rights can be traded since they are property. Additional rights must be purchased.
Gains from trade (arbitrage efficiency)	Lower due to lower duration, lower volume of trade (only credits), and failure to equalize marginal bycatch reduction costs across vessels.	Higher due to longer duration, higher volume of trade (rent, lease, sell), and greater likelihood to equalize marginal bycatch reduction costs across vessels.
Exclusivity	Exclusive use to vessel or group allocated limit	Exclusive use to vessel or group allocated right
Security	Same	Same
Supply	Endogenously created by vessel, firm. Depends upon within-period production decisions. Overall limit exogenously created by management authority.	Exogenously created by management authority and equal to overall limit. Does not depend upon within-period production decisions.
Production costs	Lower due to residual bycatch without property right or cost (implicit output subsidy)	Higher since residual bycatch has property right and cost.
Strength of economic incentive to reduce bycatch	Weaker, since lower costs of production due to unpriced and uncosted residual bycatch (implicit output subsidy) and shorter duration of credit and limit	Stronger, since higher costs of production due to priced and costed residual bycatch and longer duration of credit and limit.
Strength of dynamic economic incentive to innovate bycatch-reducing technological change	Weaker because of implicit output subsidy, lower cost increase, and shorter duration of credit and limit	Stronger because entire bycatch is given a cost, hence vessels have stronger incentive to innovate in order to lower costs and longer duration of right
Cost-effectiveness/efficiency	Weaker, includes implicit output subsidy due to unpriced and uncosted residual bycatch. Smaller gains from trade.	Stronger because entire bycatch receives price and cost incentivizing vessels to reduce scale of production and lower bycatch-target catch ratio. Larger gains from trade.
Entry/exit	Entering or existing vessels receive mandated limits *gratis*. Vessels that exit or reduce bycatch lose the limits.	With property right, all or part of bycatch can be traded upon exit. Vessel exit does not diminish the quantity of rights held by multi-vessel firms.
Relationship to direct regulation	Direct regulation made flexible, and thereby complement	Substitute to direct regulation

FISHERY EXAMPLES

The Scottish Conservation Credit Program

In response to declining cod stocks and the need to reduce bycatch in the Scottish trawl fishery under European Union regulation, the Scottish Conservation Credit Program (SCCP) started in 2007 after the European Commission implemented the second revision of the Cod Recovery Plan (Van Riel et al., 2015). This Plan required further reductions in fishing effort in North Sea cod fishing grounds. The European Union's full retention policy subsequently replaced the SCCP.

The SCCP combined private, voluntary solutions with direct regulation and incentive-based management through a compulsory absolute penalty-and-reward credit system (PWC) system to reduce cod (*Gadus marhua*) target catch in the Scottish North Sea mixed species demersal fishery and low-value cod bycatch from the Norway lobster (*Nephrops norvegicus*) fishery (hereafter all material is from WWF, 2009; Fernandes et al., 2011; Holmes et al., 2011; Scottish Government, 2011; Curtis, 2017). The trawl fishery operators were required to discard over-quota and undersized cod, both perversely incentivized by the direct regulation quota. The SCCP aimed to allow fishing for other species while avoiding cod and reducing overall effort. The program allowed enough days for vessels to catch their quota while not increasing cod removals and thereby allow cod stock recovery.

The SCCP achieved bycatch avoidance through time-area closures and real-time spatial management triggered by monitoring the cod catch per hour of tow. Minimization of bycatch was achieved through optional and voluntary technology standards, notably gear (Orkney trawl, square mesh panels, "one net rule" that ensure only regulated, more selective gear is used) and operating requirements, such as the move-on requirement when the catch rate is exceeded, and post-interaction activities such as release alive. Some policy instruments were complements and some were substitutes. Over time, the aggregate Total Allowable Effort was progressively tightened.

The SCCP incentivized changes in fisher behavior and decision-making through an absolute PWC system, notably days-at-sea allocated each year as an annual limit, not as a property right for future years. Days eventually transformed from nominal days to kilowatt days-at-sea reflecting the relative fishing power of the heterogenous vessels comprising the fleet. Vessels with <1.5% cod as a proportion of total catch were already exempt from days-at-sea limit. European Union capacity reduction and the Cod Recovery Plan created surplus days for rewards in the SCCP rather than explicitly retaining days from the Total Allowable Effort for rewards or relying upon penalty. There was no attempt to achieve neutrality by balancing penalty-and-reward days.

Penalties to incentivize avoidance and minimization of bycatch include lowering the vessel's balance of days to penalize non-compliance, including fishing in closed areas. Rewards include extra kilowatt days-at-sea to compensate for foregone catch created when vessels avoid cod-dense areas through voluntary real-time spatial management, using more selective gear, and more generally by demonstrating low cod catch. The SCCP further rewarded vessels through an allowance to use the days more flexibly, operating under hours- rather than days-at-sea, thereby inducing fuel conservation and more efficient operation and lower costs. Days could also be transferred between vessels, introducing elements of TC. Days were not consolidated on a fewer number of vessels with limited days per vessel.

The role of direct regulation was crucial. Although the PWC system created incentives, the direct regulation allowed the incentives to be effective. The Cod Recovery Plan immediately created an untenable situation due to cod quotas, and the hard limits for days annually declined from 330 to 180, incentivizing participation in the SCCP. The SCCP would not have worked without effective monitoring (through Vessel Monitoring Systems, observers, on-board cameras, matching processor purchases to vessel sales, limited number of ports to land, logbooks) and enforcement.

Halibut Bycatch Reduction in the Alaska Multispecies Groundfish Trawl Fishery

The large-scale groundfish trawl fishery in Alaska is comprised of five companies and 18–22 catcher processors that receive target catch share allocations and is managed by two harvest cooperatives (Alaska Seafood Cooperative and Alaska Groundfish Cooperative) (Abbott and Wilen, 2010; Abbott et al., 2015; Little et al., 2015; Fina, 2017). The companies also receive prohibited species catch allowances for halibut, red king crab, snow crab, and tanner crab that are allocated by historical usage of targets rather than bycatch history, thereby avoiding the moral hazard problem of rewarding those with high bycatch history. The five companies cooperate by meeting a minimum of once per month and engaging in regular communications. Companies (vessels) could vest their shares in a cooperative formed by participating members. Cooperatives are internally managed and provided with considerable flexibility to internally allocate catch allowances.

The bycatch program for the prohibited halibut was initiated due to high rates of economic discards in a derby fishery that arose in response to the fleet-wide bycatch caps (Abbott and Wilen, 2010; Abbott et al., 2015; Little et al., 2015; Fina, 2017). The North Pacific Fishery Management Council directed the cooperatives to develop halibut avoidance plans. Through co-management, the Council established a voluntary bycatch performance standard with a limit and then let the industry devise its own way to satisfy the limit. Each cooperative sets a fixed tonnage halibut allowance based upon historical halibut catch. This allowance is distributed among its vessels based upon groundfish target allocations. Vessels must meet halibut bycatch rate standards based upon history in a relative bycatch performance standard of the ratio of halibut to groundfish. Each flatfish species has an annual relative performance standard.

Each cooperative defines best-practice halibut avoidance and minimization of bycatch (Abbott and Wilen, 2010; Abbott et al., 2015; Little et al., 2015; Fina, 2017). Bycatch avoidance and minimization are achieved by best-practice process and technology standards. The process standards for avoidance included fishing target choice of location and time of day to fish. The technology standards for minimization include small test tows when entering an area, halibut excluders, and deck sorting to quickly return halibut bycatch to the water. Excluders generate target catch losses (opportunity cost) and can obscure deck sorting once the net is emptied on the vessel. Avoidance was also realized through real-time spatial management using regular vessel-to-vessel communication, including weekly bycatch conference calls by captains (Little et al., 2015). Such communication was found to be faster and more effective than through the commercial company Sea State that collects information in the Alaskan salmon bycatch credit program in the Alaskan pollock fishery (Bersch, 2013; Mize, 2014; Little et al., 2015). Internal cooperative and vessel cooperation leads to faster communication.

"Yardstick" management (see Shleifer, 1985 for economics of regulation) is practiced for the relative PWC (with an overall cap). An annual test, based on historical targets ("yardstick") eliminates bycatch excess (Fina, 2017). Vessels must achieve halibut bycatch rates based on historical average fleet performance ("yardstick"), with rates decreasing across 3 years. Bycatch in excess of the "yardstick" incurs a monetary penalty of US$25,000 – US$100,000 per violation that escalates by the amount of target catch. A low catch threshold allows the vessel operator to avoid a penalty if bycatch is kept at low levels. Vessels can sell target quota to another vessel, which implicitly constitutes a trade of the relative bycatch-target catch-bycatch credit. There is in effect very little trade. Quarterly monitoring applies to vessels that fail an annual test. Halibut limit forfeitures without redistribution form another penalty. Reallocation would otherwise create moral hazard through a perverse incentive of discouraging communication.

The fourth quarter test applies an aggregate relative rate performance standard to all flatfish targets (12.1 kg halibut per mt of groundfish), thereby addressing another moral hazard problem (Fina, 2017). Vessels approaching the end of the year with substantial amounts of bycatch quota would otherwise not face an economic incentive to maintain avoidance through the end of the year (in this case, the economic incentive's relative price effect overwhelms any intrinsic motivation held by vessels).

CONCLUDING REMARKS

Credit systems are a form direct regulation of target catch, bycatch, effort, or habitat impact through performance or process standards, implemented through limits or quotas made flexible. There are three fundamental types of these incentive-based approaches. The first, transferable credit systems, are cap-and-trade and create a price in the credit market. The first evolved out of direct regulation made flexible through the creation of unused limits – credits – with the option to compensate excess use of one source by excess control of another source. The second type of credit system incentivizes changes in fisher behavior and decision-making through penalties and rewards in either cash or kind for increasing or decreasing bycatch, respectively. This two-part policy instrument combines a reward (indirect subsidy) with a penalty (indirect tax). The third type combines the transferable credits and cap-and-trade with penalty-and-rewards to create a three-part policy instrument.

Credit systems, as incentive-based policies, are potentially cost-effective for producers and hence the fishery and society writ large. Credit systems can easily complement and incentivize adopting technology standards, such as required gear design and operating standards, and further advancements in bycatch reducing technological change. Bycatch credit systems can be either absolute or relative, the latter specified as the ratio of bycatch to target catch of a species or as the ratio of bycatch to an essential individual input such as gear or fishing time. Relative bycatch credit systems do not necessarily impede the continued growth of the relevant target species. Credit systems can be formed solely at the industry level, at the individual vessel level, or a mixture of both.

Credit systems are particularly promising when rights-based management is not possible. Without property rights, credit allocations are of limited duration, potentially revocable, with less at stake, and less uncertainty. Entry into the fishery is readily accommodated. Credit systems may be are particularly promising for fisheries in which multilateral coordination – typically through consensual decision-making – is difficult to achieve with as "permanent" of a policy instrument as property rights. Hence, credit systems are potentially very promising for international fisheries. Credit systems price bycatch but do not price residual bycatch or habitat impact, creating the "implicit output subsidy," so that the resulting increase in costs due to batch is lower and the incentives are weaker than under rights-based management.

The Scottish credit system illustrates how to successfully design a penalty-reward (indirect tax-subsidy) system centered around days-at-sea to incentivize avoiding bycatch and technology standards of gear design to incentivize minimizing bycatch. A crucial feature is the threat of more stringent direct regulation. The Alaska multispecies trawl system also illustrates successful application of the first type of credit system to incentivize bycatch avoidance and minimization. A crucial feature entails the application at the group level, with the fishery management authority providing overall guidance, monitoring, and enforcement. A second crucial feature entails allowing individual multi-vessel companies to reduce bycatch by internally reallocating credits within the company from a source able to successfully reduce bycatch at a lower cost than another source.

In sum, credit systems provide an incentive-based approach to bycatch reduction that can stand alone or complement other policies. Besides readily complementing technology standards such as gear and equipment requirements, credit systems provide additional flexibility and a means to address otherwise unmanaged components of direct bycatch regulation, such as time-area management. Credit systems can also complement capacity management that uses cap-and-trade credits or rights-based management through penalty-reward credits for age and size-related issues not otherwise readily addressed by a property right or credit on catch or effort. Credit systems provide a credible, and in some ways superior, alternative to rights-based management. Credit systems are generally superior to rights-based management in international fisheries and even national fisheries where resistance to rights-based management, or insurmountable difficulties in reaching agreement, limits their use. Although credit systems may be "second best" to rights-based management, they still provide improvements in bycatch reduction and economic benefits compared to a total absence of bycatch measures. When management authorities seek an alternative to direct regulation, credit systems offer a promising alternative, since they grew out of direct regulation made flexible and cost-effective. Credit systems may also serve as an intermediate step between direct regulation and rights-based management. Finally, examples already in place in the environmental policy realm, such as cap-and-trade in carbon markets, can enhance the attractiveness of such measures in multilateral fishery management.

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An Economic Perspective on Policies to Save the Vaquita: Conservation Actions, Wildlife Trafficking, and the Structure of Incentives

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The Upper Gulf of California is a diverse and highly productive ecosystem supporting some of the most important fisheries in Mexico, yet a history of weak fisheries management and illegal fishing threaten the area's biodiversity and undermine human well-being in the communities along its shores. The vaquita (*Phocoena sinus)* is endemic to these waters and is on the brink of extinction due to incidental entanglement in gillnets used by small-scale fishers. The resurgence of an illegal gillnet fishery for totoaba (*Totoaba macdonaldi*), whose swim bladders are highly prized in Hong Kong and continental China, has accelerated the steep decline of the vaquita population. Vaquita is one of a growing number of cases linking illegal wildlife trade, organized crime, and biodiversity decline. This paper provides a summary of key reflections of a panel of fisheries economists gathered at the ninth forum of the North American Association of Fisheries Economists (NAAFE) to evaluate the policies implemented in the Upper Gulf through an economic lens and updated to reflect more recent developments. The panel recognized that poor fisheries management, lack of effective enforcement, distant demand for an illegal product, corruption, and few viable economic alternatives confound efforts to address vaquita bycatch. The complexity of these problems requires a holistic, multidisciplinary approach, combining top-down, direct regulation and bottom-up, participatory and incentive-based instruments.

Addressing chronic deficiencies in enforcement, particularly in the very small area where the remaining vaquitas are found, is crucial to prevent imminent extinction. Equally crucial are sustained actions to support legal fishers able to make a good living – with a direct stake in healthy marine ecosystems – as key components of policies to address bycatch and reduce wildlife trafficking. The situation in the Upper Gulf of California is dire, yet similar threats to other marine mammals and wildlife trafficked species may benefit from the experience of the vaquita.

Keywords: vaquita, totoaba, bycatch, ilegal wildlife trafficking, incentive-based management, conservation

INTRODUCTION

Mexico's endemic porpoise, the vaquita (*Phocoena sinus*), is the world's most critically endangered marine mammal (International Union for the Conservation of Nature [IUCN], 2020b). It is endemic to the upper Gulf of California (UGC) where unsustainable bycatch in small-scale gillnet fisheries has long been recognized as the only factor driving the species toward extinction (Norris and Prescott, 1961; Brownell, 1988; Rojas-Bracho and Taylor, 1999; D'Agrosa et al., 2000; Rojas-Bracho et al., 2006). Since the early 2010s, the resurgence of illegal fishing for totoaba (*Totoaba macdonaldi*), driven by the black market for totoaba swim bladders in Hong Kong and continental China, has caused vaquita numbers to plummet at nearly 50% per year (Jaramillo-Legorreta et al., 2019). In summer 2018, the population estimate indicated fewer than 19 individuals remain (Jaramillo-Legorreta et al., 2019). The last best estimate from the area where vaquitas were most recently detected acoustically estimated 10 individuals, including 3 calves, and all appeared in good health (Rojas-Bracho et al., 2020). Ensuring protection of these surviving vaquitas from gillnets could still save the species (Morin et al., 2020; Rojas-Bracho et al., 2020; International Whaling Commission [IWC], 2021).

A number of governmental policies and programs have been enacted in the UGC to protect vaquita from gillnets (see reviews in Rojas-Bracho et al., 2006; Bobadilla et al., 2011; Rojas-Bracho and Reeves, 2013; Cisneros-Montemayor and Vincent, 2016). Despite these efforts, the region continues to experience widespread illegal fishing with gillnets, loss of income and markets for legal fishers, and the continued decline of the vaquita population (Comité Internacional para la Recuperación de la Vaquita [CIRVA], 2019; United Nations Educational, Scientific and Cultural Organization [UNESCO], 2019; International Union for the Conservation of Nature [IUCN], 2020b; International Whaling Commission [IWC], 2021). Illegal and unsustainable fishing threatens the region's rich biodiversity and undermines the economic and human potential of the communities. Formal and informal institutions[1] are challenged to protect the region's natural resources due to the increasing presence of organized crime and a history of tolerated corruption and non-compliance with regulations (C4ADS, 2017; Crosta et al., 2018; Environmental Investigation Agency [EIA], 2019; Aceves-Bueno et al., 2020; Felbab-Brown, 2020).

This article contends that resolving protected species bycatch in small scale fisheries requires addressing underlying issues of fisheries management and governance and, equally crucially, in finding viable alternative methods of fishing and economic activities for local communities. In the case of vaquita, the need for a multifaceted approach has been championed by the international vaquita recovery team since its inception (Comité Internacional para la Recuperación de la Vaquita [CIRVA], 1997, 2014, 2019), recommended by international organizations (United Nations Educational, Scientific and Cultural Organization [UNESCO], 2018; International Whaling Commission [IWC], 2019; International Union for the Conservation of Nature [IUCN], 2020b), scientists (Rojas-Bracho et al., 2006; Bobadilla et al., 2011; Rojas-Bracho and Reeves, 2013; Aburto-Oropeza et al., 2016; Cisneros-Montemayor and Vincent, 2016; Aceves-Bueno et al., 2020), and recognized in the policies of the Government of Mexico. However, enacted policies failed to fully embrace the long-term investment necessary for successful community engagement and economic development that can generate buy-in and improve compliance by providing lasting benefits. These well-intentioned programs also failed to consider the consequences of conservation policy on local communities. The problems in the UGC were exacerbated well beyond the ability of traditional fisheries management to provide solutions due to corruption and the related illegal and lucrative alternative, the black market for totoaba swim bladders.

A panel convened at the ninth forum of the North American Association of Fisheries Economists reviewed conservation actions that had been implemented to protect vaquita and support local communities. This paper provides a brief history of the government policies applied in the region, followed by a review of socio-economic instruments and impacts, both intended and unintended. Key points and recommendations from the panel are summarized and updated to reflect policies enacted through early 2021. The focus is on the interplay between top-down, direct regulation and bottom-up, incentive-based[2] approaches (Squires et al., 2021), the external pressures from

[1] Understanding institutions as the "rules of the game," both the formal legal rules and the informal social norms that govern individual behavior and structure social interactions (North, 1990).

[2] "Incentive-based" instruments refer to all kinds of instruments that could change human behavior without having to command the change in behavior via a direct regulation. Such kinds of instruments could include: property-based instruments (fishing quotas, territorial use rights for fisheries); economic instruments (taxing the conduct or activity that the regulator wants to decrease); market instruments (cap and trade policies); or correcting information asymmetries, among others.

the totoaba black market, and how these factors altered the structure of incentives that drive decisions about fishing in the region. While the situation for vaquita is dire, similar threats to other marine mammals (Brownell et al., 2019) and other trafficked wildlife species (Felbab-Brown, 2017) may benefit from the experience of the UGC.

BACKGROUND

Small-Scale Gillnet Fisheries in the Upper Gulf of California

The UGC is a highly productive marine ecosystem supporting some of the most profitable and seasonally diverse small-scale fishery resources in Mexico (Erisman et al., 2015; Brusca et al., 2017). Gillnets have been used in the UGC since the 1930s, initially for sharks and totoaba, and then adapted for other species over the ensuing decades. With mesh sizes ranging from about 7 to 30 cm depending on target species, the nets are easy to deploy and cost-effective to use given the strong tidal currents in the area. Currently, gillnets are used to harvest several high value species: blue shrimp (*Litopenaeus stylirostris*) and brown shrimp (*Farfantepenaeus californianus*) from September to March; Gulf corvina (*Cynoscion othonopterus*) in March and April; and sharks, rays, and several kinds of finfish, such as bigeye croaker (*Micropogonias megalops*) and Spanish mackerel (*Scomberomorus concolor)* from February to June (Cudney-Bueno and Turk-Boyer, 1998; Erisman et al., 2015). These fisheries are major contributors of income, employment, and food security in the coastal communities and also provide a strong sense of cultural identity and social relevance (Lluch-Cota et al., 2007).

The most recent assessment of small-scale fisheries in the region (Pérez-Valencia et al., 2015; **Appendix 1**) indicated that there were 1,688 fishing licenses for 876 registered pangas. Pangas are fiberglass, outboard-powered boats 6 to 8 meters long, each operating with two or three local crew members (Cudney-Bueno and Turk-Boyer, 1998). Given the multiple target species, most fishers need two or more licenses to fish throughout the year, the average being 1.92 licenses per panga. **Appendix 1** shows the distribution of licenses and pangas among the three main fishing towns in the UGC in 2015, the most recent year for which reliable data were available[3]. The level of fishing effort in 2015 reflects the impacts of policies analyzed in this paper.

The local economies of San Felipe in Baja California (population in 2018 ∼ 19,000) and El Golfo de Santa Clara in Sonora (population in 2010 ∼ 4,000) are largely dependent on fisheries that overlap with vaquita habitat (Cudney-Bueno and Turk-Boyer, 1998; Erisman et al., 2015). The Cucapá, an Indigenous community of the Colorado River Delta, and small-scale fishers from Puerto Peñasco in Sonora (population in 2020 ∼ 62,000), fish in the UGC but primarily outside of vaquita habitat.

A broad range of problems threaten the long-term sustainability of the region's fisheries resources and biodiversity, including the population trends for vaquita and totoaba (Rojas-Bracho and Reeves, 2013; Cisneros-Montemayor and Vincent, 2016; Cisneros-Mata, 2020). The complex socio-ecological context, and a host of underlying institutional deficiencies at multiple levels, hinder efforts to implement more sustainable practices. These include a lack of inter-agency coordination, conflict among stakeholders and between fisheries and conservation policies, limited institutional capacity, lack of enforcement and compliance, corruption, and weak fisheries management that has resulted in open-access, overcapitalized fisheries with high levels of IUU (illegal, underreported unregulated) fishing (Lluch-Cota et al., 2007; Cisneros-Mata, 2010; Bobadilla et al., 2011; Erisman et al., 2011; Cisneros-Montemayor et al., 2013; C4ADS, 2017; Pasini et al., 2017; Crosta et al., 2018; Mangin et al., 2018; International Union for the Conservation of Nature [IUCN], 2020a,b; Cisneros-Mata, 2020; Felbab-Brown, 2020).

Before the re-emergence of the illegal totoaba fishery, gillnet fisheries for blue shrimp were the most profitable. Most of the product was exported to the U.S. market where the high quality, large size class UGC shrimp commands high prices (Ardjosoediro and Bourns, 2009; Mesnick et al., 2019). From 2001 to 2011, 53 and 58% of gross revenues from fishing in El Golfo de Santa Clara and San Felipe, respectively, were from blue shrimp (Erisman et al., 2015). Net annual profit per panga for shrimp has been estimated at 2,200 USD in El Golfo de Santa Clara and 2,700 USD in San Felipe (Barlow et al., 2010). Total ex-vessel gross revenues from all target species for both communities are estimated at less than 20M USD (Barlow et al., 2010; Erisman et al., 2015). Net earnings, however, vary widely among permit owners depending on the number of boats and permits they hold. While these are admittedly imprecise estimates of income, they nevertheless provide a general indication of the net income from fishing and therefore, provide a useful basis for considering the attraction of other economic activities.

Totoaba Poaching and the Black Market for Totoaba Swim Bladders

Totoaba – a large (up to 2 m and over 100 kg) long-lived sciaenid that congregates in large schools to spawn in the shallow waters of the UGC – has driven the development and fate of the communities of the UGC for nearly a century. The high value of totoaba brought early prospectors to the UGC in the 1920s as they responded to demand for the fish's swim bladder. The swim bladders were initially consumed by the region's Chinese immigrants and then later exported to China and the Chinese communities of California for a highly valued soup, while the meat was most often left on the beach (Chute,

[3] Fisheries authorities never completed the census of fishers in the UGC. However, given the UGC is a designated natural protected area, all economic activities (including fisheries) are required to present an Environmental Impact Assessment (EIA). Fishers without an EIA authorization would not be able to obtain the permits issued by the port authorities to go fishing; this makes the list of vessels and fishers included at the EIA the most reliable data on the real fishing effort in the region. In 2015, an expansion of the Vaquita Refuge as a No Take Zone, was a *de facto* fishing ban. Since then, fishers have not presented an EIA, making the information from 2015 the most reliable data for the fishing effort in the region.

1928, 1930; Flanagan and Hendrickson, 1976; Cisneros-Mata et al., 1995). The history of totoaba in the UGC demonstrates lost opportunities across multiple fisheries sectors, losses that could have been prevented with stronger fisheries management (Cisneros-Mata, 2020; see also Mangin et al., 2018).

In the 1920s a commercial fishery was established after an agreement between the United States and Mexico to develop a market for the whole fish (Cisneros-Mata et al., 1995). Totoaba soon became a highly prized commercial and sport fish, responding to a growing U.S. market where the fillets commanded high prices (Cisneros-Mata et al., 1995; True, 1996). As the fishery grew, early fishing methods evolved and efficient, large-mesh nylon gillnets became the preferred gear (Flanagan and Hendrickson, 1976). Annual yields increased rapidly and trade in totoaba with the U.S. promoted the development of roads and fishing infrastructure in the UGC communities (Flanagan and Hendrickson, 1976).

Fishing effort peaked in the 1940s. By the 1970s, totoaba landings had plummeted due to intensive overfishing, large numbers of juvenile totoaba bycaught in commercial shrimp trawls, and fishermen moving into the then more lucrative shrimp fishery (Cisneros-Mata et al., 1995; Cisneros-Mata, 2020). Mexico banned commercial and sport fishing for totoaba in 1975. The species was listed on CITES (Convention on International Trade in Endangered Species of Wild Fauna and Flora) Appendix 1 (1977), placed on the The U.S. Endangered Species List (1979) and the Mexican List of Endangered Species (1994), and has been listed as “Critically Endangered” by the IUCN since 1996 (but see Cisneros-Mata, 2020). Gillnets targeting totoaba were prohibited (1992) and trade in totoaba or any part of a totoaba is illegal under Mexican law, U.S. law, and CITES. Yet throughout this period, poaching of totoaba continued (Cisneros-Mata, 2020).

While there were some indirect indications of recovery of the totoaba by the early 2010s (Valenzuela-Quiñonez et al., 2015), a rapid and likely unsustainable resurgence of illegal fishing was also underway (Cisneros-Mata, 2020). The poaching has been driven by increasing demand in China where rising incomes and an expanding middle class fuel a market for totoaba swim bladders (known as “maw”), which are prized in traditional medicine and as gifts and investments (C4ADS, 2017; Crosta et al., 2018). Compounding the problem, criminal networks began trafficking in totoaba swim bladders and developed the black market, taking advantage of routes used for other illegal products, including narcotics (C4ADS, 2017; Crosta et al., 2018; Ladkani, 2019; Aceves-Bueno et al., 2020; Belhabib et al., 2020; Felbab-Brown, 2020).

The price of one kilogram of dried swim bladder in southern China has varied over time, ranging upwards of 20,000 to 80,000 USD or more (Environmental Investigation Agency [EIA], 2016a,b; Crosta et al., 2018). Most prized are the swim bladders of adult females, with large (over 1 kg), high-quality swim bladders garnering the highest prices. At the peak of prices in 2012, such a swim bladder could fetch over 155,000 USD (Environmental Investigation Agency [EIA], 2016b). UGC fishers are reported to receive between 3,500 and up to 8,500 USD per kilogram (Environmental Investigation Agency [EIA], 2016b; C4ADS, 2017; Crosta et al., 2018), a fraction of the retail value but an enormous income for a local fisher relative to other income sources (for comparison, ex-vessel price for blue shrimp are ∼12 USD per kilogram; Erisman et al., 2015), and a temptation otherwise honest fishers may not be able to resist (Aceves-Bueno et al., 2020). One night of fishing with a catch of a few totoabas can earn a fisher in the UGC more than what would be earned in one year of legal fishing (Crosta et al., 2018). Catching small totoabas will pay for the costs of fishing, but catching one or more large female totoabas is like winning the lottery; one fisher reported earning 116,000 USD in a single day of fishing (pers. comm. with local fishers[4]).

Given the expected revenues from totoaba poaching relative to revenues from legal fishing, prohibitions on totoaba fishing and the use of gillnets meet with considerable resistance from many fishers. Totoaba swim bladders have been dubbed ‘aquatic cocaine’ for their high value, and the illicit trade is enabled and fueled by corruption, poor enforcement, and lack of compliance with regulations (C4ADS, 2017; Crosta et al., 2018; Environmental Investigation Agency [EIA], 2019; Felbab-Brown, 2020; Cisneros-Mata, 2020). Organized crime is gaining increasing control of the region’s fishing activities, including for shark, shrimp, and corvina (C4ADS, 2017; Belhabib et al., 2020; pers. comm. with local fishers (see text footnote 4)). Organized criminal groups are also supplying gear and financing some of the costs of poaching, with some fishers becoming increasingly indebted to the cartels, making it even more difficult to break the cycle of illegal fishing or incentivize compliance with regulations (Crosta et al., 2018; Ladkani, 2019; Alberts, 2021). Social unrest and violence are becoming more frequent as fishers feel they have few alternatives, criminal networks compete for control, and illegal fishers clash with efforts to protect vaquita from gillnets (C4ADS, 2017; Crosta et al., 2018; International Union for the Conservation of Nature [IUCN], 2020a).

Vaquita Bycatch and Status

As with many small cetaceans around the world, gillnets present the greatest threat to vaquita survival (Rojas-Bracho and Taylor, 1999; Rojas-Bracho and Reeves, 2013; Brownell et al., 2019). Gillnets set for totoaba pose the greatest risk due to the intensity of fishing, overlap with core vaquita habitat, and fishing practices with large mesh size nets. Other potential threats that have been suggested, but no evidence exists to support them as significant risk factors, including inbreeding depression, pollutants, and ecological changes as a result of reduced flow from the Colorado River (Rojas-Bracho and Taylor, 1999; Brusca et al., 2017; Flessa et al., 2019; Gulland et al., 2020; Morin et al., 2020). Vaquita are listed under CITES Appendix 1 (1979), on the U.S. Endangered Species Act (1985) and the Mexican List of Species at Risk of Extinction (1994), and have been listed as “Critically Endangered” by the IUCN since 1996.

In 1997, the first abundance survey of vaquita estimated 567 individuals [95% confidence interval (CI) 177 – 1,073]

[4]Local fishermen in San Felipe, personal communication to Lorenzo Rojas Bracho, communicated to the authors

(Jaramillo-Legorreta et al., 1999). Just over a decade later, a visual and acoustic survey documented 245 animals (95% CI 68–884) indicating an average annual decline of 7.6% attributed to mortality in gillnets set for shrimp and finfish (Gerrodette and Rojas-Bracho, 2011). Since the resurgence of gillnet poaching for totoaba, continued visual and acoustic monitoring of the population has documented a steep decline at about 50% per year (Thomas et al., 2017; Jaramillo-Legorreta et al., 2019). The number of vaquitas was estimated to be 59 in 2015 (Baysian Credible Interval (CRI) 22–145; Taylor et al., 2016) and less than 19 in 2018 (CRI 6–19; Jaramillo-Legorreta et al., 2019). As noted above, a recent photo-identification effort focused in the area where vaquitas were most recently detected acoustically, estimated 10 individuals, including 3 calves (Rojas-Bracho et al., 2020). Researchers last sighted vaquitas in November 2020 (International Union for the Conservation of Nature [IUCN], 2020a). Since then, acoustic monitoring has been suspended for the season due to the intensity of illegal fishing.

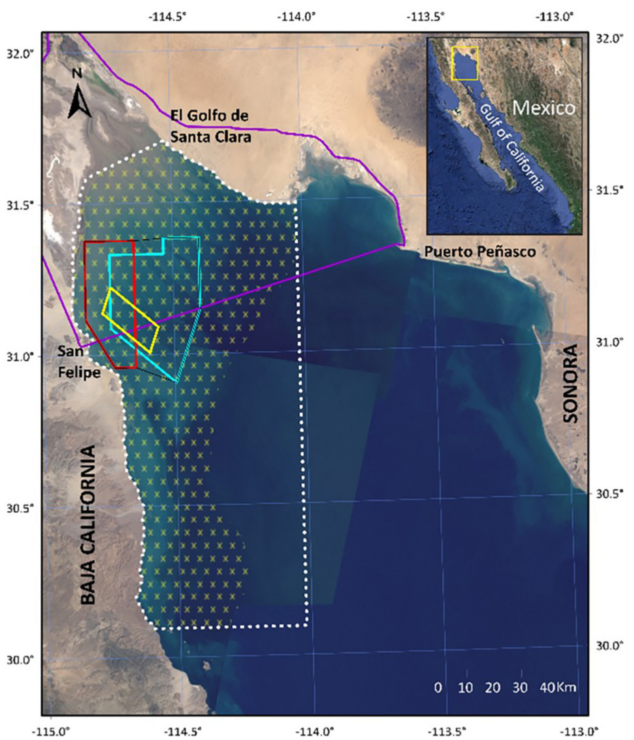

FIGURE 1 | Historical distribution of vaquitas (yellow hatched area) in the upper Gulf of California. The Upper Gulf of California and Delta of the Colorado River Biosphere Reserve (outlined in purple), designated by UNESCO in 1995, because of the unique habitat and presence of endangered species. The Vaquita Refuge (agreed to in 2005 and enacted in 2008 as a no fishing zone) is outlined in aqua blue. The gillnet exclusion zone (where fishing with gillnets is banned but other types of fishing are allowed) was given straight boundaries (dotted white) described by single latitude and longitude to facilitate enforcement and enacted in 2015 (Comité Internacional para la Recuperación de la Vaquita [CIRVA], 2014). Due to the drastic decline in vaquita numbers due to the resurgence of the illegal totoaba fishery, an enhanced enforcement zone (red) was recommended by Comité Internacional para la Recuperación de la Vaquita [CIRVA] (2017) in the area where the remaining vaquitas are thought to spend most of their time and that has high levels of illegal totoaba fishing effort. The Zero Tolerance Area (ZTA) is where CIRVA recommends nets must be removed within hours of being set (outlined in yellow). Despite these designations for vaquita protection, gillnets continue to be used and the population continues to decline toward extinction. Landsat satellite composite imagery provided by U.S. Geological Survey, National Aeronautics and Space Administration (NASA) and Esri, Inc. Projection UTM. Datum WGS84. Figure and caption adapted from Comité Internacional para la Recuperación de la Vaquita [CIRVA] (2019) and Jaramillo-Legorreta et al. (2019).

Vaquita have the smallest range of any cetacean (**Figure 1**). The remaining individuals appear to inhabit a tiny area (288 km^2 or roughly 12 × 24 km) designated as the Zero Tolerance Area (ZTA) (Comité Internacional para la Recuperación de la Vaquita [CIRVA], 2019). The ZTA lies within the legally defined Vaquita Refuge and within sight of San Felipe (**Figure 1**). There are a number of reasons to believe that if vaquitas were immediately protected from gillnets throughout their range, but particularly in the ZTA, the population could recover (International Whaling Commission [IWC], 2021). These few remaining vaquita appear in robust health, have low levels of pollutants, can calve every year (instead of every 2 years as formerly believed), and have persisted with low levels of genetic diversity but no signs of inbreeding depression (Taylor et al., 2019; Gulland et al., 2020; Morin et al., 2020; Rojas-Bracho et al., 2020). Yet, even if complete protection of these remaining vaquitas were guaranteed, recovery would take decades (Taylor et al., 2016, 2019). Saving vaquita therefore requires pursuing both immediate actions to prevent extinction and sustained, long-term efforts to permanently transition fisheries in vaquita habitat away from gillnets.

NATIONAL POLICIES IN MEXICO FROM 2007 TO 2018

Bobadilla et al. (2011) describe multiple periods of conservation action in the UGC. Initial efforts (1950–1970) focused on protecting commercial fish stocks; in a second period (1970–1990) efforts shifted toward social issues and promoting growth in small-scale fishing; and in a third period (1990–2007) the focus was on sustainable development. A fourth period, beginning in 2007, centered on single-species conservation driven by the steep decline of the vaquita population. Most of this paper addresses this fourth period, which featured a variety of policy instruments designed to reduce or eliminate gillnets in vaquita habitat and to compensate fishers for lost fishing income through economic instruments such as buy-outs and other monetary compensation programs. The paper finishes with a brief update on 2019-present (early 2021), which has been characterized by a lack of cohesive national policies in the region and the enactment of international trade sanctions.

From 2007 to 2018, the Mexican government invested heavily (around 145M USD) to implement programs and actions to prevent the extinction of the vaquita while providing support for the local fishing communities (**Figure 2**, **Table 1**, and **Appendix 2**). In this period, three programs were implemented to reduce or eliminate gillnets in vaquita habitat while providing for the livelihoods of fishers and related industries: (1) Vaquita Conservation Program for Sustainable Development in 2007, (2) Action Program for the Conservation of Species from 2008 to 2015 (*PACE-Vaquita*), and (3) Plan for the Comprehensive Care of the Upper Gulf of California (*Comprehensive Care Plan*).

The Vaquita Conservation Program for Sustainable Development (2007) was part of PROCODES, a nationwide subsidy program of the National Commission of Natural Protected Areas (CONANP). The focus was to provide specific alternative livelihoods in exchange for fishing licenses (Comisión Nacional de Áreas Naturales Protegidas [CONANP], 2009). In 2007, the Government of Mexico announced the Program for the Conservation of Species at Risk (PROCER), and developed a more specific instrument for vaquita: the PACE-Vaquita (2008–2015). PACE-Vaquita was a voluntary, multi-faceted program with four main components:

(1) Buy-out with alternative livelihoods: granted fishers the opportunity to start a new business in exchange for permanently surrendering their fishing license(s) and was an extension of the PROCODES program. Between 2007 and 2014, 370 licenses were purchased.
(2) Rent-out: payment for ecosystem services program (PES) provided financial compensation for not fishing in the Vaquita Refuge (**Figure 1**). Around 876 fishing license owners participated in this program for three years, which basically turned the refuge into a No Take Zone (NTZ).
(3) Switch-out: provided opportunities for fishers to permanently exchange gillnets for other non-entangling fishing gears. A total of 370 fishers participated in this program.
(4) Alternative technology development: paid fishers for participating in tests to develop alternative methods of fishing without gillnets. In return, participants agreed not to use gillnets during the year of testing. Thirty-eight fishers participated in 2009 and up to 126 in 2010; after this time, participation was almost non-existent.

While the PACE program was showing some positive results with fishers, the vaquita population decline accelerated (Jaramillo-Legorreta et al., 2016). The worsening situation led the Government of Mexico to suspend the PACE program and implement a new strategy to save the species. The Comprehensive Care Plan was announced in 2015 and differed from the PACE in that the program was mandatory. It had four main components:

(1) An increase in the size of the NTZ from 126,000 to 1.3 million hectares (the Gillnet Exclusion Zone; **Figure 1**).
(2) The suspension of all small-scale fisheries from the Gillnet Exclusion Zone for two years. The new regulations prohibited only gillnets and longlines (sometimes also used to catch totoaba), but in practice the prohibition also included fishing with gears which were originally designed as alternatives to gillnets.
(3) Monetary compensation provided to fishers and related industries for the loss of income from the fishing ban; 109M USD from 2015 to 2018.
(4) Enhanced enforcement effort and coordination with the support of the Mexican Navy and Federal Police.

Despite the unprecedented financial and political investment, the Comprehensive Care Plan also failed to end gillnet fishing in vaquita habitat and the vaquita population continued to decline. Fishers grew increasingly frustrated at the ban on earning a living from fishing with gillnets, the lack of effective enforcement against poaching, the inconsistent and insufficient support to develop alternative methods of fishing, and corruption and disparities in the government compensation system. The Center for Biological Diversity (CBD]) analyzed the distribution of government compensations among the fishing communities and found large disparities (13 individuals received 20% of the compensation; Olivera and Uhlemann, 2016),[5] making a few individuals extremely wealthy while leaving some other families without adequate income support and compromising food security.

Table 1 underscores another important point, notably that payments to compensate fishers for not fishing in the NTZ (122M USD) significantly surpassed the investment in developing alternative fishing methods and alternative livelihoods (23.1M) (**Table 1** and **Appendix 2**). Compensation was an emergency measure to support communities and buy time for vaquita, but the opportunity was missed to link future compensation payments to fishers' participation in efforts that would benefit vaquita conservation or rebuild the region's fisheries, economy, and community well-being, such as gear development and retraining programs (Comité Internacional para la Recuperación de la Vaquita [CIRVA], 2017).

In April 2017, the Government of Mexico declared extraction of endangered species a criminal felony comparable to organized crime (involving at least 3 fishers) (Diario Oficial de la Federación [DOF], 2017a) which was previously a minor offense. For the first time, those caught and convicted of totoaba poaching would be subject to a substantial fine. They would also be subject to a criminal trial and, if convicted, serve prison time (Diario Oficial de la Federación [DOF], 2017a). There has been inspections and seizures (Procuraduría Federal de Protección al Ambiente [PROFEPA], 2018), but few arrests, prosecutions, or penalties for illegal fishing, poaching, or trafficking in Mexico and respect for authority and rule of law is decreasing in the UGC (Expansión Política, 2019; Felbab-Brown, 2020). For example, of 174 formal cases filed by PROFEPA for capture, trafficking, and distribution of totoaba products (Procuraduría Federal de Protección al Ambiente [PROFEPA], 2019), only 10–13% of the arrests received a criminal sanction (Rivera, 2018; Martinez, 2019). There were no convictions between December 2018 and the end of 2019 (Procuraduría Federal de Protección al Ambiente [PROFEPA], 2019) and there has been only one set of arrests in 2020 (El Universal, 2020).

[5]Some of the factors impacting the distribution of compensation funds could include political power (fishing leaders) and economic power (fishing cooperatives linked with buyers or those that have a concentration of licenses).

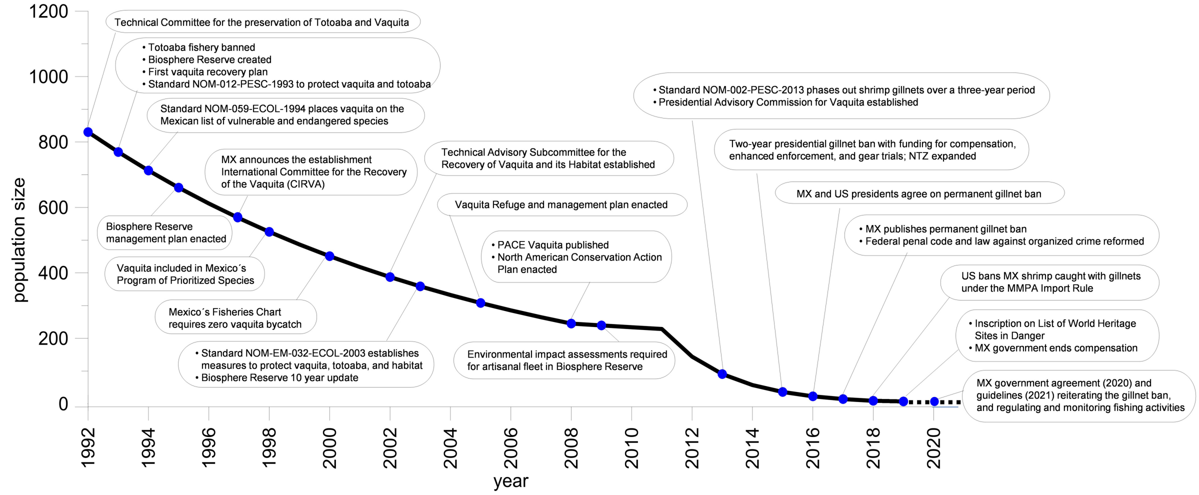

FIGURE 2 | The decline of vaquitas has been continual despite many laws and regulations being enacted to reduce vaquita bycatch, but not resulting in the elimination of gillnets in vaquita habitat. See text and reviews in Rojas-Bracho et al. (2006), Bobadilla et al. (2011), Rojas-Bracho and Reeves (2013), and Cisneros-Montemayor and Vincent (2016) for more information. Figure with schematic depiction of vaquita population trajectory adapted from Comité Internacional para la Recuperación de la Vaquita [CIRVA] (2014), Jaramillo-Legorreta et al. (2019), and Rojas-Bracho et al. (2020).

TABLE 1 | Summary of payments to fishers by the Government of Mexico to implement actions to protect vaquita from gillnets, 2007–2018.

Policy Instrument	Expenditures	Intentions	Outcomes
Buy-outs with alternative livelihoods	12.7M USD	Reduce fishing capacity and build community options.	Increased fishing effort with longer gillnets; limited number of successful new businesses established; participation decreases to zero as program is only voluntary
Compensation for No Take Zones	122M USD	Remove fishers from vaquita habitat and maintain fisher income.	Illegal fishing with gillnets in the No Take Zone increases due to the increase in profitability and lack of sanctions for totoaba poaching
Alternative gear and preferential markets	10.4M USD	Provide alternative ways of fishing without harming vaquita.	No replacement for gillnets, market opportunity lost as other products replace UGC shrimp, market access not able to incentivize conservation actions; no eco-label exists to address information externality; no data available to assess net profitability of gillnet-free fisheries
Social participation in decision making	–	Promote community engagement in decision making	Suspended in 2015; no formal venue for regular engagement among regional stakeholders exists

*See **Appendix 2** for annual breakdown of expenditures and additional explanation.*

With vaquita continuing to die in gillnets, and under mounting pressure to save the species, the Government of Mexico published an agreement prohibiting the use of gillnets in June 2017 (previous versions had been temporary; Diario Oficial de la Federación [DOF], 2015, 2017b). In October, 2017, the Mexican Ministry of the Environment (SEMARNAT) led a team of international partners in an emergency field effort to rescue as many vaquitas as possible and temporarily place them in captivity with the goal of releasing them once gillnets were eliminated, but the effort was suspended when an animals died (Rojas-Bracho et al., 2019). Additional efforts to protect the remaining vaquitas include a program led by SEMARNAT, together with conservation organizations, the Mexican Navy, and local fishers to remove active and abandoned "ghost" gillnets in vaquita habitat. Between October 2016 and March 2020, approximately 1600 gillnets have been retrieved from vaquita habitat (Sea Shepherd Conservation Society, 2019; Comité Internacional para la Recuperación de la Vaquita [CIRVA], 2019; International Union for the Conservation of Nature [IUCN], 2020a, unpublished data).

In December 2018, a new administration for the Government of Mexico was inaugurated and the compensation to fishers and related industries was discontinued (International Union for the Conservation of Nature [IUCN], 2020b). In September 2020, the administration published a new agreement (Diario Oficial de la Federación [DOF], 2020) and guidelines (Diario Oficial de la Federación [DOF], 2021) for the protection of the vaquita. The new agreement included: banning the use of gillnets and requiring that fishers surrender their nets within 60 days; establishing mandatory inspections on every fishing trip; and creating specific zoning rules for the ZTA, encompassing the area with the remaining vaquita sightings. To date, these policies have not been implemented and the testing and financing of new gear outlined in the agreement has not been enacted. This has led to the continuation of unauthorized fishing with gillnets for shimp and finfish, as well as illegal fishing for totoaba (International Union for the Conservation of Nature [IUCN], 2020b; Sea Shepherd Conservation Society, 2020). For the past several years, social unrest and protests, as well as law suits and appeals for compensation, have been increasing as fishers feel their way of life is under attack and they have no alternatives. Reports of violence by illegal fishers against net removal vessels, legal fishers, and Mexican authorities are also becoming more frequent (Comité Internacional para la Recuperación de la Vaquita [CIRVA], 2019; Expansión Política, 2019; Felbab-Brown, 2020; International Union for the Conservation of Nature [IUCN], 2020a).

The alarm over the declining status of vaquita and continuation of illegal fishing has triggered international attention. UNESCO initiated review and subsequently inscribed the Islands and Protected Areas of the Gulf of California on the List of World Heritage Sites in Danger (United Nations Educational, Scientific and Cultural Organization [UNESCO], 2019). In 2018, in response to litigation by conservation organizations, the United States banned the importation of fish and fish products caught with gillnets within the range of the vaquita under the new U.S. Marine Mammal Protect Act (MMPA) Import Provisions Rule (Federal Register [FR], 2018). The rule requires nations exporting fish and fish products to the United States to have marine mammal bycatch measures that are comparable in effectiveness to U.S. commercial fishing operations. The ban was followed by a finding of "comparability" that allowed imports of products caught in the same region with alternative vaquita-safe gears. However, the comparability finding has since been revoked because of Mexico's failure to implement a comparable regulatory program and to enforce existing regulations (Federal Register [FR], 2020).

Vaquita is also identified as an issue of concern in the new 2020 United States-Mexico-Canada Agreement (USMCA) on trade (United States Trade Representative [USTR], 2020). In addition, and in response to the lack of effective action to counter totoaba trafficking, CITES drafted a resolution to restrict Mexico's exports to international markets for more than 2000 listed species, representing an important source of foreign revenue into the country (Convention on International Trade in Endangered Species of Wild Fauna and Flora [CITES], 2019). While the deadline has been extended, the possibility of sanctions remains open, imposing a threat to important Mexican industries and livelihoods.

REVIEW OF POLICY INSTRUMENTS

The members of the NAAFE panel reviewed several of the key policy instruments implemented in the UGC to protect vaquita through an economic lens and evaluated the intended and unintended consequences of these policies. The following section provides a summary and further discussion of the main findings on four key instruments: buy-outs with alternative livelihoods, compensation for not fishing in the NTZ, alternative gear and markets, and social participation in decision making.

Buy-Outs With Alternative Livelihoods

The buy-out programs in the UGC were designed to provide financial incentives to fishers to turn in gillnets in exchange for funds to invest in alternative livelihoods. During PROCODES (2007), authorities offered limited investment options to fishers who opted to take part, while during the PACE (2008–2015) authorities provided a greater range of opportunities for investment by accepting proposals presented by the fishers. The buy-out programs were voluntary and required that participants permanently cancel their fishing licenses. Below, we discuss the effectiveness of the buy-out in building alternative livelihoods and as a measure to reduce fishing effort.

Ávila-Forcada et al. (2012) analyzed participation in the PACE program. They found that individual fishers' social, economic, and demographic characteristics influenced decisions on whether to participate in this voluntary program and determined the level of participation as the program progressed. For example, fishers with a single license preferred to avoid risk by retaining their license and continuing to fish, while fishers with multiple licenses were more likely to participate. In addition, fishers with skills in other economic activities were more likely to participate, as well as older fishers who took the opportunity to retire from fishing.

Effectiveness of Building Alternative Livelihoods

A preliminary survey conducted in 2011 revealed that after three to four years, 70% of new businesses financed with PACE had survived (Ávila-Forcada et al., 2020). The survey data were used to analyze the factors associated with the survival of these new ventures and found that the businesses more likely to survive were those operated by women, located in San Felipe, not involved in fishing or tourism, and co-financed with loans from other sources[6]. These results highlight the importance of focusing on women in fishing families and the key role of financial services in the transition to alternative livelihoods. They also highlight the importance of looking at the household as a relevant economic unit.

Despite this initial success, from 2010 to the end of PACE in 2015, participation of fishers in the buy-out program was close to zero. The decline in participation was associated with the reduction of the payment: buy-out payments decreased from a single payment of 59,701 USD in 2008 to 31,750 USD in 2010, and to 26,718 USD in 2014[7] due to a lower budget for PACE

[6] Ávila-Forcada et al. (2020) classified the type of business in three categories: (1) tourist-oriented (cabins, restaurants, souvenir shops, and similar); (2) non-tourist oriented (beauty salons, tortilla shops, stationery shops, and similar); and (3) fishery-related (freezer plants, aquaculture activities). The type of business was found to be a non-significant variable of the model.

[7] The payment was calculated as the lifetime value of a fishing permit, considering a discount rate of 10%. The goal of the buy-out was to retire as many gillnets as possible within a defined budget. In economic terms, the government offered the terms of the buy-out as a "reverse auction," designed to minimize the amount expended in order to get as much product as possible; inputs are a target or a budget, with the price offered increasing each iteration. Reverse auctions are designed such that the buyer reveals the price to pay and the seller decides to sell or not, knowing that the price will go up but not knowing anything about the remaining budget. In the logic of a reverse auction, the amounts that the government of Mexico should present would increase year by year. However, the buy-out operated in the opposite direction with time than what would be recommended by auction theory (Barlow et al., 2010).

(see **Appendix 2**). In addition to the reduction in the payment, the effectiveness of the program was limited due to the lack of malleability of human capital which is a barrier to exiting the fishing sector as described initially by Clark et al. (1979) and elaborated on by Clark and Munro (2017). Often, fishers are not ready or able to switch to other employment or to become small business entrepreneurs, and the UGC is no exception; those with malleable skills were those who were more apt to succeed in the program while most needed additional training to acquire competence in a different sector (Ávila-Forcada et al., 2012). There were attempts to train fishers in different trades, and an office established for a brief time to assist fishers (e.g., McGuire and Valdez-Gardea, 2008), but not enough support was devoted to this critical component in order for it to be successful. Ultimately, UGC fishers were not willing to give up the profits that could be made in the gillnet fisheries for which they were familiar and skilled. The profitability of these fisheries, along with little enforcement, and the money available from government compensation programs, provided strong incentives to remain fishing with gillnets.

Other obstacles to exiting the fishery included the lack of retirement plans for older fishers; low access to scholarships and training programs for young people prior to joining the fishing sector; and land tenure problems, especially in El Golfo de Santa Clara. Most of the fishers in this town have homes built on land they do not legally own (Bobadilla et al., 2011). If they leave the region, they do not have the certainty of keeping their home upon returning or the ability to profit from its sale. Lastly, the program coincided with the global economic recession, particularly that of the United States, which had adverse effects on the Mexican economy, including tourism (Villarreal, 2010).

The panel noted that for economic alternatives to be successful, it is critical to ensure well-functioning financial, property, and regulatory institutions that enable small business development. There are costs to switching professions, such as training, interest on loans, and opportunity costs for lost time before a new business becomes profitable, which may dissuade fishers from participating. The panel also noted that alternative livelihood efforts in the region only rarely considered the skills and needs of women or opportunities to strengthen the next generation by investing in education, food security, or health care, thereby limiting the ability of communities to evolve. There was agreement on the importance of establishing a diversified economy to avoid the dependency on one single sector and the need to allow local people to decide on the selection of alternatives.

Effectiveness of Buy-Outs for Reducing Bycatch

A buy-out program can perversely result in an increase in fishing effort. Fishers who take part in a buy-out could reinvest their funds to expand their fishing capacity on other vessels, such as by increasing the power of the engine or the size of the fishing gear (Clark et al., 2005; Curtis and Squires, 2007). This concept, known as capital stuffing, is defined as the tendency to invest in non-restricted inputs (such as hull, engine, gear) when one input (vessel number or size) is limited, often in response to regulations to limit entry or reduce fishing effort (Pope, 2009).

Prior to the buy-out program, Cudney-Bueno and Turk-Boyer (1998) reported that a typical shrimp operation used two gillnets of $\sim$ 400 meters each; after the buy-out, Pérez-Valencia et al. (2015) documented the use of two gillnets of 800 meters each. Cisneros-Montemayor (2017) used a static bio-economic model to analyze the economic rationale for increasing the size of gillnets. The starting point of the model is open access in which each panga (skiff) used 800 meters of gillnets. When introducing a buy-out into the model, the model shows an initial reduction of the number of pangas but an increase in the size of gillnet to 1,600 meters to return to the original equilibrium. This research, other studies of buy-out programs (Curtis and Squires, 2007), and anecdotal information from the UGC all suggest an increase in fishing capacity per unit of legal effort was a consequence of the buy-out (It is important to note the legal length of a gillnet in the UGC was 200 meters). Fishing capacity also reportedly increased due to an increase in the number of illegal new and "cloned" pangas[8]; an increase in engine capacity (new, bigger, dual motors); and an increase in the size of gillnets, or some combination of these Comité Internacional para la Recuperación de la Vaquita [CIRVA], 2014).

Considering the limited and then declining participation in PACE and PROCODES, the limitations on alternative livelihoods posed by non-malleable human capital, and the evidence that gillnet effort in vaquita habitat did not decline, the panel raised significant concerns about the buy-out programs for the UGC and the unintended consequence of an increase in gillnet length. The panel emphasized the importance of establishing a sound fisheries management system encompassing permits, capacity, effort, access rights, and gears, as a first condition for supporting bycatch policies, which then must be enforced.

Compensation for Not Fishing in the No Take Zone

The Vaquita Refuge, an area of 126,000 hectares, was created in 2005 and established as a NTZ in 2008. From 2008 to 2014, UGC fishers received compensation for not fishing in the NTZ, also known as a PES policy to protect biodiversity. Annual payments varied from 3,000 to 4,500 USD per panga (regardless of the number of licenses). As a point of comparison, net annual income per panga with a shrimp license in 2010 was estimated at 2200–2700 USD and for finfish 857–1935 USD in San Felipe and El Golfo de Santa Clara, respectively (Barlow et al., 2010). The PES program excluded any mechanisms to incentivize license holders to share the payment with their crew members. Information from a vessel tracking program during the period shows that fishing intensity inside the NTZ was lower than in the rest of the UGC (Erisman et al., 2015), reflecting some compliance and a reduction of fishing activity inside the NTZ, which encompassed half of the vaquita population in a relatively

[8]The term "cloning" here refers to the use of a single license to operate two or more pangas. Cloned pangas have the same type of engine and the same boat name and license number printed on the side of the vessel. They carry duplicate licenses on board and share any other components that would make it difficult to differentiate the original panga from the "cloned" one/s. Usually, cloned pangas are deployed at a distance from one another in order to avoid detection by the authorities.

small area[9]. However, fishing effort was not reduced in the other half of the range of vaquitas, resulting in a higher density of nets outside the NTZ (Comité Internacional para la Recuperación de la Vaquita [CIRVA], 2014). In 2015, when vaquita numbers dropped to less than 100 individuals, the Mexican government enacted emergency measures to expand the NTZ – 10 times larger than the original area. The increase encompassed the full range of vaquita and the known areas of illegal gillnet fishing for totoaba.

Payments for not fishing in the NTZ also increased in 2015: pangas with three licenses received up to 2,000 USD for each month of the fishing season. Despite adoption of the emergency gillnet ban in 2015, and significant financial compensation coupled with a new scheme to provide a subsidy to workers in related industries, reports of illegal fishing activities for totoaba increased dramatically. The rate of decline of vaquita abundance was the highest recorded to this time, 49% a year (2015–2016; Thomas et al., 2017). CIRVA had recommended that instead of lump-sum payments, compensation should be given only if fishers invested in vaquita friendly gear or participated in gillnet-free fisheries (Comité Internacional para la Recuperación de la Vaquita [CIRVA], 2017). However, compensation was made on a regular basis with no requirements to participate in training or experimental gillnet-free fishing in exchange for the payments.

The panel noted the critical importance of considering the consequences of suspending legal fisheries for shrimp and finfish. While suspending legal fisheries reduced the risk to vaquita from gillnets used to target shrimp and finfish, the benefit was likely offset by increased incentive to target totoaba. The panel again highlighted the importance of a sound fishery management system, that is enforced, as a starting point in the establishment of a NTZ, which also needs to be developed in collaboration with the fishery sector. In other marine protected areas in the Gulf of California, fishing spillover, new opportunities for tourism, sportfishing, and commercial fishing, and continuous negotiations have compensated for the closure (Bobadilla Jiménez et al., 2017; Cisneros-Montemayor et al., 2020).

Alternative Gear and Markets

Beginning in 2004, experiments were conducted by the Government of Mexico, conservation organizations, academics, gear experts, and the fishing sector to develop and test promising gears that would allow fishers to make a living without risking vaquita entanglement (Herrera et al., 2017). A small number of fishers in the region have been key participants in these experiments, interested and willing to develop, test, and use alternative gears (PescaABC, 2017).

These early efforts identified a number of promising gear types with commercially viable catch ranges (catch rates were dependent on fisher skill; Herrera et al., 2017). However, the Mexican authorities failed to continue to make progress in transitioning UGC fisheries away from gillnets (United Nations Educational, Scientific and Cultural Organization [UNESCO], 2018). Efforts have continued to fall short, including in the development of new gears, providing the necessary permits, and securing safe places for fishermen to conduct trials on the water. Consequently, most UGC fishers remain reluctant to use alternative gear, claiming it produces less catch and preferring the gillnets that have been profitable to use for decades. For example, in 2013 an important regulation outlined a three-year phase out of shrimp gillnets at the UGC as part of a modification to the national standard for shrimp fishing NOM-002 (Diario Oficial de la Federación [DOF], 2013). The application of this standard was delayed because of the expansion of the NTZ in 2015, lack of enforcement, and the reluctance of the fishing community to accept the technological change. Those fishers willing to use new gears at the time were frequently not able to get the required experimental fishing permits from the fisheries authorities. They also lacked financial support and have been physically blocked from access to fishing areas by gillnet fishers. Researchers have not been able to complete critical cost-earnings analyses with sufficient data to estimate profitability, improve fishing methods, and develop markets.

As PACE ended in 2015, the efforts for testing and implementing alternative gear have been inconsistent and continue to fall far short of the effort and investment needed (United Nations Educational, Scientific and Cultural Organization [UNESCO], 2018; Comité Internacional para la Recuperación de la Vaquita [CIRVA], 2019). Some of the efforts since 2015 include the testing of suripera nets for shrimp and encircling techniques for corvina conducted by INAPESCA; tests with fish traps conducted by fishers in San Felipe from Pesca Alternativa de Baja California (PescaABC); trials of suripera nets and a traceability system for shrimp organized by Museo de la Ballena with local fishers, which were successful in harvesting commercially viable catches; and some trials with small trawls conducted by individual fishers who obtained a commercial permit for catching shrimp. All these efforts have been done without a systematic plan by the fisheries authorities, while most of the fleet is allowed to fish illegally with gillnets, often observed in the NTZ.

Gear experts agree that over time, the alternative gear is likely to become more efficient as fishers become familiar with their use and gear configurations are tailored for local conditions (Herrera et al., 2017). To complement these efforts, southern California seafood buyers and chefs have demonstrated interest in purchasing shrimp harvested with methods that do not endanger vaquita (Aquarium of the Pacific, 2016; Ocean Awards, 2016; Sustainable Fisheries Partnership, 2019). There has also been interest in linking committed producers to responsible consumers, applying a "vaquita friendly" ecolabel as an instrument to support fishers using vaquita-safe gears. Eco-labels and preferential markets have the potential to empower and reward fishers, such as in the form of a price premium or market access, for responsible fishing practices (Poindexter et al., 2017). Such an approach is particularly well-suited

[9] Rojas-Bracho and Reeves (2013) explain the technical difficulties that fishers experienced in identifying the NTZ that made compliance difficult to follow.

for the high-quality, large-sized, blue shrimp from the UGC (Mesnick et al., 2019).

The panel noted the importance of understanding the large amount of human and social capital invested in gillnet fisheries for the last half century, making change inherently difficult. They also noted the importance of legal markets that support fishers' ability to make a good living, as important insurance against the temptations of illegal markets (Felbab-Brown, 2017). The panel agreed on several points with respect to alternative gear and market opportunities. These included the critical need for fisheries authorities to facilitate the permitting, testing, and use of alternative gears and to take advantage of the expert technical advice provided by fishers themselves, as well as numerous gear technicians from government and academic research institutions. Verifiable and transparent seafood traceability systems are also indispensable requirements for conservation and market access, and fishers can be linked directly with buyers to address issues of asymmetric information. Finally, fishery management with stringent and enforced capacity and catch limits is key in ensuring that incentivizing policies will have the intended results.

Vaquita conservation in Mexico has been influenced, to some extent, by policies in other export market countries and multilateral fora. The ban under the U.S. MMPA import regulations, and the focus on vaquita and totoaba in the new USMCA trade agreement, has garnered attention and some action from the Government of Mexico, but has not stopped the use of gillnets in vaquita habitat. Fishers who continue to target shrimp with illegal gillnets may their product domestically, or it may be laundered and sold in the U.S. market (Felbab-Brown, 2020; Mendez, 2021; pers. comm. with Vaquita Enforcement Study Group[10]). Legal fishers using alternative gears are not able to access to the lucrative U.S. market, and opportunities to support these fishers through potential price premiums and niche market access in the U.S. are not possible at this time, but can be pursued in domestic markets. The ongoing poaching and trafficking of totoaba swim bladders also risks far-reaching sanctions under CITES for producers all across the country, but Mexico has not demonstrated that the vaquita and totoaba are effectively protected (Convention on International Trade in Endangered Species of Wild Fauna and Flora [CITES], 2019).

Social Participation in Decision Making

One of the most valuable, but often overlooked, aspects of the policies enacted in the UGC was the creation of the Group of Monitoring and Evaluation (OES) as part of PACE. OES held a total of 22 sessions from 2008 to 2013, averaging one meeting every 10 weeks. CONANP chaired OES which included other federal agencies, local governments, municipalities, conservation organizations, academics, and the fishing sector. Participants evaluated the progress of the policies and discussed adaptive management measures including compensations, changes in regulation, fishing management, alternative gear, and alternative livelihoods. OES could be defined as a formal cooperation-building approach for the government and the fishing sector, different from the numerous but *ad hoc* listening sessions held by the government agencies with the fishing sector, after OES ended. This participatory regulatory process played an important role in building buy-in to comply with regulations. As Symes and Hoefnagel (2010) and Benham (2017) show, when the subjects of regulations participate in the design of the regulations, higher compliance is observed.

As a result of OES, the fishing sector agreed to present an Environmental Impact Assessment (EIA) in which the sector would report its expected fishing effort, recognize its impacts, and propose mitigation measures to reduce the negative impact on the ecosystems[11]. Using this instrument, attempts were made to control fishing effort, and legal fishers agreed to participate in monitoring activities (Pérez-Valencia et al., 2015).

In the arena provided by OES, the fishing sector was also able to present their own ideas for regulations and solutions. For example, a key discussion at OES was the length of gillnets. As noted above, regulations (for shrimp) established a gillnet length of 200 meters, yet fishers regularly used two gillnets of 800 meters in length. At OES meetings, the fishing sector proposed one gillnet of 600 meters in length. From the fishers' perspective this meant a reduction of 62.5%, while government officials perceived this as an increase of 300% from the legal level. The proposal did not pass, but this is an example of how the fishing sector was starting to participate in designing regulations that could balance vaquita protection with their own interests. While this was not the level of reduction called for by conservationists, it was an unprecedented offer presented by fishers themselves.

When switching from PACE to the Comprehensive Care Plan, OES disappeared, removing the only systematic venue for building relationships and developing the non-written norms and tools of engagement to help ensure buy-in with regulations. After OES, a Presidential Commission was created. The Presidential Commission was a closed group that included fisheries representatives but rarely included fishery sector stakeholders. Other agreements between Government of Mexico officials were made in closed sessions with fisheries representatives.

The panel and experts in social participation agree that having a constant framework for engagement allows space for cooperation to emerge (Leslie et al., 2015; Nenadovic and Epstein, 2016). A community fishery monitoring program for corvina in El Golfo de Santa Clara, which employs local women and is built with government support and a group of technical advisors, shows elements of such a framework (Environmental Defense Fund [EDF], 2021).

[10]Personal communication from the Vaquita Enforcement Study Group to the authors

[11]Minutes of the 14th session of the OES, 10 June 2010.

DISCUSSION

Although several policy instruments and unprecedented financial investments were made by the Government of Mexico in the UGC, they have been unsuccessful in eliminating the threat that gillnets pose to vaquita. We contend that a failure to prioritize and support alternative fishing methods and livelihoods for the coastal communities was a key missing component, compounded by the failure to address underlying issues of fisheries management and governance. Where these have failed, black markets, organized crime, and economic uncertainty have proliferated, driving ongoing use of gillnets and risk to vaquita. In this section, we examine how the current structure of incentives in the region makes illegal activities relatively more profitable for fishers, discuss the key factors undermining compliance with regulations, and make recommendations to improve outcomes.

The Economics of Fishers' Compliance

From an economic perspective, the willingness to participate in an illegal activity can be estimated based on three parameters: the expected payout from the illegal activity, the severity of the sanction (penalty) for participating in the illegal activity, and the probability of being sanctioned (caught, prosecuted, and convicted) (Becker, 1968; Freeman, 1999; among others). Sumaila et al. (2006) conducted cost-benefit analyses of global patterns in illegal, unreported, and unregulated (IUU) fishing that quantified these factors, finding that the expected benefits from IUU fishing far exceed the expected cost of being apprehended. Assuming a probability of being sanctioned equal to 20%, the authors found that penalties would need to increase up to 24 times to equal the expected profitability of illegality (Sumaila et al., 2006).

Expected Monetary Benefit From the Illegal Activity

A fisher's choice is based on a comparison between the expected income from illegal fishing and expected income from a legal activity (fishing with alternative gears, fishing outside the exclusion areas, or taking a job that does not involve fishing). For illegal fishing, the expected net income is the net revenues minus the consequences of being caught. The consequences are the product of the probability of getting caught and sanctioned and the severity of the anticipated penalty. Given the relative values in this choice set, especially the high monetary value of totoaba swim bladders and very low likelihood of being caught and sanctioned, incentives remain very strong for UGC fishers to engage in illegal fishing.

Penalty for Participating in the Illegal Activity

As noted above, a federal resolution in 2017 declared illegal fishing a major felony comparable to organized crime (Diario Oficial de la Federación [DOF], 2017a), yet there have been few arrests or convictions. In the case of the UGC, both the penalty and the probability of being caught and prosecuted are low and both components should be evaluated. Setting the appropriate sanctions is complex and issues of fairness, political acceptance, and civil liberties need to be considered (Polinsky and Shavell, 2000; Felbab-Brown, 2017). Evidence from law enforcement demonstrates, however, that it is the probability of being sanctioned that is fundamental to creating deterrence effects (i.e., incentivizing people to comply), even far more so than the size of penalties (Kleiman, 2009).

Probability of Being Sanctioned

When expanding the NTZ, the Government of Mexico also significantly increased investment in surveillance, including personnel, high speed military-style boats, drones, and special cameras. However, there was not a clear strategy for using these technologies in enforcement (United Nations Educational, Scientific and Cultural Organization [UNESCO], 2018), and the efforts have not been well coordinated nor sustained. The jurisdiction among UGC enforcement agencies is complex and rules governing their activities limit effective action. Despite enforcement assets, and the possibility of criminal felony there is a lack of will by authorities to apply them; low and sporadic rates of effective arrest and prosecution result in low deterrence effects and therefore poor compliance in the region (Felbab-Brown, 2020).

Facing low probabilities of being caught and prosecuted, and high expected payoff for illegal activities, illegality becomes more attractive for many fishers in the UGC and can attract poachers from outside the area. Addressing the structure of incentives in the region requires a strategic approach that takes the full set of expected payouts and penalties into consideration so that regulations have the intended consequences.

Factors Undermining Compliance

Additional systemic factors and external pressures further undermine compliance with regulations and the effectiveness of policies to mitigate bycatch in the UGC, including the following.

Weak Fisheries Management

The panel noted a common point that complicated the implementation of the policies in the case of the UGC: the fishery lacks the most basic management measures, including any form of rights-based management. A history of weak fisheries management resulting in open-access, overcapitalized fisheries, keep UGC communities struggling for their livelihoods (Lluch-Cota et al., 2007; Erisman et al., 2011; Cisneros-Montemayor and Vincent, 2016; Pasini et al., 2017; Mangin et al., 2018; Aceves-Bueno et al., 2020). Moving toward rights-based fisheries management in the UGC is possible as seen in the case for corvina (Ortiz et al., 2016). Charles (2009) notes that defining fisheries rights aligns fishers' interests with management measures and therefore engenders greater compliance with fishery regulations.

Complex Regulatory Environments Carry High Coordination Costs

The UGC offers a particularly complex regulatory environment: multiple government agencies and multiple fishing organizations are involved in policy regulation and administration, requiring considerable coordination in designing and implementing environmental policies (Cisneros-Mata, 2020). Stronger collaboration amongst the fishing sector and regulators will lower transaction costs and improve information flow.

Illegality and Corruption

There is a longstanding history of corruption and tolerance to illegality that has surrounded fisheries in the UGC. Corruption is a key enabler of the illict totoaba trade (Environmental Investigation Agency [EIA], 2019). It undermines the ability of law enforcement to fight poaching and criminal networks and it reduces the deterrent effects of enforcement. Enforcement requires a strategic approach that embraces the entire compliance and enforcement chain (C4ADS, 2017; United Nations Educational, Scientific and Cultural Organization [UNESCO], 2018; Environmental Investigation Agency [EIA], 2019; Aceves-Bueno et al., 2020; Felbab-Brown, 2020). Enforcement and compliance with regulations in the UGC is also difficult because many fishers are willing participants in poaching and illegal fishing operations, view regulations as illegitimate or an imposition of conservation values that go against their economic interests (Felbab-Brown, 2018; Aceves-Bueno et al., 2020). In these cases, enforcement is both socially and politically unsustainable, as well as costly (Felbab-Brown, 2018). Building economic alternatives with the fishing community can be key to gaining acceptance and compliance, lowering the costs of enforcement, particularly when participants perceive regulations to serve their economic interests.

Lack of Community Buy-In

In addition to the probability and severity of sanctions, decisions to comply with regulations are also influenced by intrinsic motivations, as described by theories of psychology and sociology (Sutinen and Viswanathan, 1999; Squires et al., 2021). Consideration of these motivational inputs as well as social norms can contribute to building compliance. Examples of social norms that can be considered include social influence, moral values, sense of justice, and the perceived legitimacy of regulations (Hatcher et al., 2000). Participatory regulatory processes, in which participants are empowered to play a prominent role in decision making and where their views can be heard, may help achieve better compliance (Hanna, 1995).

However, it is important to note, that the role of individual private entities should be limited in the design of regulations to secure the public interest and avoid regulatory capture, which occurs when a regulatory agency is co-opted by the interests of a minority (Dal Bó, 2006). To address some of the regulatory capture problems, agencies can limit discretion and empower public interest groups, including other sectors and industries. When this can be accomplished, public participation can lead to efficient forms of cooperation that enhance the attainment of regulatory goals and strengthen democracy (Ayres and Braithwaite, 1991).

Pressure From Outside of Mexico: Demand From Distant Markets

The UGC finds itself at the crossroads of several global market forces. The U.S. demand for totoaba meat in the early years, and then shrimp, have been key drivers of vaquita bycatch. The market for totoaba swim bladders in Hong Kong and continental China results in derived demand and the current crisis. Chinese traffickers and Mexican cartels trafficking drugs to the U.S. market place additional outside pressures on the region (C4ADS, 2017; Crosta et al., 2018; Environmental Investigation Agency [EIA], 2019; Felbab-Brown, 2020). These external forces fuel poaching of totoaba and illegal fishing at the local level and empower criminal activity and corruption. The tools of traditional fisheries management are not sufficient to address these drivers (Aceves-Bueno et al., 2020). Tackling the complex supply chains linking fishers in the UGC and end consumers in distant markets requires a portfolio of policy instruments, enforcement tools, industry and government accountability, public information campaigns, and multidisciplinary expertise adapted for the special circumstances of reducing demand and trafficking in prohibited wildlife products (Felbab-Brown, 2017, 2018).

Inequality, Impoverishment, and Crime

Although the UGC has some of the most diverse and profitable fisheries resources in Mexico, the history of weak fisheries management, open access fisheries, and overcapitalization contributes to inequality and impoverishment, as is the case in fisheries around the country (Mangin et al., 2018). The literature on poverty and crime shows a positive correlation between absolute poverty and illegal activities (Patterson, 1991; Scorzafave and Soares, 2009; Short, 2018). In the UGC, the loss of income from the closure of gillnet fisheries and limited alternative sources of income were further compounded by the disparities created by the compensation plan. Some fishers with few alternatives turn to totoaba poaching and illegal fishing, risk becoming further trapped by debt to the cartels, and cannot exit (Ladkani, 2019; Alberts, 2021). As Felbab-Brown (2017), Aceves-Bueno et al. (2020), and others have pointed out, viewing poaching only as an activity of criminals ignores important underlying social injustices. Policies that compromise economic security may end up exacerbating the conditions that incentivized illegal activities in the first place (Cisneros-Montemayor and Vincent, 2016; Felbab-Brown, 2017; Aceves-Bueno et al., 2020).

RECOMMENDATIONS

Despite the very small area and the singular threat to survival, saving vaquita is complex, multi-faceted, and context dependent. It requires a holistic approach, using local knowledge, multidisciplinary expertise, and a broad range of tools and policy instruments. Solutions to eliminating vaquita bycatch require combining top-down regulations and effective enforcement with bottom-up, participatory, and incentive-based approaches to improve buy-in, and therefore bolster compliance with regulations. The holistic approach recognizes that illegal fishing in the UGC is not solely a problem of enforcement; it also reflects the social, economic, and political context of coastal communities. Not surprisingly, fisheries regulations that are seen as conflicting with the livelihoods of local fishers tend to provoke resistance (Gezelius and Hauck, 2011). Conservation needs to be designed and structured to benefit local communities through early and inclusive processes. Conservation actions should be geared to support legal fishermen in their ability to earn a living and to ensure they have a direct stake in a healthy marine ecosystem. These approaches require more time and a step-wise deliberative process, but will yield more long-lasting and effective results in addressing bycatch and wildlife trafficking. The panel recommends the following.

Strengthen Fisheries Management With a Clear Definition of Access Rights

Many of the issues in the region stem from a legacy of poor fisheries management with overcaptalized and mostly open-access fisheries (Sumaila, 2012). Charles (2009) identifies the wide variety of access rights, noting two primary categories: (1) access rights specifying which vessels may participate in the fishery; and (2) harvest rights defined by species, gear, and time. Fishers within the region who are most impacted by restrictions should benefit from the resource through formal recognition of tenure rights (Cisneros-Montemayor and Vincent, 2016) whether in terms of rights to participate or rights to a share of the harvest.

Build Solutions With Communities

Increased participation in regulatory design is linked to increased compliance (Symes and Hoefnagel, 2010; Benham, 2017). Creating a fishery management system that empowers fishers as part of the decision-making process is key to building buy-in and therefore compliance. Such a participatory regulatory process can be built with broad community and institutional engagement so that trust, accountability, and legitimacy increase, and the political costs of enforcement are reduced (Gezelius and Hauck, 2011).

Improve Enforcement

Even with greater community buy-in, effective enforcement will remain key. Sustained enforcement needs to center on three points: intense enforcement in the vaquita's range to prevent the entry of gillnets into vaquita habitat, particularly the ZTA; robust enforcement in distant retail markets to reduce demand; and elimination of the operational layer of smuggling and organized criminal networks (Felbab-Brown, 2018, 2020). To have any hope of saving the last remaining vaquita, the speed and prevalence of prosecution that results in conviction need to radically increase in the UGC. This requires more effective deployment of detection and interdiction assets, full use of detection technologies, and greater diligence of authorities at the local, national, and international levels to enforce regulations. Cross-agency coordination, the prioritization of enforcement of the gillnet ban, and reduction of corruption among regulators and enforcers *as well as* the reduction of the political costs of enforcement, such as through the development of legal markets are also needed. Well-designed enforcement that is seen as legitimate by key stakeholders and is sustained, and sustainable, is essential for the effectiveness of conservation policies.

Create an Optimal Structure of Sanctions

To achieve an optimal level of deterrence, re-thinking the structure of sanctions in the UGC is needed, including consideration of fairness, pathways for offenders to avoid future violations, and the importance of swift, certain, and consistent levels of enforcement. In addition, and in a context of tolerated corruption such as the UGC, it is important to consider the possibility of including rewards to enforcement officers to avoid bribery (Polinsky and Shavell, 2001). Appropriate sanctions (including permit revocations, boat and catch seizures, fines, and prison terms) need to be commensurate with the socioeconomic condition of the offender and their level of involvement in illegal activities and not push low-level offenders farther into debt or illegality. Frequent violators should be fined at escalating rates, and fishing leaders as well as members of organized criminal groups arrested. Felbab-Brown (2017) describes that for law enforcement to have pronounced deterrence effects for homicide, for example, arrests and effective prosecutions need to reach about 40%. Mexican laws and regulations are in place; the will to apply them is needed in order for sanctions to influence the decision to participate in illegal fishing or trafficking (Felbab-Brown, 2017, 2020).

Invest in Economic Opportunities

To build compliance with conservation, and to counter the attraction of illegal activities, legal fisheries must be able to provide jobs, profits, and economic security (Cisneros-Montemayor and Vincent, 2016; Felbab-Brown, 2017; Comité Internacional para la Recuperación de la Vaquita [CIRVA], 2019). Creation of alternative livelihood opportunities remains an urgent imperative in the UGC. Collaboration between fishers and fishing authorities, non-governmental organizations, gear experts, and industry must substantially increase to scale, develop, and socialize a comprehensive and transparent gear transition program, with explicit consideration of ecological capacity and impacts, as well as to design and implement a fisheries monitoring and traceability system. A broad range of livelihood alternatives needs to be considered (both on and off the water) that can yield comparable net income levels, and considering factors such as risk and preferences. It is equally important to consider the household as the relevant economic unit and evaluate the full spectrum of small businesses that provide services to the community, such as stationary shops, restaurants, beauty salons, etc. A broader array of economic activities can lead to more resilient coastal communities and increase compliance with policies that serve local economic interests (Cisneros-Montemayor and Vincent, 2016; Ávila-Forcada et al., 2020).

Invest in Human Capital

Crucial to sustainable development in the UGC is long-term investment in human capacity, gender equality, and other factors that enable individuals to create their own livelihood opportunities not exclusively dependent on fishing (Cisneros-Montemayor and Vincent, 2016). Becoming a skilled fisher requires years of investment in human capital, which is lost when a fisher is asked to switch professions. Building capacities among fishers to learn other business skills, creating educational opportunities for young fishers or retirement for older ones, and solving land tenure problems for those willing to migrate, are just some of the alternatives for diversifying livelihood options in the community. Although controlling the overall fishing effort of the fishing fleet through some form of limits on access or output is an important part of improving fishery management, it can result in a smaller fleet and fishing labor force. Thus, investing in the malleability of human capital is essential. A broader perspective is needed to embrace the valuable contributions of women and to build short- and long-term opportunities for women and children, such as micro loans, education, and training.

Use Market Tools, but With Caution

Markets are powerful tools as the ultimate drivers of derived demand for fishing. Buyers, distributors, processors, and producers share responsibility for the legal provenance of their products and need to develop and implement a comprehensive chain of custody and traceability system that can be shown to hold all parts of the supply chain accountable for their sourcing. Bycatch conservation can be addressed by standards and certification and in consumer markets through ecolabels and information programs. However, the regulatory and traceability frameworks to prove compliance must be agreed upon beforehand, implemented, and verified. This also requires clear access rights and catch limits to avoid perversely incentivizing over-fishing which would act against conservation efforts. Domestic markets can be tested for small groups of fishers motivated to try methods that do not endanger vaquita, carry trackers, and linked to consumers seeking responsibly harvested seafood. Market tools can also be used to reduce demand for totoaba swim bladders as part of combatting illegal fishing for totoaba. Linked through organized crime throughout the market chain, the situation requires a comprehensive approach with partners who understand illicit trade and economies (Felbab-Brown, 2017; Aceves-Bueno et al., 2020). An interesting parallel can be found in the global effort to combat shark finning. Public statements by key Chinese government officials and celebrities helped reduce the demand for shark fin soup in the years 2007 to 2013 as did, crucially, the publicization of high content of dangerous mercury concentration in shark fins (Vallianos et al., 2018).

CONCLUDING REMARKS

Every day the people of the UGC communities engage in economic activities to earn a living. What the economic perspective brings is a focus on developing a new structure of incentives in which legal activities benefit communities more than illegal activities. The urgency of the situation requires immediate actions to protect the remaining vaquita. The complexity of the situation requires an integrated, multi-faceted approach to policies, combining technical expertise in gear and economic development, social participation in the regulatory process, socially relevant and structured incentives, and swift and certain enforcement of regulations. The situation for vaquita is dire, yet lessons from this case study may apply in other coastal, small-scale fisheries where conservation policies to reduce bycatch may be seen as conflicting with local economies, and more broadly in conservation issues around the globe.

AUTHOR CONTRIBUTIONS

OP, SM, ES-R, DS, and RL hosted the NAAFE workshop, "Saving the world's most endangered marine mammal: role of economic incentives for affected communities" in 2017. SÁ-F, AC-M, GM, RO-R, RR, JFS, and URS participated in the workshop and contributed to the ideas summarized here. OP prepared the initial meeting report which served as inspiration for the manuscript. ES-R, SM, RL, and SÁ-F conceptualized and wrote the manuscript. All authors made direct intellectual contributions to the work, refined key concepts, and approved the submitted version for publication.

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Appendix 1 | Number of pangas and licenses in the Upper Gulf of California. Adapted from Pérez-Valencia et al. (2015).

Town	Number of pangas	Number of fishing licenses				
		Gillnet fisheries			Non-gillnet fisheries	
		Finfish	Shrimp	Shark	Crab	Others [a]
San Felipe	304	242	220	32	36	20
El Golfo de Santa Clara	451	415	423	33	32	11
Puerto Peñasco	121	37	8	16	96	67
TOTAL	876	694	651	81	164	98

Other species include octopus, clams, scallops, and other shellfish.

Appendix 2 | Payments to fishers by the Government of Mexico to implement actions to protect vaquita from gillnets (1,000's USD), 2007–2018.

Year	Gear Substitution	Buy-out	Compensation No Take Zone	Technical Gear Development	TOTAL
2007	380	2,784	0	0	3,163
2008	2,766	9,008	2,336	0	14,110
2009	1,731	536	1,480	509	4,257
2010	1,179	23	1,841	1,941	4,984
2011	0	0	2,385	0	2,385
2012	46	0	1,979	42	2,067
2013	47	0	2,024	42	2,113
2014	761	344	796	928	2,829
2015	0	0	30,210	0	30,210
2016	0	0	27,639	0	27,639
2017	0	0	28,243	0	28,243
2018	0	0	23,066	0	23,066
TOTAL	6,910	12,695	121,999	3,462	**145,066**

All figures are in 1000's USD based on the average Mexican peso to US dollar exchange rate for each year. Figures correspond exclusively to monetary transfers to fishers from the Government of Mexico (mainly administered by the Ministry of the Environment, SEMARNAT). Figures do not include financial costs of enforcement, which increased significantly during the years of the Comprehensive Care Plan (beginning in 2015). Summary of monetary investment by the Mexican government in the four main components of strategies to remove gillnets from vaquita habitat: gear substitution, buy-outs, compensation for not fishing in the No Take Zone, and investment in technological development of alternative fishing gear. Monetary payments and compensation to fishers effectively ended in 2018. Adapted from Comisión Nacional de Áreas Naturales Protegidas [CONANP], 2009; 2010; 2011; 2012; 2013; 2014; 2019.

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Spatial Management to Reduce Entanglement Risk to North Atlantic Right Whales in Fishing Gear: A Case Study of U.S. Northeast Lobster Fishery 2002–2009

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Despite the use of gear requirements and access restrictions to manage lobster fishery interactions with north Atlantic right whales since 1997, the population is likely below 370 animals. The Dynamic Area Management (DAM) program (2002–2009) used "real-time" right whale sightings data to provide temporary protection using closures or whale-modified-gear to reduce entanglement. Our ex-post evaluation uses a flexible framework to identify strengths and weaknesses of the program. Biological and economic implications of the program are evaluated using a relative risk of entanglement index (RREI) calculated with spatially and temporally explicit data on density of right whales and fishing effort. An illustrative closure optimization model demonstrates the trade-offs between the non-monetary benefits of risk reduction and the opportunity cost of closures under alternative decision rules (benefit-ranking and cost-effectiveness). Annual aerial sampling to detect DAM areas was low (<3%), yet in some months' the 17% of area covered by all northeast right whale management areas encompassed up to 70% of the region's population. Despite their small spatial footprint, dynamic and static measures may have reduced total risk by 6.5% on average, and DAM zones may have created an indirect economic incentive for some fishers to adopt the whale-modified-gear. Similar RREI index values in some months with inverse levels of fishing effort and whale presence highlight the need to consider fishing and whales jointly to reduce risk. These temporal-spatial patterns are critical in policy instrument design. Further, optimization results illustrate how different decision rules can attain equivalent non-monetary benefits of risk reduction at different opportunity costs to industry; the implications of whale-modified-gear and compliance factors are explored. We recommend that DAMs be considered as part of a suite of policy instruments, and highlight how recent technological advances may support lower cost data collection and faster implementation given limited public sector budgets. This case study highlights the need for evaluation of past policy instruments with a lens beyond biological outcomes, and sets the stage for further empirical analysis to better understand harvester responses to management measures designed to protect right whales and the resulting private and public sector trade-offs.

Keywords: mitigation, Dynamic Area Management, bio-eoconomic tradeoffs, policy instruments, bycatch, marine mammal, cost-effectiveness (economics), compliance

INTRODUCTION

The complex interplay between commercial fisheries and marine protected species may require multiple policy instruments to support conservation, as in the case with the north Atlantic right whale (*Eubalena glacialis*; right whale) and the American lobster (*Homarus americanus*) fishery in the northeast United States (U.S.). Bycatch is the single greatest cause of cetacean mortality (Read et al., 2006). Over the past 20 years a growing regime of policy instruments have been applied to the American lobster fishery to reduce the entanglement of right whales in fishing gear in order to allow the endangered population to recover; however, recovery has not occurred (National Oceanic and Atmospheric Administration (NOAA), 2017). While past regulatory measures have been assessed prior to implementation (ex-ante), there has been relatively few ex-post evaluations of the effectiveness or success of past measures for marine mammals and fishing interactions as there is rarely a control or counterfactual to determine what would have occurred in the absence of bycatch management (Little et al., 2015). Such studies can provide insights and guidance in the modification of existing measures and development of new measures. We use a case study approach to develop a framework to jointly consider the conservation and economic considerations of management focused on risk reduction for a large migratory protected species. The case study evaluates the Dynamic Area Management (DAM) program for right whale protection between 2002 and 2009, adding to the literature on policy instruments used to protect marine protected species from interactions with commercial fishing activity (see Fonner et al., 2020 for a recent review).

The right whale population is endangered. The population was as high as 500 in 2010 but is declining (Pace et al., 2017); the most recent abundance estimate is 428 animals (National Oceanic and Atmospheric Administration (NOAA), 2020), with the 2019 population likely less than 370 (Pettis et al., 2020). The leading known causes of mortality for right whales are entanglement in fishing gear and vessel strikes (National Oceanic and Atmospheric Administration (NOAA), 2020). The distribution between the two causes of mortality is unknown, however, recent information indicates the vast majority of serious injuries are entanglement-related and this mortality is widely underestimated (Pace et al., 2021). Further, based on the distribution of vertical lines by gear type, roughly 96% of the vertical lines in U.S. east coast waters were attached to lobster trap/pots for any month on average (National Oceanic and Atmospheric Administration (NOAA), 2014: Exhibit 5–4), suggesting lobster gear likely poses the greatest probability of a whale entanglement. Implementation of measures to reduce the likelihood and severity of entanglements for right whales and other large whales began in 1997, with the first Atlantic Large Whale Take Reduction Plan (the Plan), while implementation of measures to reduce mortality from ship strikes began in 2008 (73 FR 60173, October 10, 2008).[1] Ex-post evaluations of the measures for vessel speed and traffic relocation, which are not part of the Plan, have considered biological outcomes (Laist et al., 2014) and compliance (Lagueux et al., 2011; Silber et al., 2014). Ex-post evaluation of the Plan's success has focused on biological outcomes for the combined measures in the Plan (Knowlton et al., 2012; Pace et al., 2014) and on the Take Reduction Team process (Borggaard et al., 2017). However, the lack of data on entanglement or bycatch rates and the impacts of fishing practices, gear characteristics, area, and/or environmental factors on those rates (Borggaard et al., 2017) make the ex-post evaluation of biological outcomes of a specific management measure, such as DAM, difficult. Evaluation of biological outcomes is further complicated by the progression and overlap of management measures over time. However, policy instruments may be designed to achieve multiple goals or objectives, and the use of other evaluation criteria including, among others, economic, social-normative and longevity (Bisack and Magnusson, 2016) may be appropriate for evaluation of individual measures, such as the DAM program.

All the policy instruments in the Plan were based on a command-and-control, direct regulatory approach, such as input controls (time-area closures) to reduce fishing effort and technical standards (gear modifications) to reduce entanglement rates (Squires and Garcia, 2014; Lent and Squires, 2017). The instruments within the Plan progressed from seasonal closures of critical habitat to static and dynamic gear-modification zones, then to broad-based mandatory gear modifications. This sequence follows the first two stages of the mitigation hierarchy (Squires and Garcia, 2014; Milner-Gulland et al., 2018) of avoidance (closures) and minimization (gear modifications).[2] The mitigation hierarchy approach suggests a sequence of steps (avoid, minimize, restore, compensate) that minimizes ecological risk across all steps (Squires and Garcia, 2014), which may be implemented simultaneously and will interact in the achievement of the biological goal or target (Milner-Gulland et al., 2018).

[1]The Atlantic Large Whale Take Reduction Plan was first published in 1997 with amendments to the rule in 2000, 2002, 2007, 2014, and 2015. See: https://www.fisheries.noaa.gov/new-england-mid-atlantic/marine-mammal-protection/atlantic-large-whale-take-reduction-plan.

[2]The development of "dis-entanglement teams" under the Atlantic Large Whale Take Reduction Plan would be part of the third stage of the hierarchy (remediation), while offsetting seems difficult to imagine for large whales, such as the right whale.

If each step in the hierarchy is implemented to the maximum extent practicable before moving to the next step, diminishing conservation returns per monetary unit spent can result with suboptimal conservation; this can be reduced using a least-cost approach within and between steps (Squires and Garcia, 2018; Squires et al., 2018). Alternatively, incentive-based policy instruments that incorporate the social cost of entanglement in fishing may outperform other instruments and provide an alternative approach to command-and-control.

The team tasked with developing the Plan must deal with the complexity of designing policy instruments that implicitly minimize economic impacts to the target fishery (e.g., cost to lobster fishery) while explicitly minimizing the entanglement of non-target species (i.e., benefit to right whales). Cost-benefit analysis identifies options that maximize net benefits and yield an efficient solution; however, cost-benefit analysis requires monetization of benefits and costs which may not be possible. The formal joint consideration of both biological or non-monetary benefits and economic costs has grown in conservation planning (Ando et al., 1998; Naidoo et al., 2006; Robin et al., 2006), including in the marine environment (e.g., Stewart and Possingham, 2005; Delavenne et al., 2012; Oinonen et al., 2016). For protected species, ex-ante regulatory economic analysis often takes the form of cost-effectiveness analysis,[3] with a comparison of non-monetary biological benefits (i.e., minimize risk of entanglement) with costs (i.e., profit reductions of the fishery) for a discrete number of alternatives. Joint consideration of costs and non-monetary benefits within a optimization framework can yield different outcomes (i.e., compared to when costs not considered) where the magnitude of the efficiency gains depends on the correlations between, and relative variabilities of, benefits and costs across investment or management measures (Ferraro, 2003). One of the challenges of this type of analysis is a quantitative measure of non-monetized benefits of the policy instrument (i.e., management measures).

The focus of the Plan on measures to reduce the likelihood and severity of entanglement by large whales in commercial fixed-gear fisheries suggests it is appropriate to discuss benefits from the Plan in the context of "risk reduction." The common concept of risk includes what can happen, the likelihood of that happening, and the consequence if it happens (Kaplan and Garrick, 1981). Several measures of entanglement risk have focused on the likelihood aspect defined as a co-occurrence between the gear of concern and whales. Wiley et al. (2003) developed an index of Relative Interaction Potential between large whales and commercial fisheries gear in Stellwagen Bank National Marine Sanctuary, United States. To evaluate alternatives for the reopening of the winter black sea bass fishery in the southeastern United States, Farmer et al. (2016) developed Relative Risk Units which considered right whale survey encounters (a proxy for density) and fishing effort measured in soak time. Brilliant et al. (2017) used the probability of encounter of right whales for the Atlantic waters of Canada, which multiplied the annual probability of occurrence of a fishing set within a grid cell by the probability of a right whale within the cell. Vanderlaan et al. (2011) considered the third component of risk by including a measure of lethality to individual whales. Depth was used as an estimate of end-buoy line length, with an encounter based on the relative probability of whale presence and gear presence for the Fundy-Scotia area, Canada.[4]

The DAM program was an innovative "real-time" instrument when introduced, an early example of dynamic ocean management which uses near real-time data to guide the spatial distribution of commercial activities (Lewison et al., 2015). Actual examples of dynamic management remain relatively novel, despite analyses that suggest dynamic area management for fisheries (e.g., Hobday and Hartmann, 2006; Hobday et al., 2010; Needle and Catarino, 2011; Dunn et al., 2016) and marine protected species (e.g., Grantham et al., 2008; Hazen et al., 2018), and reduce bycatch and protected species interactions (Lewison et al., 2015). Using an optimization approach to achieve defined reductions in bycatch, Grantham et al. (2008) found that bycatch reductions could be achieved with lower costs to fishers with spatially and temporally moveable (dynamic) closures as compared to static temporary or permanent closures. Based on 10 case studies of real-time spatial management approaches, Little et al. (2015) concluded that for dynamic management, greater monitoring and control of individual practices may be required to allow for better management and utilization of target and bycatch quota if individual incentives to avoid bycatch are weak. In a review of nine examples, Lewison et al. (2015) identify regulatory frameworks and incentive structures, stakeholder participation, and user aligned technological applications as key to successful implementation of dynamic ocean management. While Little et al. (2015) called for additional evaluation of dynamic management, they note that in many cases rigorous assessment of performance is made difficult by the lack of a control or counterfactual. Empirical evaluations using quasi-experimental data, such as difference-in-difference, require the identification of a counterfactual (e.g., Ardini and Lee, 2018).

We develop a flexible framework that considers biological and economic factors simultaneously that can be used to assess alternative policy instruments including closures and gear modifications, and use this framework for the evaluation of the DAM program. This is an initial step toward an analysis that can assess a broader range of policy instruments. To set the stage for an evaluation of the DAM program we provide background on the U.S. American lobster fishery and the DAM program as part of the Plan to reduce fishery interactions with right whales. Given the critical role of the NOAA-Fisheries aerial survey in the implementation of the DAM program, we review monthly flights. We then assess the biological and economic implications

[3]Cost-effectiveness analysis compares mutually exclusive alternatives in terms of the ratio of costs to a single quantifiable, but not monetized, effectiveness measure (Boardman et al., 2011).

[4]Risk reduction was also used to evaluate alternatives measures to reduce ship strikes, not part of the Atlantic Large Whale Take Reduction Plan. Examples of risk calculations included the co-occurrence (in time and space) of whales and vessels (Fonnesbeck et al., 2008), the multiplication of predicted whale density by a measure of shipping intensity (Williams and O'Hara, 2010), and the relative probability that a vessel will encounter a whale (in time and space) multiplied by the probability of lethality resulting from an encounter (Vanderlaan et al., 2008; Wiley et al., 2011; Conn and Silber, 2013).

of the DAM program relative to other spatial measures, using spatially and temporally explicit data on whale and fishing effort density to develop a relative risk of entanglement index (RREI) for right whales. Using reductions in risk as the non-monetary benefit measure and fishing revenues as a proxy for opportunity costs, we develop an illustrative closure optimization model. We evaluate two management decision rules (benefits-ranking and cost-effectiveness) subject to the same risk reduction constraint, solving for the optimal number of closures (i.e., avoidance) under each decision rule. Then under the cost-effectiveness rule, we consider minimization measures (i.e., gear modifications) to reduce the risk of entanglement, and the implications of non-compliance with gear requirements. Taking into account data quality issues, we demonstrate the utility of our framework and present general results that highlight the tradeoffs between biological and economic objectives. Finally, we summarize our general observations on DAM for right whales, discuss the strengths and challenges of implementing dynamic management, and offer recommendations for future policy instrument design.

BACKGROUND

The U.S. American Lobster Fishery

The American lobster (*Homarus americanus*) has been one of the top three most valuable individual commercial species landed in the United States since 1997, accounting for an average of 9% of the national value of domestic landings.[5] The fishery occurs from Maine to Cape Hatteras, North Carolina although the vast majority of landings come from waters of Maine (**Figure 1**). Maine's dominance of the fishery has increased from 57% of all landings in 1997 rising to 82% in 2018. During the period 2002–2009, the United States landed an average of 39,856 metric tons of lobster with an average annual dockside-value of $393M (US$2012), with about 79% of landings and 75% of value from Maine.

The fishery is cooperatively managed by the states and National Oceanic and Atmospheric Administration Fisheries (NOAA-Fisheries) under the framework of the Atlantic States Marine Fisheries Commission. States have jurisdiction for implementing measures in state waters within 3 nautical miles of shore, while NOAA-Fisheries implements complementary regulations in offshore federal waters 3–200 nautical miles from shore. The reporting requirements for vessels have changed over time and vary by the type of permit held. Vessels that hold a federal lobster permit as well as any other federal fishery permit must submit vessel trip reports (VTR), which include amongst other information, catch and an average fishing location for each trip. However, if the vessel only holds a federal permit for lobster, VTR are not required. State-level data collection exists, but varies by state and over time.

The lobster fishery predominantly uses pots and traps, with the term "lobster trap" broadly referring to any structure or device, other than a net, that is designed to be placed on the ocean bottom and is capable of catching lobsters. A number of traps may be linked together with "groundline" to create a "trawl," the location of which is identified using one or more buoys attached to the trawl using a vertical line arrangement. Specific gear requirements vary by area, but include requirements for marking and deployment. Management measures include biological requirements (e.g., minimum and maximum size), as well as effort constraints including limited access and either individual trap allocations or caps on the number of traps per permit.

The market for lobster is for human consumption and it is sold live or it is processed and primarily sold frozen (Pereira and Josupeit, 2017). There is a high degree of integration in the processing and trade of lobster between the United States and Canada, with each country being the largest trade partner for the other (Department of Fisheries and Oceans Canada (DFO), 2018). Parts of the Maine lobster fishery first received sustainability certification from the Marine Stewardship Council (MSC) in 2013; however, the most recent certification was suspended in summer 2020 as a result of concerns regarding right whale entanglement. [6]

Measures to Reduce Right Whale-Fisheries Interactions[7]

The range of the north Atlantic right whale, which historically included much of the north Atlantic, is primarily in the northwest Atlantic from northern Florida, United States to Newfoundland and Labrador, Canada (National Oceanic and Atmospheric Administration (NOAA), 2020). The species is protected under the *Endangered Species Act* (Endangered Species Act (ESA), 1973) and the (Marine Mammal Protection Act (MMPA), 1972) of the United States and the *Species at Risk Act* (Species at Risk Act, 2002) of Canada. This species has been extensively studied within the south and central part of this range in U.S. waters and the Scotian Shelf in Canada, and biologically important areas for calving, migrating, feeding and mating have been identified (LaBrecque et al., 2015).

To protect these whales, in 1994 two areas of critical habitat as defined by the ESA were designated in the United States, the Great South Channel (**Figure 2**, solid gray area) and Cape Cod Bay (**Figure 3**, solid gray area) (59 Federal Register 28805, 3 June 1994). This was followed in 1997 by the Atlantic Large Whale Take Reduction Plan (the Plan), which was developed under the authority of the MMPA to address incidental takes or bycatch of large whales, defined as non-intentional, accidental death or injury that occurs during an otherwise lawful activity, such as permitted fishing. While the Plan was developed to implement measures to reduce the likelihood and severity of entanglement in commercial fixed-gear fisheries, including lobster, for north Atlantic right whale, north Atlantic humpback whale (*Megaptera novaeangliae*) and

[5]Calculated based on data from NOAA commercial landings, available here: https://foss.nmfs.noaa.gov.

[6]Marine Stewardship Council. See documents associated with the Gulf of Maine Lobster fishery at: https://fisheries.msc.org/en/fisheries/@@search.

[7]All Federal Register (FR) notices are available at https://www.federalregister.gov/ while links to final Atlantic Large Whale Take Reduction Plan regulatory documents (last accessed January 6, 2020) are here: https://www.fisheries.noaa.gov/action/atlantic-large-whale-take-reduction-plan-regulations-1997-2015.

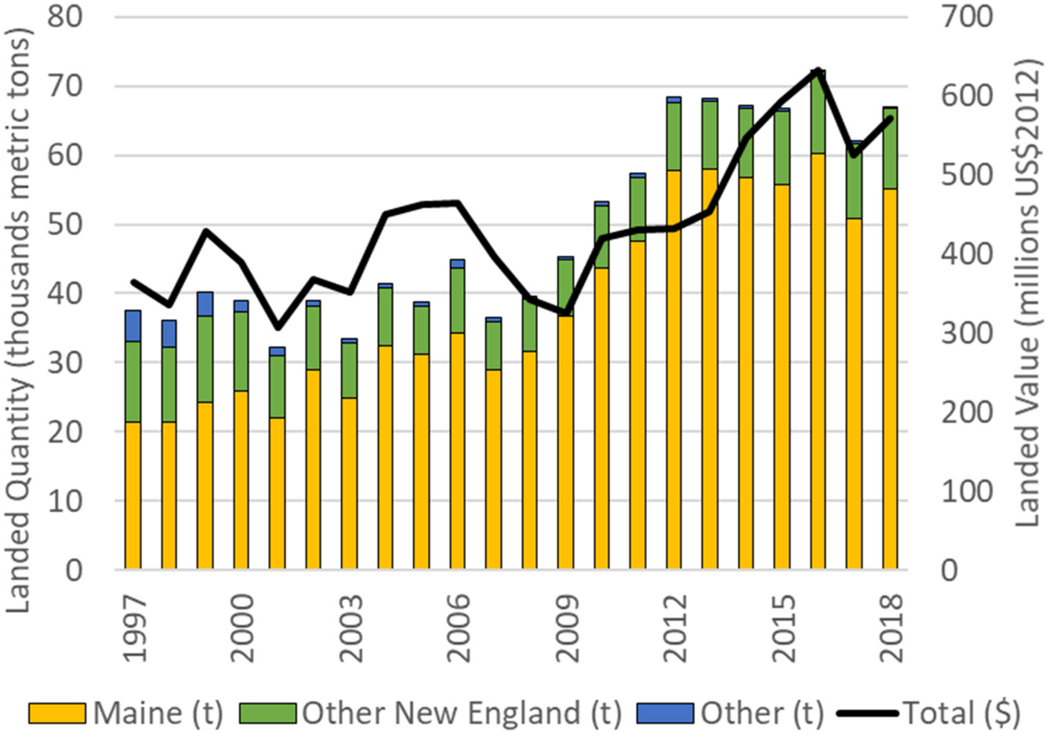

FIGURE 1 | United States Lobster (*Homarus americanus*) landings for Maine (stacked yellow column), other New England states (stacked green column) and the rest of the United States (stacked blue bar) in thousands of metric tons and total landed value (solid line, right axis) in millions of United States dollars (US$2012), 1997–2018. New England coastal states include Maine, New Hampshire, Massachusetts, Rhode Island, and Connecticut. Source: NOAA Fisheries. Commercial Fisheries Statistics. https://www.fisheries.noaa.gov/national/sustainable-fisheries/commercial-fisheries-landings.

fin whale (*Balaenoptera physalus*), and benefited minke whales (*Balaenoptera acutorostrata)*, the spatial management rules in the Plan focused on the right whale.

Marine animals can become entangled in fishing gear, with impacts ranging from no permanent injury to immediate drowning. Large whales, such as right whales may not be in danger of immediate drowning as they can often pull all or parts of the gear off the ocean floor, and so the likelihood of observing an entanglement at the contact point is low. When these whales travel with gear they can suffer serious injury and potentially mortality, even if they eventually lose the gear. Most adult right whales have scarring evidence of multiple encounters with gear which can affect health, reproduction and survival (Robbins et al., 2015; Knowlton et al., 2016).

The objectives of the Plan are determined by statutory benchmarks of the MMPA, which require reductions in levels of bycatch in relation to the Potential Biological Removal level (Wade and Angliss, 1997) and timelines to achieve the reductions. Observed fishing mortality for right whale was, and continues to be, above the Potential Biological Removal level.[8] Borggaard et al. (2017) provides a timeline for the major rule making actions under the Plan between July 1997 and May 2015, as well as the process for the development of the rules; the Plan continues to be adapted based on need with the most recent proposed changes published December 31, 2020 (85 Federal Register 86878, December 31, 2020).

Assessing the biological effectiveness of the Plan in achieving a reduction in entanglements has been challenging (Borggaard et al., 2017). This can be illustrated by data from 1990 to 2017, where of the total modeled right whale mortalities, on average only 36% of right whale carcasses were detected annually[9] and approximately half of the carcasses had sufficient information to determine the cause of death based on necropsies and other analyses (Henry et al., 2019; Pace et al., 2021). Given right whales ability to carry gear for long distances, in instances where gear is recovered there is significant uncertainty in assigning the location, type and configuration of the gear involved in an entanglement, despite ongoing gear marking requirements starting with the first Plan (62 FR 39157, July 22, 1997). These technical difficulties, coupled with the statistical rarity of observing entanglements for large whales, makes it nearly impossible to assess the biological success of a particular policy instrument within the Plan.

[8]Stock assessment reports which include information on mortality and the Potential Biological Removal level can be found here: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-stock-assessment-reports-species-stock (accessed August 13, 2020).

[9]Pace et al. (2021) used an abundance estimation model to derive estimates of cryptic mortality for North Atlantic right whales and found that observed carcasses accounted for only 36% of all estimated deaths from 1990 to 2017.

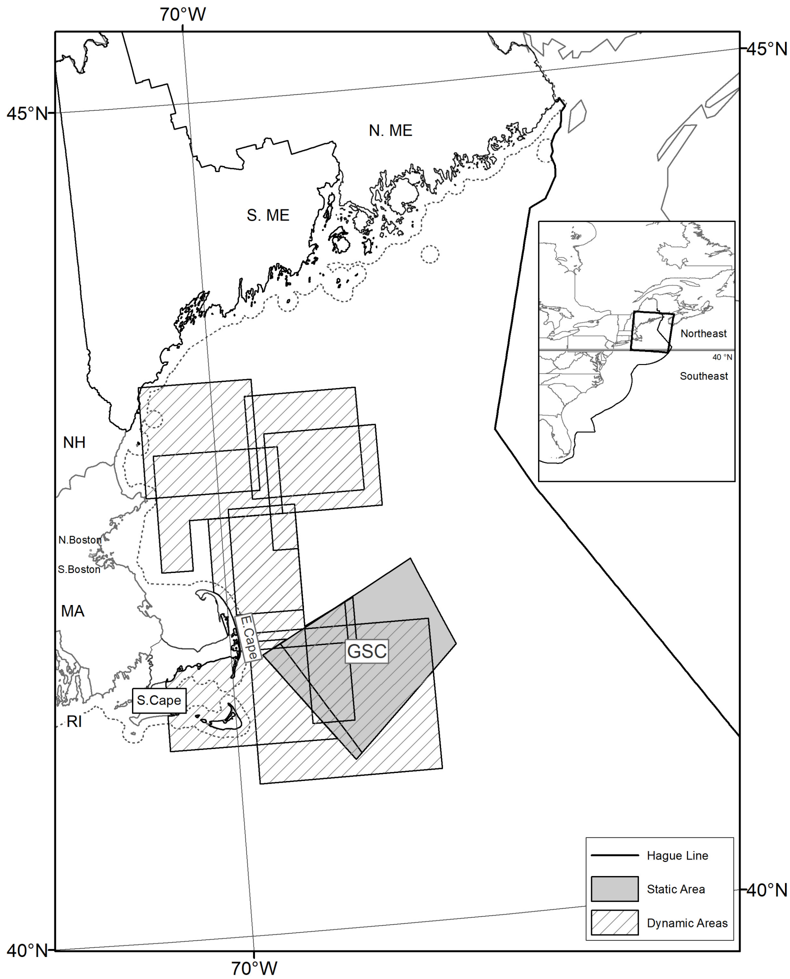

FIGURE 2 | Right whale management areas in the northeast United States that required *removal* of lobster fishing gear during certain times of year (i.e., *avoid* risk), as identified in the Atlantic Large Whale Take Recovery Plan, 2002–2009, with port groups and state waters (dash line 3 nm from shore) identified. Great South Channel (GSC) critical habitat area (solid gray area), and Dynamic Area Management (DAM) zones (hatched area).

The first Plan limited closures to critical habitat areas and gear modifications were identified as the preferred alternative to minimize economic impacts (62 FR 16519, April 7, 1997: 16531), despite the uncertainties regarding the effectiveness of these modifications. In response to high levels of serious injury and mortality, regulations were temporarily implemented in 1997 to seasonally manage lobster gear, and later made permanent. Lobster gear was prohibited in Great South Channel (April 1–June 30) while modified lobster gear could be used in Cape Cod Bay (January 1–May 15) (50 CFR 229.30). A limit on floating buoy lines and the need to select modifications from a list of options applied to almost the entire U.S. lobster fishery and changed over the years.

Both the Seasonal Area Management (SAM) and DAM programs were implemented in 2002 as part of the second amendments to the Plan. The SAM program (67 FR 1142, January 9, 2002) included additional gear restrictions for two temporally and spatially static areas (**Figure 3**, solid gray area) defined by historical north Atlantic right whale aggregations (Merrick et al., 2001). SAM West (March 1–April 30) and SAM East (May 1–July 31) zones required sinking or neutrally buoyant groundline and a single buoy line per trawl in addition to the sinking or neutrally buoyant buoy line and weak links required more broadly.[10]

The DAM program was a complement to the SAM program (67 FR 1133. January 2, 2002) (**Figures 2**, **3**, hatched area). The program operated year round in U.S. waters north of 40°N with restrictions for each individual zone to be in place for 2 weeks, unless extended or removed. The implementing regulations laid out criteria for when a DAM zone would be triggered, how the size of the zone would be determined and the process for notifying fishers of the DAM zone and requirements. The ex-ante regulatory analysis of mandatory gear removal in DAM zones estimated forgone lobster revenues

[10]Weak links are rope attachments that are manufactured to break at a strength lower than that of the rope used, or otherwise reducing the breaking strength of lines used in fishing.

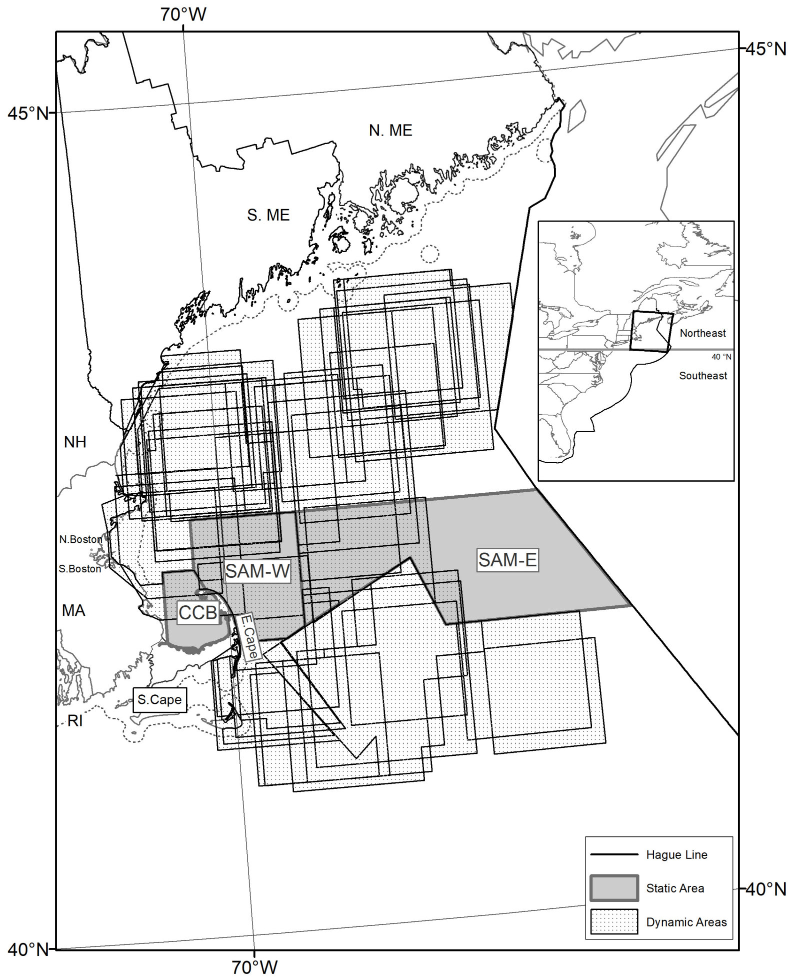

FIGURE 3 | Right whale management areas that required the use of whale-modified-gear during certain times of year (i.e., *minimize* risk), as identified in the Atlantic Large Whale Take Recovery Plan, 2002–2009, with port groups and state waters (dash line 3 nm from shore) identified. Static areas included Cape Cod Bay (CCB) critical habitat area and Seasonal Area Management zones East (SAM-E) and West (SAM-W) (solid gray area). Dynamic areas include Dynamic Area Management (DAM) zones (hatched area).

of $3.2M (US$2000, or $4.1M US$2012) for the first year (National Marine Fisheries Service (NMFS), 2001). Within the regulation, however, authority remained with the NOAA-Fisheries administrator to determine on a case-by-case basis if an area would be subject to a mandatory or voluntary closure. Of the seven DAM zones implemented between April 1, 2002 and September 25, 2003, only one mandated gear removal while the others were notices for voluntary removal of gear (**Figure 2**, hatched areas).

As an alternative to gear removal in the DAM zone, the NOAA-Fisheries administrator could determine if gear with specific modifications could be allowed to fish in the zone. In September 2003, a standardized list of acceptable gear requirements became available for use in DAM zones (68 FR 51195, August 26, 2003). Gear that did not meet the requirements was to be removed within two days of the publication in the Federal Register of a notice of a DAM zone. The gear specifications were required no matter where the DAM zone was identified unless more restrictive requirements were in place; for example, the Great South Channel critical habitat closure would prevail over DAM gear requirements. The gear specifications identified for use in DAM zones in 2003 followed those for SAM areas with the exception that an additional buoy line was allowed in the DAM zones, meaning SAM requirements were considered more restrictive. The ex-ante regulatory analysis to allow use of modified gear in DAM zones was estimated as $31,100–$93,700 (US$2002 or $38,400–$115,600 US$2012) for the fleet (National Marine Fisheries Service (NMFS), 2003).

In April 2009, amendments to the Plan ended the SAM and DAM programs (73 FR 51228, September 2, 2008), and marked the next step in implementation of the management strategy identified in 2003. Broad-base gear modification requirements for the gillnet and trap/pot fisheries, over much of the United States Atlantic coast from Maine to Florida, became effective year-round in the northeast and seasonally in other areas (Borggaard et al., 2017). Amendments in 2014 and

2015 focused on reducing risk from vertical line, as well as expanded gear marking requirements and time and area closures (Borggaard et al., 2017).

MATERIALS AND METHODS

Assessing the performance of policy instruments for right whale protection requires an understanding of the spatial-temporal density of the whales and fishing effort in the management areas to assess the risk of entanglement. We compare dynamic (i.e., DAM zones) and static (i.e., SAM zones and Cape Cod Bay critical habitat) and closed (Great South Channel critical habitat) right whale management areas, to the remaining "open" area, in U.S. waters north of 40 degrees N latitude to The Hague line with Canada, an area of approximately 176,400 km^2.

In the "Data" section we present the data on the NOAA-Fisheries aerial surveys used to trigger the DAM zones, commercial lobster fishing data, and right whale data. The "Methods" section describes the methods used to calculate a RREI, and an illustrative optimization closure model under two management decision rules, a management objective based strictly on benefits ("benefit-ranking") vs. an objective that simultaneously considers costs and non-monetized benefits ("cost-effectiveness"). Additional scenarios then consider implications of the use of whale-modified-gear and non-compliance.

Data

The temporal scale for all data are at the month level for April 2002 to March 2009, the period in while the DAM program was active. ArcGIS software is used to establish a standardized coordinate system to produce an initial 20 km^2 vector grid of the data (see **Supplementary Section 1**).

Data for Detecting and Implementing a DAM

The North Atlantic Right Whale Sighting Survey, a NOAA-Fisheries aerial survey program, provided real-time data to NOAA-Fisheries managers to determine if and where to trigger a DAM. The Survey locates and records the seasonal distribution of right whales off the northeastern coast of the United States (Khan et al., 2010). The goal of each flight is to identify locations of large aggregations of right whales. After each flight, the sightings data were used to determine whether the density of right whales was above the threshold to trigger a DAM (i.e., 0.04 right whales per nm^2). If the density threshold was reached, the survey team notified the Greater Atlantic Regional Fisheries Office. The Regional Office was responsible for issuing a Federal Register notice to notify the public, the fishing industry in particular, that a DAM zone was to be implemented. The notification included whether the DAM was voluntary or mandatory, the location coordinates, any specific conditions (e.g., gear requirements), and the start and end date of the DAM zone requirements. The Greater Atlantic Regional Fisheries Office provided the authors the notice details for each DAM zone, along with the GIS shape files for all right whale management areas, including critical habitat, SAM and DAM zones (**Figures 2**, **3**). Roberts et al. (2016) incorporates these data from the North Atlantic Right Whale Sighting Survey in the density map estimates discussed below. The recorded NOAA aerial survey flight track line data were assigned to the 20 km^2 vector grid (see **Supplementary Section 1**).

The aerial survey data are the primary source for *detecting* potential DAMs and are considered an important element of the overall performance of the DAM program. We are interested in measuring the sampling coverage, the actual number of flight-days per year compared to the number of flights required to sample the Bay of Fundy/Gulf of Maine water body for each of 365 days of the year. A bare-bone survey without any fly-backs over areas (C. Khan, pers. comm., August 2020), would require 2,920 flight-days (=8 simultaneous flights per day for a one-day-snapshot × 365 days) to have 100% sampling coverage (Palka, 2012).[11] The annual and temporal frequency distribution of flights can shed light on how well U.S. waters were sampled to detect DAMs and therefore provide additional right whale protection beyond the static right whale management areas.

Right Whale Data

For the United States Atlantic and Gulf of Mexico, Roberts et al. (2016) integrated 23 years of aerial and shipboard cetacean surveys, linked them to environmental covariates obtained from remote sensing and ocean models, and built habitat-based density models for 26 species and 3 multi-species guilds using distance sampling methodology. The density maps for these regions are the first to be published in the peer-reviewed literature. We use the monthly right whale density maps supplied by Roberts et al. (2016) to quantify the total right whale population within our spatial-temporal strata (W_{ta} for month t, area a). The aggregate monthly data are static across years and are based on survey effort from 1998 to 2017, and therefore they do not represent the density of whales in a given area in a particular year (Roberts et al., 2016; Center of Independent Experts, 2019). The right whale density map data was converted to the 20 km^2 vector grid, consistent with the aerial survey track lines.

Commercial Fishing Data

Data from several sources were used to provide an overall estimate of the spatial distribution of lobster fishing trips. In the northeast United States, NOAA-Fisheries collects VTR; these records identify northeast "lobster trap/pot" fishing trips. This is the only source for trip locations. For each trip (i.e., the "raw" data), a single location point is recorded to represent the place where the most hauls occurred for a trip. NOAA-Fisheries also collects first-point of sale data for federally managed fish from entities that buy fish directly from federally permitted fishing vessels; these data are referred as "dealer" data and include value and volume but do not include trip location. The Atlantic Coastal

[11]The 2011 NOAA-Fisheries Northeast Fisheries Science Center aerial abundance line transect survey covered the Gulf of Maine/Bay of Fundy stratum ranging from New York, United States to St. John, New Brunswick, Canada (about 40°N–45°N latitude) and from the shore to about the 100 m depth contour, using a NOAA Twin Otter airplane during 4–26 Aug 2011 (Palka, 2012, see **Figure 3**). On-effort surveying occurred during 8 good weather flight days for a total of 34.5 h for a total of 5,313.2 km^2 track line. We assume flight-days is equivalent to number of individual flights.

Cooperative Statistic Program collects "dealer" data for state permitted vessels and shares these data with NOAA-Fisheries. The NOAA-Fisheries "dealer" data used in this analysis includes both federal and state landings data. A common variable across all datasets is port of landing and weight of catch. The NOAA-Fisheries Greater Atlantic Regional Office's vessel permit database identifies federally permitted vessels' physical characteristics, such as horsepower, length and gross registered tons. We use the Atlantic Coastal Cooperative Statistic Program as an additional source of information to identify vessel characteristics for state permitted vessels.

We assume the NOAA-Fisheries dealer data are a census of lobster landings and VTR data are a subset of these data. The 2002–2009 NOAA-Fisheries dealer data contains roughly 94–100% of lobster landings as reported by the Atlantic State Marine Fisheries Commission (2018). Trips recorded in the VTR are used to spatially allocate dealer landing not recorded in the VTR to six different port groups (see **Figures 2**, **3** for location of port groups). This provides data to estimate the overall spatial and temporal distribution of trips for the American lobster fishery.

In 2002, the NOAA-Fisheries dealer data indicated that northern and southern Maine combined landed approximately 66% of the total lobster weight (**Table 1**) while 34% was landed by the remaining southern port groups. The distribution was similar in 2009. The share of total lobster landings reported in the VTR ranged from high of 99–100% in New Hampshire to a low of 6–7% in northern Maine. Trips from northern Maine were largely allocated to the open area, as the majority of right whale management areas are adjacent to southern ports (**Figures 2**, **3**). Since exclusion of northern Maine data would provide an incomplete picture of the scale and distribution of fishing trips within the lobster fishery, we include northern Maine dealer data despite the low VTR reporting rates.

The large difference in VTR coverage by port groups would allow for finer spatial resolution for some (e.g., New Hampshire) port groups but not others (e.g., northern Maine). To accommodate this difference in data resolution, the spatial stratification chosen for fishing areas is at the level of right whale management area (i.e., dynamic, static, closed) and open (i.e., remaining area with general lobster gear requirements). Trip, the decision point for every harvester, is our unit of fishing effort for this analysis. We estimate trips by year y, month t, port group p, and fishing area a for each year; we then sum over port groups to estimate fishing effort, E_{yta}. Fishing area a is either a dynamic, static, closed or open management area. Details on E_{yta} calculations can be found in the **Supplementary Section 2**.[12]

In the northeast, the total number of lobster trap/pot fishing trips ranged between 210,320 and 284,951 between 2002 and 2009, using the combined VTR and Dealer data (**Table 2**). Total landings range between 32,603 and 45,026 metric tons, while revenues range between \$324.4M and \$457.1M (US\$2012). New Hampshire and north Boston port groups fished as much as 6.6 and 6.2% of their annual lobster trips in right whale management areas in later years, respectively. See **Supplementary Table 2** for the total number of fishing trips by port group and the percent of trips in right whale management areas.

TABLE 1 | The distribution by port group of total lobster landings (dealer), and the percent of landings for that port group in the Vessel Trip Report (VTR).

	2002		2009	
	Dealer	VTR	Dealer	VTR
Northern Maine	0.45	0.07	0.53	0.06
Southern Maine	0.21	0.14	0.21	0.08
New Hampshire	0.03	1.00	0.03	0.99
North Boston	0.06	0.36	0.04	0.39
South Boston	0.06	0.75	0.05	0.50
East of Cape Cod	0.01	0.48	0.01	0.27
South of Cape Cod	0.18	0.26	0.13	0.71
Total	1.00		1.00	

Source for data: NOAA-Fisheries dealer and VTR data. Data requests to: nmfs.gar.data.requests@noaa.gov.

Methods

A relative risk of entanglement index (RREI) is calculated to allow spatial-temporal comparisons of potential non-monetary benefits of right whale management areas between 2002 and 2009. We develop an optimization model using closures to compare the cost frontier for alternative management decision rules (benefit-ranking vs. cost-effectiveness) to reduce the risk of entanglement.

Estimating Risk of Entanglement

Entanglement risk is jointly determined by the co-occurrence of gear and whales in the water in time and space, and the potential impact of the gear on whales. Following from others (Wiley et al., 2003; Vanderlaan et al., 2011; Farmer et al., 2016; Brilliant et al., 2017), we use fishing effort and whale presence within an area to develop a risk of entanglement. We measure whale presence in terms of number of whales per grid cell (20 km^2) based on Roberts et al. (2016). Comparisons are made between right whale management areas (static and dynamic) and the open area.[13]

The risk of entanglement (RE) is the product of the number of whales and the density of trips. We use trips as a first proxy for vertical lines in the water.[14] An important risk factor is gear lethality (Vanderlaan et al., 2011), although there is no scientific data to support parameter estimates of the impact on whales of alternative gear designs. Thus, for this first estimate of risk, we assume a homogenous gear configuration within a given time and

[12] VTR data records the configuration of fishing gear (i.e., number of hauls, traps-per-haul, soak time, and catch, etc.) for individual trips; however, the data for fixed gear (i.e., sink gillnet, lobster traps, etc.) was not recorded consistently by harvesters making it problematic for analysis. While the NOAA-Fisheries Northeast Fisheries Observer Program, which collects data for a sample of trips, records information at the haul level and provided details on fishing effort (e.g., number hauls, number of traps, soak time, catch, etc.), the data for 2002–2009 are too sparse to be used for spatially explicit estimates of the number of hauls per trip. This framework would have benefited from census level reporting of VTR records along with vertical line location and configurations, as would other management objectives.

[13] Since the closed area (i.e., GSC) has no fishing effort, the risk of entanglement is zero.

[14] This proxy can be modified by weighting the trips in different times or areas to reflect the average alternative gear configuration in terms of number of lines, should a complete set of data become available.

TABLE 2 | Total trips, landings (metric tons) and landed value (revenues in millions of US$2012) and percent of total for the Dynamic Area Management (DAM) zones, the static whale-modified-gear areas and for entire area north of 40-degree north latitude from shore east to the Exclusive Economic Zone.

	Trips					Landings					Revenues				
	Dynamic		Static		Total	Dynamic		Static		Total	Dynamic		Static		Total
	#	%	#	%	#	T	%	T	%	Total	M	%	M	%	M
2002	971	0.4	235	0.1	242,066	165	0.5	90	0.2	36,050	1.770	0.5	0.858	0.3	334.140
2003	259	0.1	222	0.1	210,320	50	0.2	80	0.2	32,603	0.553	0.2	0.870	0.3	324.355
2004	541	0.2	349	0.1	266,126	71	0.2	85	0.2	40,675	0.817	0.2	0.954	0.2	413.243
2005	1,076	0.4	184	0.1	266,169	121	0.3	73	0.2	41,140	1.477	0.3	0.936	0.2	457.136
2006	3,227	1.1	284	0.1	284,951	554	1.3	66	0.2	43,046	5.556	1.3	0.840	0.2	414.816
2007	3,959	1.7	185	0.1	239,584	641	1.7	39	0.1	38,216	6.886	1.8	0.539	0.1	378.753
2008	2,832	1.1	265	0.1	263,482	462	1.1	60	0.1	42,183	3.759	1.1	0.760	0.2	329.970
2009	392	3.9	201	2.0	10,147	39	1.3	18	0.6	3,045	0.460	2.3	0.227	1.1	19.729

For period April 1, 2002–March 31, 2009. Source for data: NOAA-Fisheries Dealer and VTR data. DAM GIS shape files provided by the NOAA's Greater Atlantic Regional Office, Protected Resource Division. Data requests to: nmfs.gar.data.requests@noaa.gov. GDP deflator: Federal Reserve Economic Data. Gross domestic product (implicit price deflator), Index 2012 = 100, Annual, not seasonally adjusted (https://fred.stlouisfed.org).

area, such that the relationship between trips and lines is exact. The density of fishing trips is used as a proxy for the likelihood of a whale interaction with gear, where density is the number of trips divided by the total km^2 for the relevant area ($\frac{E_{yta}}{Km^2_{yta}}$), y is year, t is month, and a is area. We assume "a single line" in the water in different times and areas presents the same level of risk to whales.[15] In Roberts et al. (2016) monthly whale numbers, W_t are constant, however, the spatial temporal placement of DAM zones varies over the years resulting in a change in the number of whales (W_{yta}) within a zone, as well as fishing effort (E_{yta}). The distribution of trips and whales is assumed uniformly distributed within a stratum. Here we estimate a risk of entanglement (RE_{yta}) for any time-area, by assuming it is proportional to the product of the number of whales at risk (W_{yta}) and the likelihood of a whale interaction ($\frac{E_{yta}}{Km^2_{yta}}$).

To make comparisons between management areas which differ in size, a RREI is calculated. The RREI is normalized 0–1, to the maximum RE $_{yta}$, the open area in July 2006 (i.e., $\mathrm{RREI}_{yta} = \mathrm{RE}_{yta}/\mathrm{RE}_{2006,07,open}$). This provides a relative measure between months and areas against this base. Changes in the RREI provide a consistent proxy for the potential non-monetary benefits of risk reduction of alternative right whale management measures.

Reducing Risk Under an Optimization Framework

The optimization closure model minimizes a management objective, subject to a set of constraints, and is solved using the R package "lpSolve" version 5.6.15 (Berkelaar, 2020). Two management decision rules are considered, one based strictly on non-monetized benefits ("benefit-ranking") and the other which considers costs and non-monetized benefits simultaneously ("cost-effectiveness"). For this illustrative model, a representative year was generated by averaging monthly data across years for number of whales (W_{ta}) and trip density ($\frac{E_{ta}}{Km^2_{ta}}$) for the risk calculation, creating an average risk profile for each month and management area type, which is assumed exogenous. While the size of fishing areas vary, size is implicitly imbedded in the calculation of cost and risk.[16]

The expected non-monetary benefit of a closure is a reduction in entanglement risk (RE_{ta}), which is a function of density of fishing trips and whale numbers. The required level of overall risk reduction ($\overline{R}$) is exogenously determined based on legal mandates (e.g., MMPA and ESA). Risk is reduced along a continuum between no expected change (i.e., Status Quo: $\overline{R} = 0\%$) to a complete reduction in risk (e.g., complete closure: $\overline{R} = 100\%$). This allows us to trace out a cost frontier. More formally, let x_{ta} be the control variable. A complete closure ($x_{ta} = 1$) or partial closure ($0 \leq x_{ta} < 1$) directly reduces fishing effort with an opportunity cost (c_{ta}) to fishery participants.[17] We assume fishing effort does not shift to other times or areas.[18] Mathematically our model is:

$$Minimize \sum_{t=1}^{12} \sum_{a=1}^{3} c_{ta} x_{ta} \tag{1}$$

$$Subject\ to \quad \sum_{t=1}^{12} \sum_{a=1}^{3} \frac{RE_{ta}(\alpha_a)(1-\beta_a)(x_{ta})}{\sum_{t=1}^{12} \sum_{a=1}^{3} RE_{ta}} \geq \overline{R} \tag{2}$$

where $0\% \leq \overline{R} \leq 100\%$; $0 \leq x_{ta} \leq 1$ (0 is open and 1 is fully closed); $0 \leq \alpha_a \leq 1$; $0 \leq \beta_a \leq 1$.

We include a measure of the efficacy of whale-modified-gear (α_a) in reducing the risk to right whales where 1.0 indicates the

[15] Individual trips could be weighted to account for lethality differences associated with different methods of deployment (i.e., number of traps in a trawl, depth) and characteristics (e.g., breaking strength of line), should that data become available.

[16] Within a cost-effectiveness analysis the issue of scale can be a concern; however, the risk constraint within the optimization eliminates this issue.

[17] For this illustration, the revenues potentially lost due to a closure are used as a proxy for opportunity costs.

[18] If instead effort is displaced this would result in an overestimation of reductions in revenues and risk to whales. For use in a management setting we would incorporate a harvester behavioral model to reflect shifts in fishing effort in response to closures (e.g., Holland and Sutinen, 2000; NOAA (National Oceanic and Atmospheric Administration), 2009; Dépalle et al., 2020).

whale-modified-gear is 100% effective at reducing the impact on the whale (e.g., rope-less gear) and 0.0 indicates that there is no change in the impact on the whale and thus risk is not reduced (i.e., no whale-modified-gear).[19] The rate of non-compliance (β_a) can also impact the efficacy of the whale-modified-gear. The net entanglement risk for a given time and area is $(RE_{ta})(\alpha_a)(1 - \beta_a)$.

In scenario 1 ("benefit-ranking"), a decision to close a fishing area is based exclusively on the non-monetary benefits of a reduction in entanglement risk. The objective is to minimize the number of closures ($c_{ta} = 1$) while simultaneously meeting the risk constraint ($\bar{R}$). Under scenario 2 ("cost-effectiveness"), a decision maker is interested in minimizing the opportunity costs to the fishing industry (c_{at}) while also meeting the risk constraint ($\bar{R}$). For these two scenarios we assume closing an area eliminates risk of a gear-whale-encounter ($\alpha_a = 1$) and there is zero non-compliance ($\beta_a = 0$, i.e., full compliance). We then investigate in Scenario 3 the impacts of closures when gear-modifications (α) are already in place and non-compliance (β) is zero (**Table 3**). In the absence of data on the efficacy of whale-modified-gear, we assume the required gear modifications in dynamic and static areas are more effective (i.e., assume $\alpha_{dynamic}$ and α_{static} both equal 0.9) compared to the general gear requirements in the "open" area as described in the "Measures to Reduce Right Whale-Fisheries Interactions" section (i.e., assume $\alpha_{open} = 0.45$).[20] Scenario 4 then considers non-compliance with whale-modified-gear requirements that is greater than zero ($\beta = 0.6$). In scenarios 3 and 4, the gear efficacy and non-compliance assumptions result in risk reductions occurring prior to running the optimization closure model. The cost frontier illustrates the total cost to industry compared to the total non-monetary benefits of a reduction in risk along the continuum between no expected change to a complete reduction in risk, for each of the four scenarios.

RESULTS

Public Sector: Detecting and Implementing DAMs

The degree to which DAMs could provide a reduction in risk to right whales was dependent on public investment in detection and the regulatory requirements for implementation. Of the 589 flight-days between April 2002 and March 2009, the NOAA North Atlantic Right Whale Sighting Survey detected right whale aggregations that triggered 66 DAMs with an average annual detection rate of 11% (=66/589) (**Table 4**). Between 2002 and 2007, the annual DAM detection rate increased from 4 to 25%.[21] From a seasonal perspective, flights in November to March had the highest DAM detection rates; yet the frequency of flights was greatest from April to June.

The detection of DAM zones, designed to provide protection for large aggregations of right whales, was influenced by the sampling frequency. The aerial survey team flew 84 flight-days a year on average between 2002 and 2009 (**Table 4**). Approximately 2.9% of the water was sampled on average in a year, based on the ratio of actual flight-days to number of flights required for 100% sampling (=84/2,920 flights).[22] Sampling varied by month. In May, the month with the largest percent of the population present in northeast U.S. waters (Roberts et al., 2016), on average they sampled roughly 5.1% [=89 flights/7 years)/(8 flights/day × 31 days)] of the grid.

The speed at which a DAM can be implemented impacts its effectiveness in delivering right whale protection. The number of days it took to implement the DAM (12.8 days, $CV = 0.21$) was only slightly less than the number of days the DAM was in effect (14.9 days, $CV = 0.25$). Over the history of the DAM program, it took an average of 7.7 days ($CV = 0.36$) from the day the DAM was triggered based on the date of the aerial survey, to the publication of the federal register notice. It took an additional 5.2 days ($CV = 0.38$) on average, between the published Federal Register notice and the first day the DAM measures were implemented, as described in the notice. The data were insufficient to determine whether whales were still present during the implementation period (i.e., 12.8 days after trigger); flights over DAM zones after implementation were too limited to calculate any statistics.

Right Whale Management Areas and Distribution

The right whale density maps (Roberts et al., 2016) indicate that more than 50% of the right whale population was in U.S. waters between November and June (**Figure 4**, stacked columns). The largest share of the total population is in the southeast United States during fall-winter (November to February), in the northeast United States in the spring (March to June) and outside U.S. waters in summer-fall (July to October).

Right whale management areas were spatially a small share of U.S. waters north of 40°N (**Figure 4**, stacked lines), yet a significant share of the right whale population was present in the management areas (**Figure 4**, stacked column). In May for example, on average 17% of the northeast area waters was

[19]Lethality of entanglement could be, but currently is not, incorporated in the model. As Vanderlaan et al. (2011) states, it is not possible to estimate in a robust manner the lethality of an entanglement among gear types. Additional information would be needed on the mechanics of a fishing-gear entanglement, the likelihood of each gear type entangling a whale, the likelihood of self-disentanglement by gear type, and the standardization of entanglement rates by fishing effort.

[20]The assumed values for α, are for illustration only, as there is no empirical evidence available. Research on efficacy of gear modifications for other marine mammals, such as pingers for harbor porpoise, has been assessed using NOAA's Northeast Fisheries Observer Program data (Palka et al., 2008). However, as discussed in the "Background" and "Data and Methods" sections observer data are not available for assessing efficacy of whale-modified-gear for right whales.

[21]Broadscale surveys were the primary survey type from 2002 to the spring of 2007 (Khan et al., 2018). The survey covers all federal waters from New York to Maine. Survey lines were flown year round whenever good weather occurred, unless there was a pressing management issue. Starting in the summer of 2007 sawtooth surveys were implemented as part of the mark-recapture effort for stock assessments. Blocks were made around areas with high numbers of right whale sightings. Sawtooth surveys also supported other management area needs, such as DAM detection.

[22]As an alternative to proportional sampling, a stratified random sampling approach (Cochran, 1977) could be implemented to evaluate sampling tradeoffs. For example, increasing sampling in areas where the variance is higher, and vice versa, may reduce the total variance for improved precision around the statistic of choice (e.g., whale sightings per unit effort).

TABLE 3 | Criteria for management decision rules in optimization framework, by scenario (Eqs. 1 and 2).

Scenario	Decision rule	Efficacy of gear modification (α_a)	Degree of non-compliance(β_a)
1	Benefits-ranking	1.0	0 (all areas)
2	Cost-effectiveness	1.0	0 (all areas)
3	Cost-effectiveness	0.9 (dynamic/static) and 0.45 (open)	0 (all areas)
4	Cost-effectiveness	0.9 (dynamic/static) and 0.45 (open)	0.6 (all areas)

TABLE 4 | The Dynamic Area Management (DAM) detection rate (=No. DAMs/No. flight-days) for broadscale and sawtooth survey methods by year and month. The total number of DAM zones triggered, North Atlantic Right Whale Sighting Survey flight days[a] and average detection rate (avg detect rate) across years and months. Shaded areas show detection rates of 20% or greater and NF = no flights.

	Detection rate by month								Total DAM (#)	Total flight-days (#)	Avg detect rate
	Broadscale survey					Sawtooth survey					
	2002	2003	2004	2005	2006	2007	2008	2009			
Jan		NF	NF	0.13	0.13	0	0.40	0	4	29	0.14
Feb		NF	NF	0.14	0.30	0.50	0.20	0.33	11	34	0.32
Mar		NF	0.33	0	0.09	0.38	0.50	0	8	43	0.19
Apr	0.06	0.11	0.08	0.08	0.29	0.20	0.11		8	71	0.11
May	0	0.05	0.08	0.06	0.22	0	0		5	89	0.06
Jun	0	0.04	0	0	0	0	0		1	98	0.01
Jul	0.13	0.08	0.33	0.07	0	0.33	0		5	56	0.09
Aug	NF	0	NF	0.13	NF	NF	NF		3	18	0.17
Sep	0	0	NF	0	0	NF	NF		1	31	0.03
Oct	0	0	0	0	0	0.20	0.20		2	50	0.04
Nov	0	0.2	0	0	0.18	0.33	0.40		6	42	0.14
Dec	0.50	NF	0.25	0	0.20	1.00	1.00		12	28	0.43
DAM zones (#)	4	6	9	6	12	15	12	2	66	589	0.11
Flight-days (#)	89	98	70	114	90	59	53	16	589		
Avg detect rate	0.04	0.06	0.13	0.05	0.13	0.25	0.23	0.13	0.11		

[a]The metric, flight days, is used; if two aircrafts flew the same day we counted that as 2 flight days. These data were extracted from the NOAA-Fisheries Oracle database established around 2010. Any differences in tallies of the number of 2002–2009 flights in reference documents (e.g., Khan et al., 2010), may be due to how flights were counted in earlier years and may have included non-right whale (i.e., seal) flights (pers. comm. C. Kahn, NOAA-Fisheries, Northeast Fisheries Science Center, Protected Species Branch, Aug 2020).

Source: Aerial survey data are from NOAA's north Atlantic right whale sighting survey database. DAM numbers published in the Federal Register Notices, provided by the NOAA's Greater Atlantic Regional Office, Protected Resource Division, Gloucester, MA. Send data requests to: nmfs.gar.data.requests@noaa.gov.

dedicated to right whale management areas: 4% was closed to lobster fishing (i.e., Great South Channel), 10% was in static areas, and 3% in dynamic areas (**Figure 4**, stacked lines). During the same month, of the total right whale population, 26% was in the closed area (Great South Channel critical habitat), 18% in the static areas, 1% in dynamic areas, 19% in open northeast waters and 10% in the southeast waters (**Figure 4**, stacked columns). May is an example month where both avoidance and minimization strategies complement each other, and may have provided protection to 45% of the total population or 70% of the population in U.S. waters north of 40°N.[23]

Fishing Effort and RREI

The DAM program may have created an economic incentive to whale-modified-gear to continue fishing in a DAM zone after it was triggered; this inference is based on increased fishing effort in DAMs, frequency of DAMs, and spatial overlap of zones. Between 2003 and 2008, years with complete data, the number of DAM zones varied from 6 to 15 with the maximum in 2007 (**Table 4**), and exhibited a high degree of spatial overlap (**Figure 3**, hatched areas). During this period, the share of total trips in DAMs increased from 0.1% to as high 1.7% in 2007 (**Table 2**).[24] The commercial fishing data show an increase in trips, landings and revenues within dynamic areas over time in both absolute value and share of total. Compared to dynamic areas, activity in static areas was lower, despite their larger spatial extent, and effort was more consistent. Much of the area within the static areas (e.g., **Figure 3**, SAM East) is farther offshore in federal waters compared to the location of dynamic areas, so fewer vessels would be able to access these fishing areas.

[23]Calculated as 45% in management areas/64% of right whales in northeast waters.

[24]While data on 3 months of the 2009 year were included in the analysis, they are not considered representative.

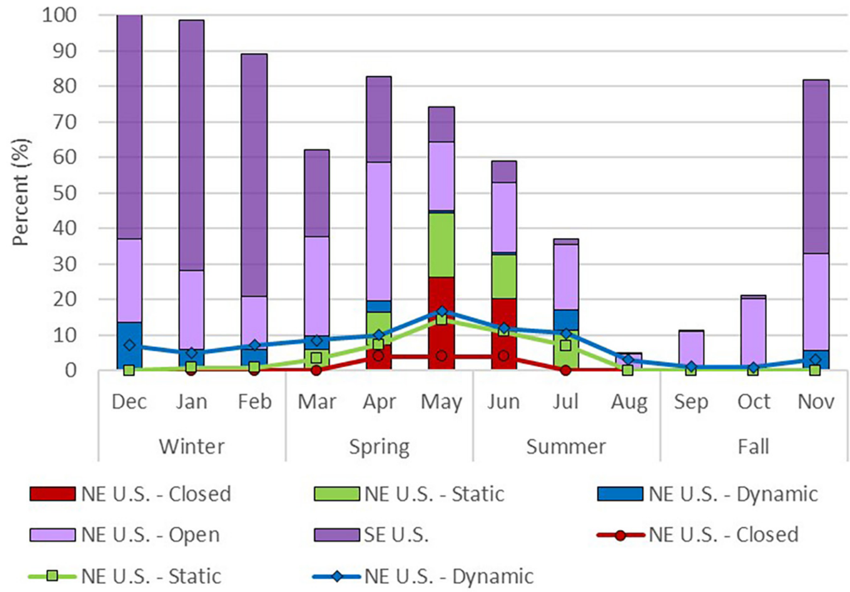

FIGURE 4 | The spatial temporal distribution of right whale population estimates (stacked bars) are compared to the share of northeast (NE) waters (stacked lines) dedicated to right whale management areas (April 2002–March 2009). Distribution of the right whale population in U.S. waters by month (stacked columns) and northeast right whale management areas including dynamic (blue), static (green), closed (red), and outside northeast management areas (i.e., open, light purple), as well as southeast (SE) waters below 40 degrees N latitude (dark purple). As much as 95% of the population estimates (August) based on Roberts et al. (2016) are not in U.S. waters; these shares unaccounted for may be in Canadian waters. In addition, the share of northeast ocean waters north of 40 degrees N latitude (stacked lines), monthly average, shows shares by right whale management area including dynamic (blue), static (green) and closed (red); remaining northeast waters are classified as "open." At the maximum, 17% of northeast waters are dedicated to right whale management in May on average.

The potential for dynamic areas to mitigate risk was highest in November to February (**Table 5**). Risk in dynamic areas accounted for 27% of the average risk for December, which may have been mitigated with whale-modified-gear, a minimization strategy in the mitigation hierarchy. The total average monthly RREI was highest in the fall (Sept-Nov) followed by the summer (Jun-Jul) (**Table 5** and **Figure 5** dashed line). Yet the percent of the right whale population present in the northeast is highest in the spring, from March to July (**Figure 5** solid line), signaling the influence of fishing effort in risk (**Figure 5** stacked bars). In some months, such as May and August the RREI value is equivalent, while the number of lobster trips (**Figure 5** stacked columns) and the percent of right whale population present (**Figure 5** solid line) have an inverse relationship.

Optimization: Reducing Risk at Lower Cost

The optimization results illustrate how different decision rules can attain equivalent non-monetary benefits of risk reduction with different opportunity costs to industry.[25] Costs are consistently lower under the "cost-effectiveness" decision rule (scenario 2) as compared to the "benefit-ranking" decision rule (scenario 1) except for the two extreme endpoints (**Figure 6** solid lines). For example, under the "benefit-ranking" decision rule a 60% reduction in risk results in a cost of roughly 60% of average revenues vs. 40% of revenues for an equivalent reduction in risk under the "cost-effectiveness" rule. The solution paths also differ (**Table 6**) in how the 60% risk reduction is achieved. For example, under the "benefit-ranking" rule four to five big open areas with high risk are the first to close. In contrast, under the "cost-effectiveness" rule, 20 smaller static and dynamic areas are closed, as these areas have a lower ratio of cost to non-monetary benefits (**Table 6**). Each unit of risk reduced was achieved at a lower cost under the "cost-effectiveness" rule (**Figure 6**). Consider May and August where total risk is somewhat equivalent (**Figure 5** and **Table 5**). Clearly the cost to reduce a unit of risk is much higher in August (low whales and high fishing effort) compared to May (high whales and low fishing effort). This leads to May being closed before August in all scenarios.

Continuing with the "cost-effectiveness" decision rule, we introduce whale-modified-gear (scenario 3), a minimization step in the mitigation hierarchy. By requiring fishermen to use whale-modified-gear and given our gear efficacy assumptions (**Table 3**), the risk of entanglement is immediately reduced by 50%, prior to any closures and assuming full compliance (scenario 3), but by only 20% in the case of non-compliance (scenario 4) (**Figure 6** dotted lines). Under these scenarios, to then reach a total reduction in risk of 60%, requires closure costs equivalent to about 10 and 30% of industry revenues, under compliance and

[25]It is not possible to provide management level recommendations from the optimization results due to data limitations and the focus on pre-2010 fishing effort. For this analysis, we extrapolate the number of trips from the Dealer data not recorded in the VTR (**Table 1**), using the VTR data, as the first step. Going forward, with additional at-sea observer data and more consistent VTR data, we should be able to construct a regression model to estimate vertical lines as a function of trip attributes.

TABLE 5 | Average monthly Relative Risk of Entanglement Index (RREI) for dynamic and static whale-modified-gear management areas, open areas, and total for the area north of 40-degree north latitude from shore east to the Exclusive Economic Zone, for April 2002 to March 2009.

		Dynamic		Static		Open		Total	
		RREI	**% of total**	**RREI**	**% of total**	**RREI**	**% of total**	**RREI**	**Rank**
Winter	December	0.086	26.6	–	0.0	0.238	73.4	0.323	6
	January	0.010	9.6	0.000	0.2	0.094	90.3	0.104	10
	February	0.008	22.5	0.000	0.1	0.028	77.4	0.037	12
Spring	March	0.003	3.3	0.006	7.9	0.072	88.8	0.081	11
	April	0.003	1.6	0.013	6.3	0.190	92.1	0.206	8
	May	0.001	0.6	0.007	3.1	0.217	96.3	0.225	7
Summer	June	0.000	0.0	0.001	0.4	0.321	99.2	0.324	5
	July	0.002	0.3	0.002	0.3	0.590	99.4	0.593	2
	August	0.001	0.4	–	0.0	0.189	99.6	0.190	9
Fall	September	0.001	0.3	–	0.0	0.383	99.7	0.383	4
	October	0.006	1.0	–	0.0	0.578	99.0	0.584	3
	November	0.089	14.3	–	0.0	0.532	85.7	0.621	1
Average		0.017	5.7	0.002	0.8	0.286	93.5	0.306	

The RREI is a value between 0 and 1; the index is normalized to the maximum risk (i.e., open area in July 2006 has an index value of 1).

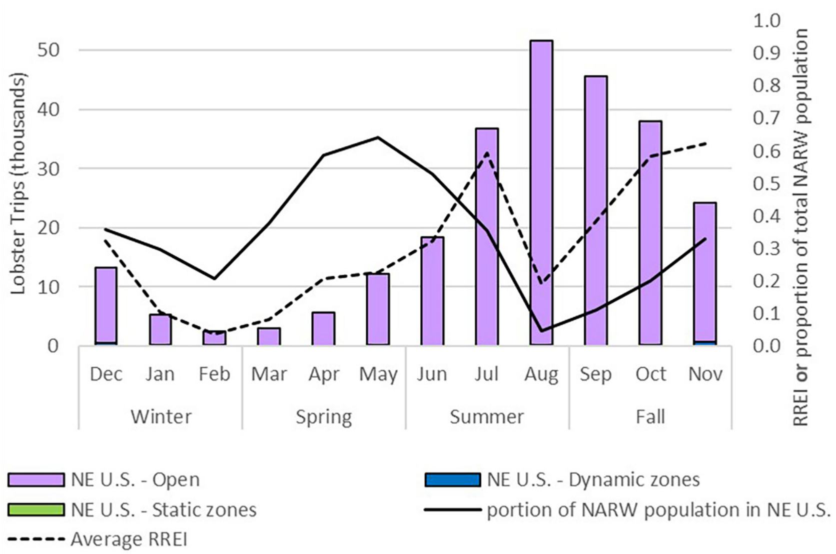

FIGURE 5 | Spatial temporal distribution in the northeast U.S. waters of the percent of the north Atlantic right whale population present (solid line, right axis) (Roberts et al., 2016), lobster fishing effort (trips, stacked columns, left axis) and the average relative risk of entanglement index (RREI, dashed line, right axis) by month (April 2002–March 2009). The average number of trips are by right whale management areas. The RREI is a consistent, relative measure of risk between areas and months with a value between 0 and 1; the base is the "open area" in July 2006 which has an index value of 1.

non-compliance, respectively. Our example illustrates that when there is non-compliance there may be an economic incentive to first address the compliance issues with the existing gear requirements, rather than implementing additional measures, such as new whale-modified-gear requirements or closures to achieve the desired risk reduction.[26]

[26]The cost of gear modifications to the fishing industry are not captured within this optimization model. This could be the case where it was (almost) costless for industry to implement a gear modification (e.g., a change in deployment that does not add cost) or whale-modified-gear is gifted/subsidized for the private sector. This may be the case as the ex-ante cost estimates for the implementation of the gear modifications for the DAM program were negligible (National Marine Fisheries Service (NMFS), 2003).

DISCUSSION

When developing a list of management alternatives, the U.S. marine mammal Take Reduction Team process relies on stakeholder consensus to balance the need to reduce marine mammal takes and implicitly minimize economic impacts on fishermen (Young, 2001). However, the assessment of costs, though informally addressed by stakeholders during the process (M. Asaro, pers. comm. October 9, 2020), are not formally

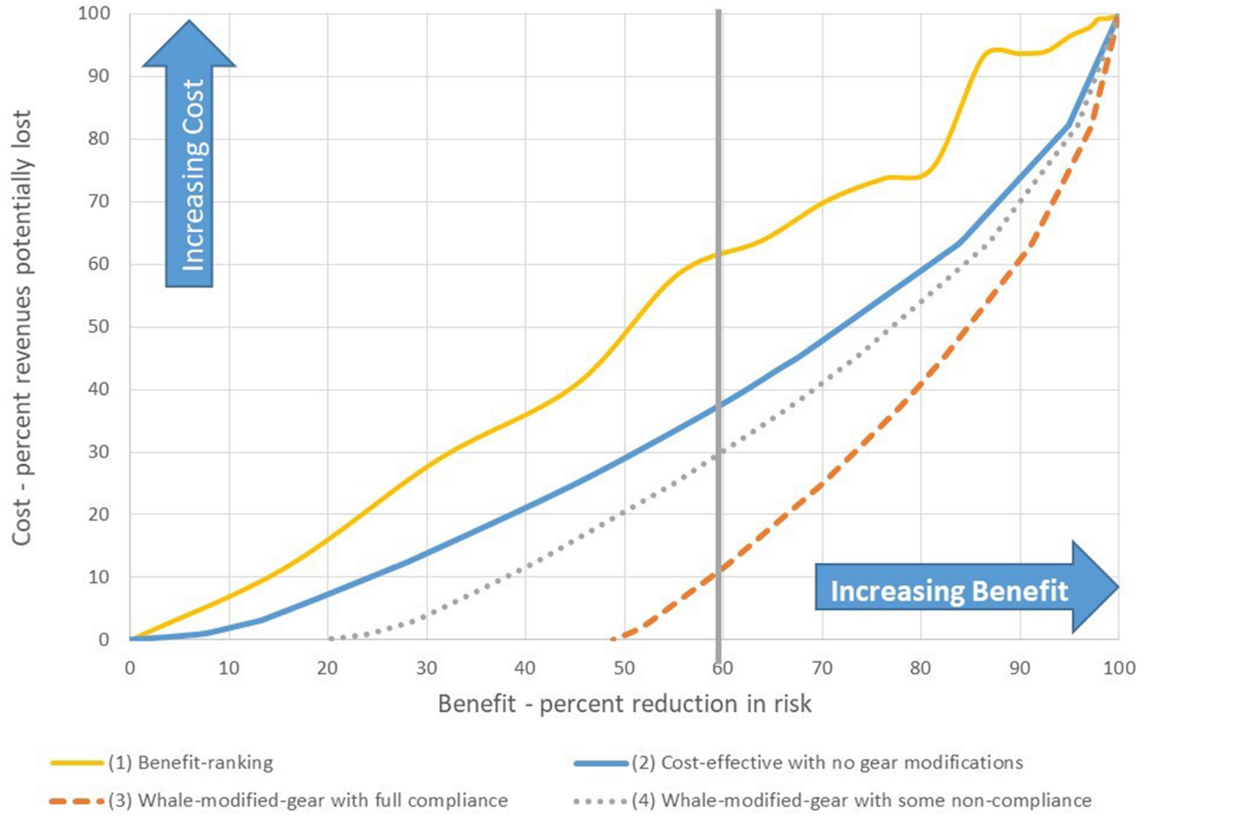

FIGURE 6 | Optimization results of trade-off between the non-monetary benefit of risk reduction and potential cost of closing months and areas (i.e., avoidance) along a continuum of no change (i.e., status quo, $\overline{R} = 0\%$) to a complete reduction in risk (i.e., complete closure, $\overline{R} = 100\%$). Scenarios (**Table 3**) include: (1) benefit-ranking with no gear modifications, (2) cost-effectiveness with no gear modifications, (3) cost-effectiveness with whale-modified-gear requirements and full compliance, and, (4) cost-effectiveness with whale-modified-gear requirements and some non-compliance.

considered until a small set of alternatives is identified and then ex-ante cost-effectiveness or cost-benefit analyses are conducted for regulatory purposes. As the optimization model illustrates, and economic theory predicts, a cost-effective solution can yield a given level of risk reduction for a lower cost than a decision based only on risk reduction. For a 60% reduction in risk from the status quo (**Figure 6**), the potential costs to reduce risk were as much as 20% lower under a decision rule that considered fishing industry opportunity costs and non-monetary benefits simultaneously (i.e., cost-effectiveness, scenario 2), as compared to a decision rule based only on the benefits of risk reduction (i.e., benefit-ranking, scenario 1). This empirical example highlights the need to integrate economic considerations throughout the management process.

A least-cost implementation approach of the bycatch mitigation hierarchy (Squires and Garcia, 2018; Squires et al., 2018) suggests that combining mitigation steps, such as was done in the Atlantic Large Whale Take Reduction Plan with avoidance and minimization measures, could further reduce economic waste and support optimal conservation when both costs and benefits are considered. Theoretically, it is possible to solve for an optimal mix of policy instruments (avoidance and minimization) which is when the marginal cost between and within mitigation hierarchy steps equates, yielding a least-cost solution for a desired risk reduction. While briefly explored in the optimization (scenario 3), the lack of information on the effectiveness of minimization policy instruments limits the conclusions that could be made. Additionally, we suggest the uncertainty associated with effectiveness of alternative management measures be included formally in the decision framework. We discuss this in more detail in our fifth recommendation.

The DAM program with whale-modified-gear, a minimization measure, was anticipated to be a lower cost alternative than the closure of the DAM zones, an avoidance measure (National Marine Fisheries Service (NMFS), 2003). This could not be evaluated due to voluntary nature of most closures and the short timeframe for the closure-based version of the program. However, the case study did identify a number of strengths of the dynamic program which can assist in the design of future mitigation measures. A major strength was the program's ability to be a flexible instrument that provided protection in response to inter-annual variability in whale density, as demonstrated by the distribution and frequency of the DAM zones. The dynamic measures may have provided incremental protection during the "shoulder" months (i.e., November to February) prior to the protection provided by static management areas in the spring.

TABLE 6 | Optimization results showing the solution paths with the cumulative percent reduction in risk (% RR) moving from no change (i.e., status quo, $\overline{R} = 0\%$) to a complete reduction in risk (i.e., complete closure, $\overline{R} = 100\%$) for scenarios 1–4.

Order	Benefit-ranking			Cost-effectiveness								
	(1) No gear modifications			(2) No gear modifications			(3) Gear modifications			(4) Gear modifications with non-compliance		
	Month-area		% RR	Month-area		% RR	Month-area		% RR	Month-area		% RR
0			0.0			0.0			48.8			19.5
1	7	Open	15.7	4	Static	0.3	4	Static	48.9	4	Static	19.8
2	10	Open	31.1	3	Static	0.5	3	Static	48.9	3	Static	19.9
3	11	Open	45.0	12	DAM	3.2	4	Open	51.7	12	DAM	21.6
4	9	Open	55.3	7	DAM	3.3	5	Open	54.9	7	DAM	21.6
5	6	Open	63.8	11	DAM	6.9	12	DAM	55.1	11	DAM	23.9
6	12	Open	70.3	4	DAM	6.9	6	Open	59.8	4	DAM	24.0
7	5	Open	76.1	3	DAM	7.0	7	DAM	59.8	3	DAM	24.0
8	4	Open	81.2	6	DAM	7.0	3	Open	60.8	6	DAM	24.0
9	8	Open	86.2	2	DAM	7.3	11	DAM	61.2	2	DAM	24.2
10	11	DAM	89.8	10	DAM	7.8	7	Open	69.8	10	DAM	24.5
11	12	DAM	92.5	5	Static	8.0	4	DAM	69.8	5	Static	24.7
12	1	Open	95.1	4	Open	13.1	11	Open	77.5	4	Open	28.8
13	3	Open	97.0	1	DAM	13.2	12	Open	81.1	1	DAM	28.9
14	2	Open	97.8	5	DAM	13.3	3	DAM	81.1	5	DAM	28.9
15	10	DAM	98.3	5	Open	19.1	6	DAM	81.1	5	Open	33.7
16	4	Static	98.7	6	Open	27.6	1	Open	82.5	6	Open	40.6
17	2	DAM	98.9	9	DAM	27.7	10	Open	91.0	3	Open	42.2
18	5	Static	99.1	3	Open	29.6	2	DAM	91.0	7	Open	55.1
19	9	DAM	99.3	7	Open	45.3	10	DAM	91.1	9	DAM	55.2
20	3	Static	99.5	11	Open	59.3	5	Static	91.1	11	Open	66.6
21	1	DAM	99.6	12	Open	65.8	2	Open	91.5	12	Open	72.0
22	3	DAM	99.7	1	Open	68.4	9	Open	97.2	1	Open	74.1
23	7	DAM	99.8	10	Open	83.8	1	DAM	97.2	10	Open	86.7
24	5	DAM	99.8	6	Static	83.8	5	DAM	97.2	6	Static	86.8
25	4	DAM	99.9	2	Open	84.6	8	Open	100.0	2	Open	87.4
26	8	DAM	99.9	9	Open	94.9	9	DAM	100.0	9	Open	95.8
27	7	Static	100.0	7	Static	94.9	6	Static	100.0	7	Static	95.8
28	6	Static	100.0	8	DAM	95.0	7	Static	100.0	8	Open	100.0
29	1	Static	100.0	8	Open	100.0	8	DAM	100.0	8	DAM	100.0
30	6	DAM	100.0	1	Static	100.0	1	Static	100.0	1	Static	100.0
31	2	Static	100.0	2	Static	100.0	2	Static	100.0	2	Static	100.0

There are 31 month-area choices ordered from first closed (Order = 1) to last closed (Order = 31); when Order = 0 all areas are open and when Order = 31 all areas are closed. Highlighted cells show alternative order for closure of May and August open areas depending on decision rule.

The dynamic and static measures combined may have reduced the total risk by 6.5% on average (**Table 5**), even with their small spatial footprint (**Figure 4** stacked lines). There was substantial spatial overlap of DAM zones over the years (**Figures 2**, **3**) and the share of total fishing effort within these zones increased over time (**Table 2**), suggesting the DAM program did not reduce fishing opportunities. Ongoing aerial surveys were required to ensure that aggregations of whales were detected and protected outside SAM and critical habitat areas. Those surveys are useful for analysis of habitat use, provide important sighting information to the Catalog and Sightings Database, and give valuable support to the disentanglement program (Reeves et al., 2007). The DAM program also proved flexible in responding to a potential safety problem for fishermen, who could be required to remove large quantities of fixed gear on short notice in poor weather (Reeves et al., 2007). By allowing whale-modified-gear and shifting to a minimization measures in 2003 the safety concern was eliminated. And while the DAM program was intended to be a temporary measure while broader gear requirements were designed and implemented (Borggaard et al., 2017), it may have provided an economic incentive for some fishers to convert to whale-modified-gear as the frequency and spatial overlap of DAM zones grew. This in turn would have reduced the cost to implement the broad-based measures in 2009.

Our study highlights some of the challenges and tradeoffs associated with the critical and increased responsibilities of the public sector to implement a dynamic management program (Grantham et al., 2008). While monitoring is required to support dynamic protection measures, such as the DAM program, it is also required for stock assessments, and to assess compliance with, and efficacy of the management measure—all public sector responsibilities. The protection delivered by DAM zones hinged on the United States government budget allocated to the NOAA-Fisheries aerial survey team for right whale detection flights. About 3% of the spatial-temporal Bay of Fundy/Gulf of Maine grid was sampled annually on average, although flights were not consistent spatially or temporally (**Table 3**). Additionally, return flights (i.e., fly backs) to designated DAM zones were sparse and did not allow an analysis to determine if the whales were still present in DAMs despite the delay in implementation. Further, data to estimate compliance was not available. For other species, such a harbor porpoise, the observer logs provide information to estimate gear efficacy; however, this was not the case for right whales. The Northeast Fisheries Observer Program logs for gillnet and trap/pot fisheries were not modified until 2007 to capture the information required to calculate compliance rates by area for different gear modifications (e.g., weak links) and low observer coverage lead to low sample sizes (National Oceanic and Atmospheric Administration (NOAA), 2012).

To manage the risk, it is critical to recognize the RREI was similar in some months when there were inverse levels of fishing effort and whale presence. While in some months as much as 70% of the population in the northeast may have been protected by the combined right whale management areas (**Figure 4**), the majority of the risk remained outside these areas and times. Fishing effort and whale distribution need to be considered jointly to reduce risk, the data on both were, and continues to be, a barrier to analysis. For our study, data related issues do limit our ability to evaluate with precision and accuracy the performance of the DAM program and other right whale management areas. However, the results of the optimization model and some general results are invariant of the data issues; this includes conclusions regarding aerial surveys, time to implement the DAMs, temporal distribution of landings and fishing effort, the reduced cost of implementing risk reduction measures to the fishing industry under a cost-effectiveness vs. a benefit ranking decision rule and that most importantly, most of the risk was outside right whale management areas for the study time period. The data related issues we encountered continue to exist (i.e., sparse spatial data for northern Maine and the constant monthly right whale density estimates). Still the lack of data (and ensuing uncertainty) does not take away from the utility of doing this modeling as long as the inherent biases and impacts on results are taken into account. Conservation decisions need to be made now, vs. waiting for new or perfect data. Although, characterizing parameter, process and/or model structural uncertainty could improve information for management decisions (Geary et al., 2020).

A critical input parameter to our relative risk index is fishing trip density. The lowest coverage of VTR trips is for the northern Maine port group, which accounts for almost 50% of landings; this could bias our estimates of risk. To address this issue we chose a larger spatial stratification scale (i.e., right whale management area) to avoid overstating the results or giving the end-user an excessive sense of precision or accuracy. Consequently, most of the trips for northern Maine were allocated to the open areas (**Supplementary Table 2**). To investigate how these data could bias our observations about the performance of right whale management areas, we tested excluding northern Maine fishing effort.[27] The test result suggested right whale management areas would have a greater impact on reducing risk (i.e., 12.6% annual reduction vs. 6.5% with northern Maine). However, including the data for northern Maine allows for a more complete estimate of risk and captures the seasonality relationship between whales and lobster fishing effort.

The main focus of our work is to examine fishing effort controls that were in place through 2009 with whale density treated as exogenous. Since the density maps are based on survey effort from 1998 to 2017, they do not reflect risk for any one-time period and we include this critical input parameter as a point estimate. We recognize the empirical estimates of the relative risk index would differ by including the variance of density of whales (and fishing effort). However, this exploratory study was not intended to provide tactical management advice, and in recognition of limited resources and simplicity, we did not incorporate this uncertainty. We expect general observations will not change with this decision, for example, the seasonal distribution of whales (and fishing) given the short period of our analysis, 2002–2009, are unlikely to change.[28] Appreciating the fact that uncertainty in whale density estimates is critical for tactical management decisions, further implementation of this model would need to take this into consideration. For instance, in recent years, there appear to be fairly large shifts in the distribution of whales; developers of the maps plan to create two density maps for separate time periods (Center of Independent Experts, 2019). At this point in time we do not have annual density maps which would reveal inter-annual variability in monthly density estimates, although the variance reported in the current density maps does reflect this to some degree.

In terms of our optimization model structure, the use of large spatial strata allows us to make the assumption that fishing effort disappears when an area is closed. When large fishing areas are closed (i.e., open ocean) this effectively eliminates fishing opportunities for that month as alternative spatial fishing opportunities are limited. Whereas, when smaller dynamic and static areas are closed (**Table 2**), the effect of this shift in fishing effort to the large open area has negligible impacts on relative risk index values. Recognizing that a harvester behavior model is critical for management decisions, a fuller implementation of the framework would require a finer spatial scale and allow uncertainty to be introduced. With a finer spatial

[27]We would like to thank an reviewer for making this suggestion.

[28]The empirical estimates of RREI would differ, if for example, both the number of whales and fishing effort were under-estimated (or over-estimated) in the right whale management areas, then the RREI in those areas would be under-estimated (over-estimated) and the RREI outside would be over-estimated (under-estimated). In the case where fishing effort and number of whales are in the opposite direction with magnitudes unknown, an empirical analysis would be required to address this question.

scale, revenue differences between the two decision rules may decrease, however, it would remain an empirical question. Input parameter estimates of gear efficacy and compliance in our model were provided to demonstrate how they can impact risk reduction targets if not considered. While data can be collected for compliance, methods, such as expert elicitation may be necessary to estimate gear efficacy parameters since data do not currently exist.

Going forward, first, we recommend DAM programs be included for consideration in the suite of policy instruments for protection of large marine mammals, such as right whales. The U.S. DAM program provided a greater level of risk reduction than static management areas despite the small spatial and temporal extent of the zones. However, for dynamic management to be effective, measures must be able to be implemented quickly, responding to real-time information (Hobday and Hartmann, 2006). Lengthy delays between sightings and implementation of United States DAM zones likely reduced protection provided to whales. Yet, mandatory changes can be implemented quickly in some regulatory systems. In Canada for example, within the "dynamic zone" in the Gulf of St. Lawrence, an area is closed on the day a whale is detected and notice through the Fisheries Notice system can require gear removal within 48 h.[29] Alternatively, voluntary dynamic measures may provide protection faster than mandatory measures. Under the voluntary speed reduction zones used to reduce the risk of ship strike mortality in the United States Dynamic Management Areas, a message is immediately sent to an industry contact email list once boundaries have been identified. The United States Coast Guard broadcasts the coordinates to mariners on the NOAA weather radio, and the media is contacted (73 FR 60173, October 10, 2008); this communication speed increases the degree of potential risk reduction from the action. While participation in voluntary measures can be limited (e.g., Silber et al., 2012), high participation has been demonstrated for a noise reduction strategy in Vancouver, Canada (Fraser River Port Authority, 2019) where a reduction in port fees incentivized participation.[30] A growing literature on voluntary approaches is helping to identify the conditions to influence success (Segerson, 2010), and positive incentives or the threat of regulatory measures can influence participation (Bisack and Clay, 2020, 2021). Understanding the conditions necessary for high participation should assist in determining whether incentives should be positive or negative.

Second, we recommend additional targeted research on the economics and norms (i.e., social, cultural, and legitimacy) of key fisheries, both in the United States and Canada, in addition to the ongoing biological research on the species. The issue of right whale entanglement is transboundary and extends along much of the Atlantic coast of both countries and overlaps with numerous fisheries, governance structures, and cultures. A single policy instrument (e.g., modified gear) is unlikely to fully address the issue. Understanding how harvesters' respond to different instruments, such as closures (e.g., Smith and Wilen, 2003; Powers and Abeare, 2009), and the impact of the governance structure on the daily choices of fishermen, will assist in the design of policies that consider the associated incentives and disincentives (Squires and Garcia, 2014). Additional solution pathways may also emerge by simultaneously modeling the economic benefits that can be attained by the lobster fishery under different lobster regulations. This will require an understanding of the yield of the resource stock, costs of harvesting the resource, and market demand models (Richardson and Gates, 1986; Ardini and Lee, 2018), along with the implications of climate change and geographic shifts in lobster stocks (Goode et al., 2019). From a management policy perspective, it is important to understand these relationships to steer clear of implementing measures that could unknowingly shift fishing effort into time-areas with a high density of right whales.

Third, we recommend that the collection of compliance data be included in the design and monitoring requirements for all regulations, even temporary regulations, such as the DAM program. While the Plan included enforcement to deter non-compliance along with outreach and education for compliance promotion (National Oceanic and Atmospheric Administration (NOAA), 2012), data that could be used to estimate compliance were not collected until the program was in its fifth year. Failure of current policy instruments to meet marine mammal management outcome goals cannot always be fixed with new instruments, and it is critical to understand if the goals of a policy instrument are being impeded by non-compliance. When assuming full compliance, the reduction in risk can be overestimated (i.e., scenario 3 vs. scenario 4). Compliance can be improved through enforcement with associated public sector costs, but alternative mechanisms exist. For example, cost-effective management may leave more revenues in the lobster fishery compared to benefit-ranking, which takes the economic needs of the entire lobster industry into account and may increase compliance. As well, incentives can influence choices people make on a daily basis, acting as what behavioral economics calls a "nudge" (Thaler and Sunstein, 2010; Mackay et al., 2018). There is some evidence that compliance increases with at-sea observers, which in the United States do not perform any enforcement function but may provide such a "nudge" to increase compliance (Bisack and Das, 2015; Bisack and Clay, 2020, 2021), although with an increase in public sector costs. However, other tools, such as taxes and subsidies (Squires and Garcia, 2014) and normative factors, such as social influences within a community (Mackay et al., 2018) may also influence compliance choices, suggesting areas for further research.

Fourth, we recommended right whale detection sampling regimes be evaluated for the trade-offs between public and private sector benefits, costs and outcomes. Monitoring of both the species and harvesters is critical to evaluation of success or shortcomings of management measures. Identifying cost-effective resource allocations for right whale detection for management (e.g., flights for protection and/or stock assessments) could improve the performance

[29] As described here: https://www.dfo-mpo.gc.ca/fisheries-peches/commercial-commerciale/atl-arc/narw-bnan/index-eng.html.

[30] See: https://www.portvancouver.com/news-and-media/news/vancouver-fraser-port-authority-expands-noise-reduction-criteria-to-encourage-quieter-waters-for-endangered-whales/.

of right whale protection measures given limited public sector funding envelopes. The outstanding question is, what is the optimal investment in right whale scientific data collection to support recovery? A simulation-based analysis that incorporates uncertainty and a cost-benefit framework of public and private sector trade-offs for right whale protection would allow for an evaluation of the economic value of scientific information for right whale protection. These types of analyses may identify incentives for the private sector to financially support some of scientific data collection costs currently carried by the public sector (Bisack and Magnusson, 2014).

Finally, we recommend that the uncertainty associated with effectiveness of alternative management measures be considered when identifying an optimal mixed of policy instruments within the least-cost mitigation hierarchy framework. This is particularly relevant when there are very different levels of uncertainty associated with the different policy instruments. For example, the effectiveness of reducing risk with avoidance measures is known; if fishing effort is removed completely from an area, the entanglement risk for that area goes to zero (point estimate) with zero uncertainty (variance). However, for minimization measures, such as whale-modified-gear, where information on risk reduction is not available, there is no measure of uncertainty (i.e., variance). Uncertainty around risk can be decomposed into uncertainty around the presence of the species of concern (i.e., right whales), the threat (i.e., different gear configurations), and the mitigation methods (i.e., effectiveness of different policy instrument choices). The precautionary approach advocates a conservative management decision with priority to the resource when there is uncertainty (FAO (Food and Agriculture Organization), 1996; Hildreth et al., 2004). In evaluating the mix of instruments, a benchmark for an acceptable level of uncertainty (variance) around total risk could be established, with the benchmark lowered for endangered species, such as right whales. A lower variance benchmark around risk reductions may result in more avoidance policy instruments rather than minimization policy instruments within the least-cost framework than would be the case otherwise.

CONCLUSION

This case study evaluated the reduction in risk provided by dynamic and static right whale management areas. Strengths identified included the year round operation of the DAM program and its flexibility to provide additional protection to occasional aggregations of right whales beyond the static areas, where they aggregate year after year. The extensive spatial distribution and repeated overlap of the DAM zones may have incentivized adoption of the whale-modified-gear. To improve the future performance of dynamic measures, we recommend the consideration of voluntary measures, additional social science research of relevant fisheries, the collection of compliance data, evaluation of detection sampling regimes, and consideration of the uncertainty within a least-cost mitigation hierarchy framework.

Despite the potential benefit of the DAM program for right whales, it is not possible to provide a comprehensive evaluation of the program due to a lack of linkage to goals and the absence of key information. The goals of the DAM program were wrapped into the larger goals of the Plan and were defined in terms of outcomes for right whales, with the link to the managed entity (i.e., fishers), not clear. The presence of marine mammal and fishing activity need to be jointly identified to manage reductions in entanglement risk and move toward more cost-effective management options. For a full evaluation, public sector cost must also be considered. The implementation of the DAM program required a high degree of sustained effort by the public sector that likely could not have been sustained indefinitely. However, there may have been additional benefits of the program and the circumstances may have changed. The DAM program reduced economic impacts on private sector harvesters in comparison to extensive closures and potentially provided supplemental protection to right whales. The high public cost of the DAM program may have been, in part, due to the era in which it was implemented. Costs today could be lower. Technological advances have yielded additional detection methods (e.g., passive and glider-based acoustics, etc., see Goulette et al., 2021) and strengthened computational methods (e.g., methods to integrate and assimilate data from the multiple detection methods), which can support improved dynamic decision support tools (e.g., Lewison et al., 2015; Welch et al., 2019).

This research provides important insights into the potential for dynamic management to support far-ranging marine protected species, such as right whales and other large cetaceans, while also raising the need for a larger research agenda to consider tradeoffs between the public and private sectors (Lewison et al., 2015). Protection and recovery of transboundary right whales requires a multi-disciplinary and comprehensive consideration of multiple policy instruments, multiple fisheries, multiple detection methods, economic tradeoff considerations, and multiple governance structures (Kitts et al., 2021). Biological, economic or social research in isolation is bound to miss key considerations, and potentially lead away from, rather than toward, second-best policy options. Bringing these components together within predictive, dynamic models of behavior of the harvesters, whales and lobsters could help steer the discussions and provide a solution path that is critical to the protection and recovery of right whales. In closing, conservation requires science-based decisions (Carter et al., 2021), and the simultaneous inclusion of socio-economic objectives in decision-science models, such as the framework presented, will yield cost-effective solutions.

DATA AVAILABILITY STATEMENT

For NOAA-Fisheries data requests, please contact nmfs.gar.data.requests@noaa.gov and for state data please contact the Atlantic Coastal Cooperative Statistics Program (www.accsp.org). Data supporting the conclusions of this manuscript will be made available by the authors, upon request.

AUTHOR CONTRIBUTIONS

KB and GM contributed equally to the analysis and the writing of the manuscript. KB organized the database and performed the analysis. Both authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2021.540966/full#supplementary-material

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REVIEW
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A History of the Tuna-Dolphin Problem: Successes, Failures, and Lessons Learned

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Multispecies aggregations of tuna, dolphins, and seabirds are prevalent and conspicuous in the vast waters of the eastern tropical Pacific and form the basis of a commercial fishery for yellowfin tuna (*Thunnus albacares*) through setting on schools of dolphins, which is among the largest tuna fisheries in the world. Incidental dolphin mortality associated with the development and early years of the fishery was high; by 1993 it was estimated that eastern spinner dolphins (*Stenella longirostris orientalis*) had been reduced to 44% and northeastern offshore spotted dolphins (*S. attenuata attenuata*) to 19% of pre-fishery levels. Efforts to reduce this mortality began at the inception of the fishery and comprised a diverse array of approaches: modifications to fishing methods and fishing gear (backdown, Medina panel, high-intensity floodlights, swimmers to disentangle and release dolphins); U.S. legislation (through the U.S. Marine Mammal Protection Act, MMPA, and subsequent amendments); international agreements (including the International Dolphin Conservation Program that established dolphin mortality limits, and the legally binding multilateral Agreement on the International Dolphin Conservation Program); and economic incentives [notably through establishment of the U.S. dolphin-safe label and positive certification by the Marine Stewardship Council (MSC)]. Together, these bycatch mitigation efforts have been remarkably successful; dolphin mortality due to entanglement as recorded by fisheries observers (hereafter, entanglement mortality) has been reduced by > 99%. Despite this, the degree to which dolphin populations have recovered remains unclear. Multiple lines of evidence indicate that individual dolphins experience multiple sets in their lifetimes and although causality has not been established, research suggests that chase and encirclement might have impacts on dolphins in addition to entanglement mortality. These impacts potentially include increased fetal and/or calf mortality, separation of nursing females and their calves, decreased fecundity, increased predation, disruption of mating and other social systems, and ecological disruption. The strong management emphasis on monitoring entanglement mortality, and the infrastructure necessary to support this monitoring (in particular, 100% observer coverage on large purse-seiners) require funding to the extent that other activities, particularly continued surveys to monitor stock status and clarify the potential influence of other effects of the fishery on dolphin populations, are currently inadequately funded.

Keywords: dolphin bycatch, tuna purse-seine fishery, dolphin safe, eastern tropical Pacific (ETP), spotted dolphin, spinner dolphin, yellowfin tuna

INTRODUCTION

It has been over 60 years since the commercial fishing industry first began to catch tuna in association with dolphins in a vast and remote eastern tropical Pacific Ocean. The incidental mortality of dolphins in this fishery formed the basis for what became known as "the tuna-dolphin problem." Efforts to lower dolphin mortality provide one of the most successful examples to date of interdisciplinary approaches to bycatch mitigation and include modifications to fishing methods and gear, changes in national legislations and international agreements, and the generalized adoption of eco-labeling and marine stewardship certification schemes. Economics, politics, law, policy, and ethics, with regard to the conservation and use of marine resources and the protection of marine mammals, have been deeply intertwined throughout, and this debate has had important echoes in the public opinion and in the media. Underlying the ways to address this issue in its entirety has been science, at the forefront of developing field and analytical methods to collect data to inform bycatch mitigation actions.

The story is long, complex, and multifaceted. The account below is presented in terms of types of bycatch mitigation efforts rather than a chronology of events. These events played out so that they often influenced one another in both constructive and counter-productive ways. For clarity, we provide a chronology in **Table 1**. In the context of bycatch mitigation, the story is mostly of success; since inception of this fishery, associated dolphin mortality has been reduced by more than 99%. And the story is ongoing, and the fishery continues to operate. However, there has been no mechanism in place to monitor the status of the associated dolphin populations for well over a decade, and so the conservation status of affected dolphin stocks remains uncertain.

THE PROBLEM: DOLPHIN BYCATCH IN THE EASTERN TROPICAL PACIFIC YELLOWFIN TUNA PURSE-SEINE FISHERY

The Tuna-Dolphin-Seabird Assemblage

In the waters of the eastern tropical Pacific Ocean (ETP), here defined as extending from 25°N to 10°S latitude and the western coasts of the Americas as far as 150°W longitude, large-bodied yellowfin tuna (*Thunnus albacares*) associate with several species of dolphins: pantropical spotted (*Stenella attenuata*), spinner (*S. longirostris*) and, to a lesser extent, short-beaked common (*Delphinus delphis*) dolphins. These assemblages are often accompanied by multispecies flocks of seabirds (dominated by Procellariidae, Sulidae, and Laridae). The tuna, dolphins, and most of the seabird species are found throughout the tropics and although they associate to a certain degree in other tropical oceans of the world, the prevalence of this assemblage is a conspicuous feature and a hallmark of the ETP. There, habitat of tuna is compressed to the warm and shallow waters of the surface mixed layer by an extensive and hypoxic oxygen-minimum zone, and the association potentially decreases the risk of predation for dolphins and/or tuna (Scott et al., 2012). Seabirds benefit from increased feeding success because dolphins and tuna chase prey to the surface (Au and Pitman, 1986; Ballance et al., 1997).

Development of the Tuna Purse-Seine Fishery and Resulting Dolphin Bycatch

These multispecies aggregations can be large (tons of yellowfin tuna—the target species, and hundreds to thousands of dolphins and seabirds), diverse (Au, 1991), and are highly visible at the ocean surface. Because of this, it is possible to visually locate large schools of tuna by searching for seabird flocks that closely track the schools and/or dolphins at the sea surface. Additionally, the association is strong—tuna remain with the dolphins even when the latter leave a feeding aggregation. The conspicuousness of this association, the reliability and tenacity of the bond between the tuna and dolphins, and the large body size of the tuna in these aggregations (bringing a higher price than smaller tuna) prompted the development of an efficient and lucrative purse-seine fishery for yellowfin tuna that continues to this day, accounting for about 61% of all purse-seine catch of yellowfin tuna in the eastern Pacific Ocean in 2018 (Perrin, 1968; National Research Council, 1992; Inter-American Tropical Tuna Commission [IATTC], 2019). (The remaining ∼39% are captured in ways that do not involve setting on dolphins, primarily by setting on tuna schools that are associated with natural and human-made free-floating objects, and by setting on tuna schools that are not associated with either floating objects or dolphins).

Prior to the development of modern purse seines, tropical tuna were caught one at a time, on hooks, using pole-and-line methods (National Research Council, 1992). The development of durable synthetic netting and a hydraulically driven power-block to haul very large nets (1500–2000 m long and 120–250 m deep) made it possible to deploy purse seines around entire schools of tuna (Gosliner, 1999). In what are known as "dolphin sets," fishermen aboard purse-seine vessels locate schools of tuna by searching for dolphins and seabird flocks. The

TABLE 1 | Timeline of significant events associated with mitigation of dolphin mortality in the yellowfin tuna purse-seine fishery of the eastern tropical Pacific.

1959	Backdown first practiced by U.S. Captain Anton Meizetich
1961	Annual dolphin mortality estimated to be 550,000
1971	U.S. tuna purse-seine captains Harold and Joseph Medina report decrease in dolphin net entanglement and kill associated with use of the Medina panel
1972	Passage of the U.S. MMPA, including requirement for dolphin mortality levels to be reduced to "insignificant levels approaching zero"
1973	Medina panel used by 60–70% of the U.S. tuna purse-seine fleet
1975	95% of dolphins captured in dolphin sets estimated to be released through backdown
1979	IATTC begins dolphin conservation program modeled on U.S. effort
1981	U.S. embargoes Mexican tuna; MMPA amended to: (a) Reduce incidental mortality of marine mammals to levels approaching zero, (b) conduct research on locating and catching yellowfin tuna not associated with incidental take of dolphins
1984	U.S. proportion of total purse-seine fleet practicing dolphin sets drops to 42% (from 75% in mid 1970s); MMPA amended to: (a) prescribe dolphin mortality quotas for U.S. fleet, (b) require comparability in dolphin mortality between U.S. and foreign fleet, (c) direct research to assess dolphin abundance and trends
1986	Annual dolphin mortality estimated to be 133,000; U.S. lifts 1981 embargo against Mexican tuna; Use of high-intensity floodlights during dolphin sets made at night becomes mandatory for the U.S. fleet
1986–1990	U.S. NOAA Fisheries conducts annual research vessel surveys to estimate dolphin abundance and trends, clarify stock structure, and characterize the ecosystem
1987	Undercover video footage from Panamanian yellowfin tuna purse-seine vessel depicting dolphin kill airs on U.S. national television
1988	MMPA amended to: (a) prohibit sundown sets; (b) require 100% observer coverage on U.S. vessels and comparable coverage for the foreign fleet; (c) place restrictions on use of explosives to herd dolphins; (d) establish vessel performance standards; (e) require research to identify alternative methods of catching tuna; (f) establish stock-specific dolphin mortality limits for foreign fleet and metrics for comparability between foreign and U.S. vessels
1990	Three largest tuna canners in U.S. announce they will no longer purchase tuna caught on dolphins; MMPA amended through Dolphin Protection Consumer Information Act thereby establishing the U.S. dolphin-safe label (defined as no sets made on dolphins during the entire trip for which tuna were captured, as verified by a certified observer)
1992	La Jolla Agreement reached thereby establishing the International Dolphin Conservation Program
1993	Eastern spinner and northeastern offshore spotted dolphins declared depleted under the MMPA
1995	U.S. fleet no longer setting on dolphins, thereby achieving zero dolphin mortality; Declaration of Panama established
1997	MMPA amended to establish International Dolphin Conservation Program Act
1998–2000	U.S. NOAA Fisheries conducts annual research vessel surveys to estimate dolphin abundance and trends, clarify stock structure, and characterize the ecosystem
1999	Agreement on the International Dolphin Conservation Program established
2001	Voluntary "AIDCP dolphin-safe label" created by the Parties to the Agreement on the International Dolphin Conservation Program for tuna caught in the eastern Pacific Ocean
2002	U.S. government scientists submit final report to Congress pertaining to research on "whether the intentional deployment on or encirclement of dolphins with purse seine nets is having a significant adverse impact on any depleted dolphin stock in the eastern tropical Pacific Ocean"; U.S. Secretary of Commerce makes "final finding" that the "intentional deployment on or encirclement of dolphins with purse seine nets is not having a significant adverse effect on any depleted dolphin stock in the Eastern Tropical Pacific ocean"; U.S. dolphin-safe definition changed to include tuna caught with dolphins if no dolphins were killed or seriously injured during those sets
2003	U.S. District Court issues hold on the 2002 change in definition of dolphin safe; U.S. NOAA Fisheries conducts research vessel surveys to estimate dolphin abundance and trends, clarify stock structure, and characterize the ecosystem
2004	U.S. District Court requires that the U.S. label definition remain unchanged from initial 1990 definition
2006	U.S. NOAA Fisheries conducts research vessel surveys to estimate dolphin abundance and trends, clarify stock structure, and characterize the ecosystem
2008	Mexico files formal complaint with World Trade Organization against U.S. claiming that the dolphin-safe label creates unfair trade discrimination
2017	Pacific Alliance for Sustainable Tuna (PAST) earns Marine Stewardship Council certification for tuna caught by setting on dolphins in the eastern tropical Pacific
2018	World Trade Organization Appeals Judges find the U.S. dolphin-safe label to be in compliance with international trade regulations
2019	Net canopies and collapses decrease from 22 and 29% of dolphin sets, respectively, in 1986–1.1% for both; Trial dolphin abundance survey conducted, funded by the government of Mexico and PAST

See main text for references. IATTC, Inter-American Tropical Tuna Commission; MMPA, U.S. Marine Mammal Protection Act; NOAA, National Oceanic and Atmospheric Administration.

search methods have evolved over time, from search primarily conducted with high-powered binoculars to search conducted primarily with radar and helicopters (Lennert-Cody et al., 2016). The helicopter is also used to confirm the presence and abundance of tuna once a sighting of birds and/or dolphins has been made. Speedboats are then used to chase the dolphins toward the purse-seine vessel, corral them into the net, and prevent their escape as the net is set around them. Once encircled, the bottom of the net is pursed capturing both the dolphins and the tuna that remain with them during the chase (**Figure 1**)[1].

Bycatch is often assumed to be or explicitly defined as incidental (as opposed to deliberate) capture, and in this context, it could be argued that dolphins should not be considered bycatch in this fishery because they are intentionally captured. However, because the dolphins are not the ultimate target species and because they are released or discarded after capture (see below), in the context of this fishery they are globally recognized as bycatch. Explicitly, it is dolphin mortality and serious injury associated with capture that is the problem (although other potential effects of bycatch on dolphins have been hypothesized, see below). For this reason, we refer to dolphin bycatch mortality or dolphin mortality as the problem that mitigation efforts have been focused on solving. Here, we follow the explicit definition of bycatch as mortality or serious injury (of dolphins) that are captured and discarded (Hall, 1996).

[1] https://www.youtube.com/watch?v=IB96vsn6XPY

Achieving the goal of releasing dolphins alive and retaining tuna in dolphin sets has been challenging because the body size of the two species is so similar. The number of dolphins killed through time has varied dramatically (**Figure 2**). Mortality in the earliest years of the fishery (1960s and 1970s) is not known with precision but without a doubt was very high (hundreds of thousands killed per year; Lo and Smith, 1971) with an estimated peak of 550,000 in 1961 (Smith, 1983). By the late 1970s, due to changes in purse-seine gear, fishing practices, and statutory regulations, mortality dropped significantly, and by 1980 had declined to about 20,000 dolphins per year. For a variety of reasons (see below), mortality increased in the late 1980s and then dropped again; since 2000 mortality has remained low (on the order of a thousand dolphins per year; Inter-American Tropical Tuna Commission [IATTC], 2015, 2019).

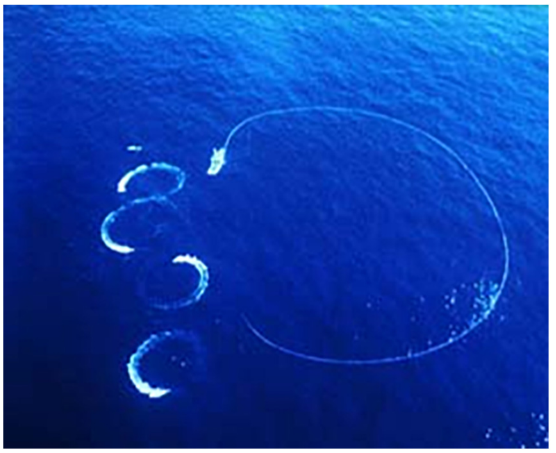

FIGURE 1 | Aerial photograph of a purse-seine set on a school of tuna and dolphins. The purse-seine vessel is deploying the net in a large circle around the entire school while a skiff holds the end of the net in place. In this photograph the net is not yet closed; four speedboats are driving in tight circles near the opening to prevent the dolphins (and tuna) from escaping. Source: Inter-American Tropical Tuna Commission.

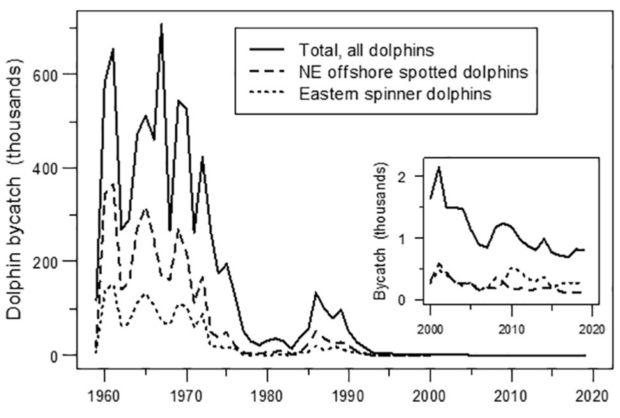

FIGURE 2 | Estimated number of dolphins killed annually in the eastern tropical Pacific tuna purse-seine fishery, total for all dolphins and separately for the stocks of the two dolphin species with the highest number killed (Wade et al., 2007; Inter-American Tropical Tuna Commission [IATTC], 2020). The inset graph has an expanded vertical scale to show details from 2000 to 2019; notice the change of scale on the *Y*-axis.

The Impact of Bycatch Mortality on Dolphin Populations

When the practice of setting on dolphins began, the taxonomy of open-ocean dolphins was poorly known; indeed, the first scientific reports of dolphin mortality came from Bill Perrin, a graduate student who was collecting specimens for a taxonomic study (Perrin, 1968). Beginning in the 1970s, observers were placed on U.S. purse-seine vessels to record dolphin mortality and collect tissue and bone samples. These data made it clear that the dolphins killed in association with this fishery were predominantly pantropical species, that spotted and spinner dolphins were the species most heavily impacted, and that some populations within each of these two species were morphologically distinct from those elsewhere. Scientists now recognize three stocks (management units recognized by the MMPA) that have been of major focus. Two are named subspecies: Coastal spotted dolphin (*S. attenuata graffmani*) and eastern spinner dolphin (*S. longirostris orientalis*), and the third, the northeastern offshore spotted dolphin, is a distinct population within the offshore spotted dolphin subspecies (*S. attenuata attenuata*). The degree of differentiation in distributions, morphology, and genetics between these and other stocks within these two species varies (Leslie et al., 2019, and references therein).

Scientists also know a fair bit about abundance of these dolphin stocks and trends in abundance through time, although estimating these metrics in the vast region that is the ETP is a formidable task and significant knowledge gaps and uncertainty remain. Prior to the start of the tuna purse-seine fishery and through the period of highest mortality in the 1960s, there were no estimates of dolphin abundance. During the 1970s, U.S. government scientists pioneered methods of ship-based line-transect surveys using 25X binoculars to estimate dolphin abundance. Although by the late 1970s it was clear that dolphin mortality was large relative to estimated population sizes, it was not until 1993 that sufficient data had been collected, and analytical methods developed, to estimate that eastern spinner dolphins had been reduced to 44% and northeastern offshore spotted dolphins to 19% of pre-fishery levels (Wade, 1993a,b, respectively). Both stocks were subsequently declared "depleted" under the MMPA (with depletion defined as stock abundance below 60% of carrying capacity).

By the mid-1990s dolphin mortality had declined to levels so low, relative to abundance, that recovery of depleted dolphin stocks was anticipated. However, surveys carried out in 1998–2000 indicated no recovery at the expected rate of 4% (Gerrodette and Forcada, 2005); recovery rates below this would have been difficult to detect given the statistical power of the survey. Modeling found equal support for hypotheses which attributed the lack of recovery to the fishery or to changes in the ecosystem (Wade et al., 2007). The intensity of fishing—it is estimated that every single dolphin of the two primary targeted species was chased and encircled multiple times every year—meant that even a small effect of this activity on dolphin survival or reproduction would be enough to plausibly explain the lack of recovery (Reilly et al., 2005, see below).

Additional surveys conducted in 2003 and 2006 produced higher estimates of eastern spinner and northeastern offshore spotted dolphins, indicating that a recovery might be starting, although the substantial uncertainty surrounding the estimates of abundance meant that the 95% confidence interval on growth rate still included 0 (Gerrodette et al., 2008). Putting the bycatch mortality and abundance estimates together in a population model indicated that the two main affected stocks were indeed increasing (Inter-American Tropical Tuna Commission [IATTC], 2009). There have been no new dolphin abundance estimates since 2006, but reported dolphin mortality has remained low, < 0.1% of population size for each stock.

BYCATCH MITIGATION THROUGH MODIFICATIONS IN FISHING METHODS AND FISHING GEAR

Dolphin mortality was immediately recognized as a problem by the yellowfin tuna purse-seine fishing industry (see below). While concern regarding the impact of this mortality in the context of conservation would not come to light for another decade (simply because so little was known about the magnitude of mortality relative to dolphin population sizes), the time required to extract dolphins from the nets was an immediate problem for fishers. Even as the fleet was still converting vessels from pole-and-line methods to dolphin sets using purse seines, the industry began working to reduce dolphin entanglement in purse-seine nets, thereby decreasing mortality associated with these sets. Subsequent to passage of the MMPA in 1972, U.S. government scientists were also directed to contribute to mitigation of dolphin mortality (see below).

Backdown

"Backdown" refers to a method developed by fishers to release dolphins from the pursed net. It involves running a vessel in reverse after the seine has been pursed and approximately two-thirds of the net brought on board the vessel. This pulls the net into a long and narrow channel with captured dolphins tending to congregate at the far end, at or near the ocean surface. As the vessel continues to move in reverse, the corkline at the far end is pulled underwater, spilling the dolphins out, over the top of the net (**Figure 3**). The tuna tend to remain below the dolphins in a deeper part of the net. This method is believed to have first been applied in the context of dolphin sets in 1959 by the captain of a vessel based in San Diego, California, U.S. Subsequent to further development in 1961, the use of backdown spread rapidly through the San Diego-based fleet, which was conducting the majority of dolphin sets at that time (Barham et al., 1977). Dolphins that do not escape on their own, are hand-hauled over

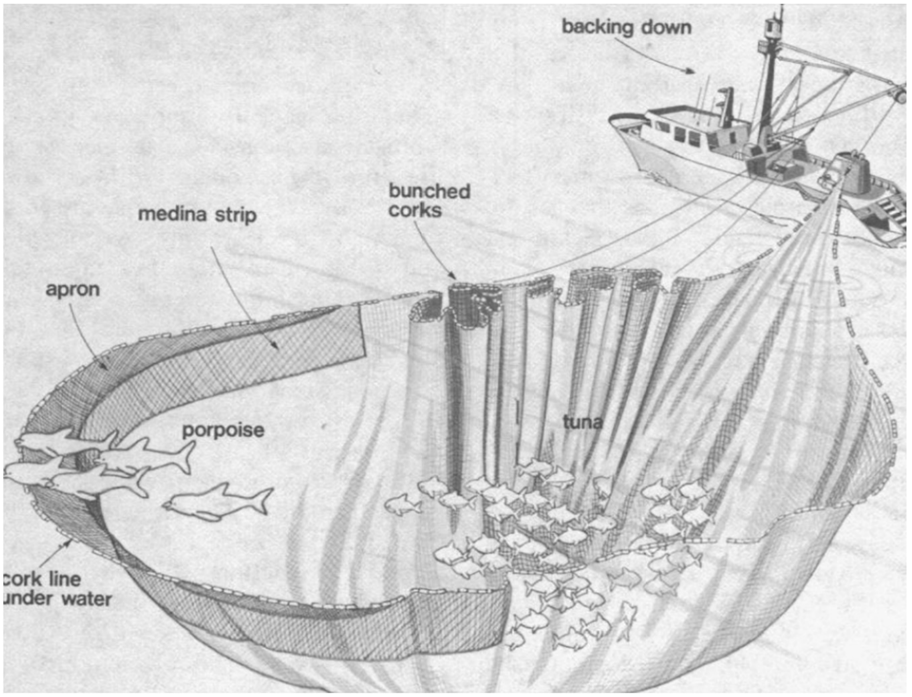

FIGURE 3 | Backdown procedure in progress. As the tuna vessel moves backward to the right in this schematic, the net is drawn into a long channel. The corkline at the far end (left) is pulled under water 1–3 m, and the dolphins (referred to as "porpoise" in this figure, and in the early years of the fishery) escape. The Medina panel (labeled "medina strip" in this figure) and Apron are panels of netting with smaller mesh size to prevent dolphins from becoming entangled as they escape. Source: Leeper, 1976.

the corkline by vessel crew who enter the water or work from a raft inside the net (see below).

Gear workshops for vessel skippers, to share information and refine backdown techniques, were held during the early 1970s and facilitated even more widespread use of the method (see below). By 1975, about 95% of dolphins captured in dolphin sets were being released during backdown, with another 3% released through other methods (Southwest Fisheries Center, 1975 as cited in Gosliner, 1999). Backdown effectiveness and post-backdown rescue were further improved through research conducted by U.S. government scientists in collaboration with the industry (Ralston, 1977 as cited in Gosliner, 1999).

The Medina Panel

The Medina panel was developed to reduce dolphin entanglement in the mesh of the purse-seine. Generally, dolphins avoid contact with the net, but when they do not, their flippers, flukes, and rostra can become entangled, and they drown. During "disaster sets," whole schools of panicked dolphins, including hundreds of individuals, have drowned. This problem was recognized early by the U.S. tuna fleet and following a 1970 meeting of fishing captains, Captain Harold Medina placed a 720 ft wide by 33 ft deep panel in the backdown area with 2-inch mesh netting (instead of the typical 4 1/4-inch mesh; Barham et al., 1977; **Figure 3**). He and his cousin, Captain Joseph Medina, Jr., who modified his vessel's purse-seine net similarly, reported a decrease in dolphin entanglement and mortality during the 1971 fishing season. By the end of the 1972 season, what became known as the "Medina Panel" had voluntarily been installed in 40–50% of U.S. tuna seiners, and in 60–70% by 1973. Subsequent research by U.S. government scientists further improved the effectiveness of the Medina Panel through adjustments in mesh size and development of the "porpoise apron," a trapezoid-shaped panel of small-mesh webbing immediately above the Medina Panel (Barham et al., 1977; Coe et al., 1984; Gosliner, 1999; **Figure 3**).

Since at least the mid-1980s, the Inter-American Tropical Tuna Commission (IATTC) has offered "net alignment" inspections to fishing vessels of the international fleet (Bratten, 1983), which help fishing captains determine the best net configuration to allow them to successfully implement backdown (see below). These inspections, also referred to as "trial" sets, typically involve 1 day at sea during which a staff member of the IATTC onboard the vessel monitors the net position and crew activities during a simulated set, including implementation of the backdown procedure. The IATTC staff member enters the water on a raft inside the net, once the net is pursed, and provides suggestions to the captain regarding net alignment so that the Medina Panel (also known as the "dolphin safety" panel) is properly positioned at the end of the backdown channel.

High-Intensity Floodlights

Although backdown and the Medina Panel greatly reduced dolphin mortality, the absolute number of dolphins killed in purse-seine sets remained high. The effectiveness of these fishing methods and gear modifications depended on a variety of factors including net buoyancy, vessel-specific gear modifications following the Medina Panel model, skill and judgment of the captain and crew, operational characteristics of the purse-seine vessel and skiff, potential fouling of the mesh with planktonic invertebrates, and wind and sea conditions (Perrin, 1969). Likely related to these factors, most dolphin mortality tended to occur during a small number of sets, often when tonnage of tuna or number of dolphins were particularly high (Barham et al., 1977).

One factor directly correlated with dolphin mortality was time of day (Coan et al., 1992). Most dolphin sets were made during daylight (90% in a sample of 20,722 dolphin sets during 1979–1988; Coan et al., 1992) because some daylight is generally required to conduct search. The relatively small proportion of dolphin sets made during the night accounted for a disproportionate number of dolphin deaths and a higher mortality rate (e.g., 10% of 20,722 sets during 1979–1988 accounted for 30% of the dolphin mortality; Coan et al., 1992). This is because restricted visibility at night impairs the ability to control the purse-seine net during backdown. In the early 1980s, the tuna industry began to experiment with the use of high-intensity floodlights to illuminate dolphins in the nets at night, thereby facilitating net control. Dolphin mortality from night sets that used other types of lights, or no lights, was significantly higher than night sets using high-intensity floodlights (Coan et al., 1992). Use of the latter became mandatory for the U.S. tuna fleet in 1986, and subsequently, for non-U.S. vessels when a "sundown set" prohibition came into effect (see below). Fishers must now complete backdown no later than 30 min after sunset.

Use of Swimmers and Divers to Release Dolphins

Dolphins that do not escape from the net during backdown are assisted over the corkline by vessel crew working from a raft within the pursed net. Dolphins can also become entangled in billows of netting ("net canopies") or in areas where sides of the net have come into contact ("net collapses"), contributing to mortality (Lennert-Cody et al., 2004). Divers work from within the net to release these entangled dolphins below the surface. Educational seminars (see below) also provide fishing captains with information on how to avoid net canopies and collapses and have been highly effective. The occurrence of net canopies and net collapses has decreased from 22 and 29% of dolphin sets, respectively, in 1986 to 1.1% for both in 2019 (Inter-American Tropical Tuna Commission [IATTC], 2020).

Fisher Education on Bycatch Mitigation

Following on gear workshops conducted by U.S. scientists, the IATTC has conducted informational seminars for fishing captains since the early 1980s about a range of matters related to setting on dolphin-associated tuna. The scope of the material presented in these seminars initially focused on the use of fishing gear to reduce dolphin mortality (Bratten, 1983) but has expanded over time. With ratification of the Agreement on the International Dolphin Conservation Program (AIDCP) in, 1999 (see below), periodic attendance at these seminars became a requirement for fishing captains to be certified to set on dolphins under this agreement[2]. The purpose of the seminars is to: (1) inform captains about gear requirements (e.g., flood lights, a minimum of three speed boats with towing bridles, and rafts, masks, and snorkels); (2) review factors that contribute to dolphin mortality (e.g., setting in high currents, the dolphin species involved[3], net canopies and collapses, and major equipment malfunctions); (3) review guidelines on actions captains can take to avoid high dolphin mortality; and, (4) provide information on prohibited actions under the AIDCP[4] (e.g., night sets, use of explosives which were historically used to herd dolphins, and other actions leading to infractions against captains and potential removal from the "qualified" list).

BYCATCH MITIGATION THROUGH U.S. LEGISLATION AND INTERNATIONAL AGREEMENTS

Three factors have had a major influence on how U.S. legislation, other national legislations, and international agreements have developed, how they have influenced one another, and how effective they have been with respect to reducing dolphin mortality. First, dolphin sets occur throughout an area that is large and remote, including the Exclusive Economic Zones of ten nations (Mexico, Guatemala, El Salvador, Honduras, Nicaragua, Costa Rica, Panama, Ecuador, Peru, and France) as well as the high seas (**Figure 4**). The sheer size and remoteness of the region, and the multinational nature of the fishery (see below), complicate data collection, regulation, and enforcement. Second, the market for tuna in the U.S. is large and lucrative, and access to this market has been and, in some cases, continues to be a strong draw for the sale of canned tuna products, including those derived from dolphin sets. Third, although the method of setting on tuna associated with dolphins was developed on U.S. vessels, and most of the fleet practicing this method were U.S. vessels in the 1960s and 1970s, very few U.S. vessels have fished this way since the early 1990s, and the fleet using dolphin sets is now comprised of vessels from 9 other nations. The result has been a change in focus on mitigation of dolphin mortality to the international arena.

U.S. Legislation

The magnitude of dolphin mortality in the ETP tuna purse-seine fishery first came to widespread public attention in the U.S. in the mid-1960s. The public outcry over the scale of dolphin mortality was one of the factors that ultimately drove the creation and passage of the MMPA in 1972. From its inception, the MMPA included provisions for reducing dolphin mortality to "insignificant levels approaching zero" after a 2-year moratorium on regulation. During this moratorium, the U.S. tuna industry and U.S. government scientists were expected to solve the mortality problem through development of improved fishing methods. Scientific studies were initiated, observers were placed on fishing boats, fishing gear was inspected, and boat captains with high dolphin mortality rates were reviewed. Nevertheless, mortality in dolphin sets continued to be high, and this prompted a series of amendments to the MMPA beginning in 1981. These amendments reiterated the goal of reducing dolphin mortality rates to levels approaching zero (although economic and technological considerations were allowed[5]) and directed the Secretary of Commerce to undertake or fund research focused on locating and catching yellowfin tuna that did not involve the incidental taking of dolphins ["Take" under the MMPA means "to harass, hunt, capture, or kill, or attempt to harass, hunt, capture, or kill any marine mammal" (16 U.S.C. 1362)].

The 1981 regulations were restrictive enough to contribute to the decision of many U.S. vessels to register under flags of other countries (thereby not subject to U.S. legislation) or to fish for tuna in other geographic regions, using other methods. As U.S. vessels left the fleet, vessels from other countries entered so that the number of vessels using dolphin sets continued to increase (Sakagawa, 1991; DeMaster et al., 1992; Gosliner, 1999). Concern that U.S. gains in lowering dolphin mortality were being offset by increased mortality from non-U.S. vessels prompted the U.S. Congress to enact additional amendments to the MMPA in 1984. These: (1) designated quotas of dolphin mortality for the U.S. tuna fleet that would carry forward in time indefinitely; (2) required that dolphin mortality associated with tuna imports would be comparable with the U.S. fleet and allowed for embargoes on tuna imports that did not comply; and (3) directed research to assess abundance and trends for the affected dolphin populations as a means of revising dolphin mortality quotas if necessary.

Following the 1984 amendments to the MMPA, a previous embargo imposed on Mexican tuna imports was lifted in 1986. However, an increase in dolphin mortality in 1986 to over 100,000 and other failures to lower dolphin mortality prompted additional amendments to the MMPA in 1988. These prohibited U.S. vessels from making "sundown sets," required 100% observer coverage on U.S. vessels (see below), prohibited the use of explosives other than seal bombs to herd dolphins, required research addressing the impact of seal bombs on dolphins, established performance standards pertaining to dolphin mortality rates for vessels and captains, and required that research independent from U.S. government scientists be undertaken to identify alternative methods of catching large yellowfin tuna that did not involve incidental take of marine mammals. The 1988 MMPA amendments also placed additional

[2]https://www.iattc.org/PDFFiles/AIDCP/_English/AIDCP_Maintaining%20qualified%20captain%20list.pdf

[3]Eastern spinner and common dolphins have higher mortality rates than offshore spotted dolphins (Lennert-Cody et al., 2004), related to behavior within the pursed net (Pryor and Norris, 1978).

[4]http://www.iattc.org/PDFFiles/AIDCP/_English/AIDCP.pdf

[5]". . .it shall be the immediate goal that the incidental kill or incidental serious injury of marine mammals permitted in the course of commercial fishing operations be reduced to insignificant levels approaching a zero mortality and serious injury rate; provided that this goal shall be satisfied in the case of the incidental taking of marine mammals in the course of purse seine fishing for yellowfin tuna by a continuation of the application of the best marine mammal safety techniques and equipment that are economically and technologically practicable." https://www.congress.gov/97/statute/STATUTE-95/STATUTE-95-Pg979.pdf

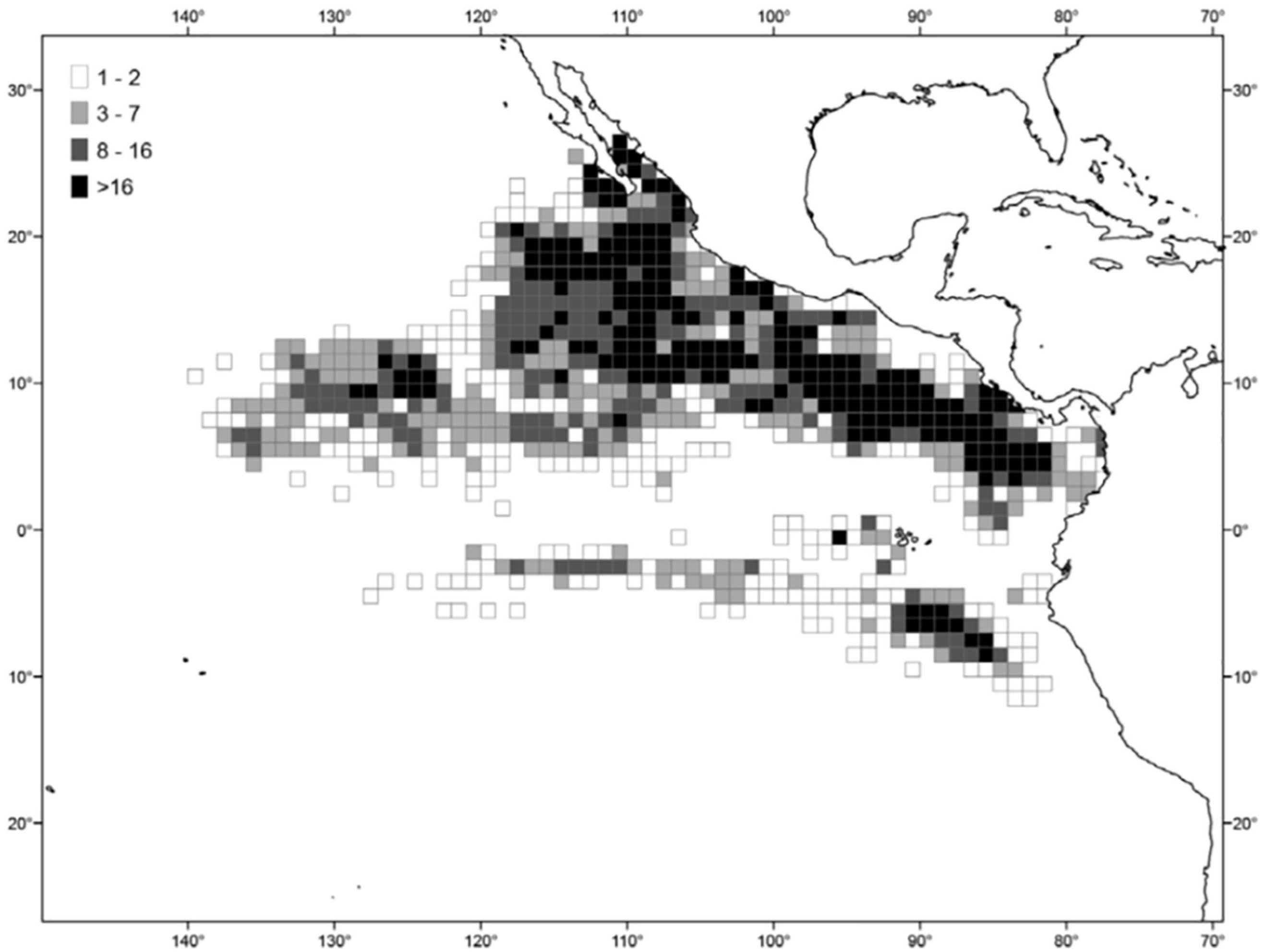

FIGURE 4 | A wide-ranging fishery—distribution of purse-seine sets on dolphins during 2010 in the eastern tropical Pacific; a total of 11,645 sets are shown (Source: Inter-American Tropical Tuna Commission [IATTC], 2015).

requirements on tuna imports by tightening the requirement that dolphin mortality be comparable to that of the U.S. fleet, establishing stock-specific mortality limits, restricting imports from third-party countries, and directing that non-U.S. vessels achieve observer coverage comparable to the U.S. fleet.

The requirement that tuna imports from non-U.S. vessels be regulated the same as tuna caught on U.S. vessels became the focus of much litigation. This was further fueled by graphic and widespread dissemination of video depicting dolphin kills in the fishery and subsequent voluntary actions by U.S. tuna canners to buy only tuna caught using methods other than setting on dolphins. The MMPA was again amended in 1990 through the Dolphin Protection Consumer Information Act (DPCIA) establishing the "dolphin-safe" label (see below). Amendments in 1992 established the International Dolphin Conservation Act. This act provided a mechanism for lifting tuna embargos by the U.S. against other countries, revised dolphin mortality quotas for the U.S. fleet, prohibited intentional sets on depleted eastern spinner and coastal spotted dolphins, prohibited commercial handling of tuna in the U.S. that had been caught on dolphins, and authorized funds for research focused on dolphin-safe methods of locating and catching large yellowfin tuna. More amendments in 1997 established the International Dolphin Conservation Program Act (IDCPA), the U.S. implementation of the International Agreement on the International Dolphin Conservation Program (see below), to which the U.S. is a Party. Among other things, the IDCPA directed scientists of the U.S. National Marine Fisheries Service (NMFS) to conduct research to determine "whether the intentional deployment on or encirclement of dolphins with purse-seine nets is having a significant adverse impact on any depleted dolphin stock in the eastern tropical Pacific Ocean." According to the statute, the answer to this question would determine whether the U.S. Department of Commerce would be allowed to change the definition of "dolphin-safe tuna" under the MMPA to match that definition adopted under the International Dolphin Conservation Program (see below).

International Agreements

As the proportion of U.S. vessels setting on dolphins decreased, and the proportion of non-U.S. vessels increased, concern and focus on monitoring and reducing dolphin mortality became international. The Inter-American Tropical Tuna Commission (IATTC), an international commission responsible for the conservation and management of tuna and other living marine resources in the eastern Pacific Ocean[6], began a dolphin conservation program in 1979, modeled on the U.S. effort. In 1992, in part due to the increasing focus on comparability of dolphin mortality to the U.S. fleet under the MMPA and the dolphin-safe label (see below), fishing countries setting on dolphins agreed to increase observer coverage, institute skipper review panels, and meet a schedule of decreasing dolphin quotas on an individual boat basis. The agreement came to be known as the La Jolla Agreement (Inter-American Tropical Tuna Commission [IATTC], 1993), and it resulted in the establishment of the International Dolphin Conservation Program. A key feature was an allowable limit of dolphin mortality, known as the Dolphin Mortality Limit (DML). Establishing this on a per-vessel basis was a remarkable success. Once a vessel reached its own DML it was required to cease dolphin sets, thereby placing the fate of a vessel in the hands of the captain and crew. Oversight of DMLs occurs through the IATTC. All vessels requesting DMLs receive them; a vessel changing flags retains its DML and its record of dolphin mortality during the year to date, and the new flag state enforces the DML obligations; DMLs are not transferable, rather DMLs from vessels renouncing or forfeiting their assigned limits are redistributed among other vessels although some *ad hoc* transfers among vessels are also allowed (Allen et al., 2010).

The La Jolla Agreement also established an International Review Panel (IRP; Bayliff, 2001), serving as an international forum for reviewing compliance-related matters. In a pioneering move to promote transparency, the IRP included not only governmental representatives of the Parties to the Agreement but also elected representatives of the industry and of environmental non-governmental organizations. The IRP was tasked with the review of cases for which the data collected by a fisheries observer indicate apparent non-compliance by the vessel with the La Jolla Agreement. For example, the IRP reviews cases of apparent use of explosives during any phase of the dolphin set, as well as the timing of the release of the net skiff relative to sundown (possible night sets). Because the IATTC has no enforcement power, disciplinary action associated with any case that is found to be in non-compliance is the responsibility of the government of the vessel's flag state.

The La Jolla Agreement was followed by the Declaration of Panama in 1995, signed by 12 nations, including the U.S. These nations reaffirmed a commitment to reduce dolphin mortality to levels approaching zero and declared their intention to formally establish strict stock-specific DMLs on a per-vessel basis. Dolphin mortality would be verified by fisheries observers, which would be placed on every boat over 363 metric tons (i.e., "large" purse-seine vessels). Although it did not come to pass (see below), some participating nations expected that in exchange for formalization of these binding commitments that would be enshrined in the AIDCP, the U.S. Congress would amend the MMPA to lift the embargoes for tuna caught in compliance with the La Jolla Agreement, allowing access to the U.S. market for all such tuna. The expectation also included a change in the definition of dolphin safe (see below) to include any tuna caught in the ETP in a set in which no dolphins were observed to be killed.

In 1998, features of the La Jolla Agreement and the Declaration of Panama were formally incorporated into a legally binding, multilateral agreement establishing the Agreement on the International Dolphin Conservation Program (AIDCP, Hedley, 2001). The AIDCP has three primary objectives: (1) progressively reduce incidental dolphin mortalities in the tuna purse-seine fishery in the Agreement Area to levels approaching zero, through the setting of annual DMLs; (2) seek ecologically sound means of capturing large yellowfin tuna not in association with dolphins; and (3) ensure the long-term sustainability of the tuna stocks in the Agreement Area, and other marine resources related to this fishery, taking into consideration the interrelationship among species in the ecosystem.

BYCATCH MITIGATION THROUGH ECONOMIC INCENTIVES: ECO-LABELING AND MARINE FISHERIES CERTIFICATION

The U.S. Dolphin-Safe Label

By the late 1980s, 40–50% of purse-seine trips by non-U.S. vessels carried a fisheries observer. This allowed for statistically reliable estimates of dolphin mortality associated with dolphin sets (Joseph, 1994). The numbers provided by IATTC through the latter part of the 1980s were high (**Figure 2**). Graphic depictions of the nature of this mortality were made public when an activist, working undercover aboard a Panamanian purse seiner in 1987, took video footage that was aired on U.S. national television (Brower, 1989). The 1988 amendments to the MMPA (requiring that dolphin mortality associated with tuna imported from other countries be comparable to the U.S. fleet) and ensuing litigation based on claims of non-compliance provided additional incentive for environmental groups to pursue a consumer boycott and to push legislation requiring that tuna be labeled according to the method in which it was caught. This perfect storm of events was followed in April of 1990 by the three largest U.S. tuna canners (Star-Kist, Chicken of the Sea, and Bumble Bee) voluntarily declaring that they would no longer purchase tuna captured in association with dolphins. Shortly thereafter (that same year, 1990), the Dolphin Protection Consumer Information Act (DPCIA) was passed with amendments of the MMPA. It established what we refer to here as the U.S. "dolphin-safe" label, mandating that no sets on dolphins were made during the entire trip for which tuna were captured, as verified by a

[6]The IATTC, one of five regional tuna fisheries management organizations in the world, is responsible for the conservation and management of tuna and other marine resources in the eastern Pacific Ocean, from the coast of the Americas to 150°W, between 50°S–50°N. http://www.iattc.org/.

certified observer. Vessels considered too small to deploy nets around dolphins (fish-carrying capacity $\leq$ 363 metric tons) were exempted (Gosliner, 1999).

A number of dolphin-safe labels then developed in conformity with the MMPA's dolphin-safe labeling definition, and these proved to be powerful marketing tools. Prominent on canned tuna, the labels, combined with environmental organization campaigns to pressure major U.S. retailors, effectively excluded tuna caught on dolphins from an extremely large and lucrative U.S. market. At that time (1990), only non-U.S. vessels were setting on dolphins and the desire to re-enter this market formed the basis for The La Jolla Agreement, The Declaration of Panama, 100% fisheries observer coverage on all large vessels[7], and the establishment of vessel-specific DMLs. Following the Declaration of Panama, the 1997 amendments to the MMPA included a provision for a change in the definition of dolphin safe. But the change was later conditioned by the U.S. Congress on research to determine whether the chase and encirclement of dolphins was having "a significant adverse impact" on dolphin populations. The logic was that if the very act of chasing, encircling, and releasing dolphins was having a negative effect on dolphin populations, it would be misleading to label tuna caught by such methods "dolphin safe." From the perspective of other signatories to the Panama Declaration, these conditions were perceived as a retreat from the commitments made by the U.S., and there were concerns by some that if increased access to the U.S. market was not realized, support for the international agreement that had been negotiated might decline. Given that the vast majority of the fishery was conducted by non-U.S. vessels, the logical extension of these concerns was that failure to change the U.S. label definition might have the ironic result of undermining international efforts to conserve and recover depleted dolphin stocks.

Nevertheless, under direction from the U.S. Congress, research to address whether the chase and encirclement of dolphins was having "a significant adverse impact" on dolphin populations proceeded. This research included estimation of dolphin abundance and trends through time, quantification of ecosystem variability, and studies on potential non-lethal impacts of the fishery on dolphins (Reilly et al., 2005). The research program was developed with the IATTC and the U.S. Marine Mammal Commission (an independent U.S. government agency established through the MMPA and charged with furthering the conservation of marine mammals and their environment). The research methods, results, and conclusions went through extensive peer review. The final research report was submitted to the U.S. Congress in 2002 (Reilly et al., 2005) and concluded that: (1) The two dolphin stocks declared depleted under the MMPA were not recovering at a rate expected given their levels of depletion and the recorded mortality from the fishery[8]; (2) available data were insufficient to clearly resolve the matter of whether or not there had been substantial ecosystem changes in the ETP that would inhibit or enhance these populations' ability to recover; and (3) the fishery may have effects on dolphins at the population level in addition to the reported mortality (see below).

On 31 December 2002, the U.S. Secretary of Commerce made a "final finding" that the "intentional deployment on or encirclement of dolphins with purse-seine nets is not having a significant adverse effect on any depleted dolphin stock in the Eastern Tropical Pacific Ocean." With this, the definition of dolphin safe changed to include tuna caught with dolphins if no dolphins were killed or seriously injured during those sets. The decision was immediately (that same day) challenged by a group of non-governmental organizations that included Earth Island Institute, the Humane Society of the U.S., and the Oceanic Society, and, in April of 2003, the U.S. District Court issued a hold on the change in definition of dolphin safe. Following another year of litigation, the court ordered that no further proceedings on the matter would be allowed due to repeated failures by the Secretary to heed Congress' intent and instructions from previous courts and required that the U.S. label definition remain unchanged (U.S. District Court for the Northern District of California, 2004).

Even prior to these developments, the U.S. dolphin-safe label had already become the subject of tension and international trade disputes that have now spanned three decades. In accordance with the DPCIA and following the establishment of the dolphin-safe label in 1990, the U.S. placed an embargo on tuna from Mexico in February 1991, and subsequently on eleven additional countries. Also in 1991, a three-person dispute resolution panel agreed with Mexico that the U.S. embargo violated the General Agreements on Tariffs and Trade (GATT), an international agreement that limits the use of trade restrictions. Although this decision was never adopted as a formal GATT ruling (Gosliner, 1999), it set the stage for subsequent litigation.

This litigation played out in various hearings and proceedings overseen by the World Trade Organization (WTO), a global international organization dealing with the rules of trade between nations. Various nations (through reserving third-party rights) were involved; the formal complaint was brought to the WTO by Mexico in 2008 with claims that the U.S. violated articles of GATT by creating unfair trade discrimination with the dolphin-safe label. The argument centered on the fact that only tuna caught in the ETP was subject to criteria associated with the dolphin-safe label and that tuna caught elsewhere could use the label without following the strict requirements imposed in the ETP. In this context, the WTO ruled against the U.S. in 2012. Modifications to the labeling policy made

[7]The 100% observer coverage is achieved through a combination of observers of the IATTC and observers of national observer programs (Bayliff, 2001).

[8]However, the NMFS surveys did show slow growth rates in the dolphin populations, albeit statistically non-significant ones, with northeastern offshore spotted and eastern spinner dolphins growing at rates of 1.7 and 1.4% per year, respectively (Reilly et al., 2005). The average of the abundance estimates for the years 1998, 1999, and 2000 were 641,153 (CV1 = 16.9%) for northeastern offshore spotted dolphins, and 448,608 (CV = 22.9%) for eastern spinner dolphins. In a letter to the U.S. Secretary of Commerce, https://iattc.org/PDFFiles/AIDCP/_English/AIDCP_%20Report%20to%20the%20US%20Secretary%20of%20Commerce.pdf the IATTC argued that given the low observed mortality rates and dolphin population sizes in the hundreds of thousands, that the slow recovery observed is what should be expected rather than the higher rates expected by NMFS (INTER-American Tropical Tuna Commission [IATTC], 2002; see also Inter-American Tropical Tuna Commission [IATTC], 2015).

in 2013 strengthened the criteria used to ensure that tuna caught in other regions and sold under the dolphin-safe label was caught without injuring or killing dolphins, but a 2015 ruling by a WTO compliance panel found these changes to be unacceptable. A second change in U.S. policy followed in 2016, but the WTO again ruled against the U.S. in 2017, and authorized Mexico to impose $163M U.S. in trade sanctions annually against the U.S. until the dolphin-safe label complied with international trade laws. The U.S. once again responded with tighter policy and, finally, the WTO found the label to be compliant. At the time of this writing, the definition of dolphin safe under the MMPA remains unchanged (i.e., tuna labeled as dolphin safe were captured using methods other than setting on dolphins).

The Agreement on the International Dolphin Conservation Program Dolphin-Safe Label

In 2001, a voluntary "AIDCP dolphin-safe label" was created by the Parties to the AIDCP for tuna caught in the eastern Pacific Ocean[9]. The AIDCP dolphin-safe label is only available to vessels that have a DML and applies to tuna caught during fishing operations in which no dolphin mortality or serious injury is observed. The vessel's fisheries observer makes the determination regarding whether the catch qualifies as dolphin safe under the AIDCP just before the tuna is brailed and loaded into wells on board the vessel. During any fishing trip, a vessel can catch tuna that qualify as dolphin safe under the AIDCP, and tuna that do not qualify for the label; each type of tuna is stored in separate vessel wells and tracked using forms[10] that follow the tuna from capture to market. Thus, tuna caught in association with dolphins can be certified as dolphin safe through the AIDCP, even though they do not meet the definition of dolphin safe under the U.S. label. Because of this difference between the AIDCP definition of dolphin safe and the U.S definition, tuna products bearing the AIDCP dolphin safe label are not allowed in the U.S. market.

Marine Fisheries Certification

Marine fisheries certifications are programs designed to increase consumer awareness of environmental impacts and sustainability of fisheries. These certifications range from regional to global in scale and impact. Typically, they establish standards for impact and/or sustainability, review fisheries, and provide fishery-specific ratings through lists or ecolabels to better inform consumers and concerned citizens. The number of marine fisheries certification programs is growing as fishing industries, environmental regulators, politicians, economists, biologists, and consumers increasingly recognize the value of promoting the sustainable use of living marine resources, and the influence that these certification programs have on the public at large.

Among the best known of marine fisheries certification programs is the Marine Stewardship Council (MSC). Established in 1996, MSC is an international non-profit organization whose stated mission is to use their ecolabel and fishery certification program to reward sustainable fishing practices and influence consumer choice when buying seafood, thereby transforming the seafood market to a sustainable basis.

The MSC became a catalyst for another effort to obtain broader market accessibility for tuna caught on dolphins in the ETP when the Pacific Alliance for Sustainable Tuna (PAST) earned MSC certification in 2017. PAST was formed in 2014 as an alliance of four companies representing over 90% of the yellowfin and skipjack tuna industry in Mexico. These companies use purse seines in the ETP to capture tuna by setting on schools associated with dolphins, or on unassociated schools ("free schools"). The sustainability assessment was carried out by a third-party certification body and included extensive review by scientists and stakeholder consultation, as is standard practice by MSC. Certification was based on the fact that the fishery adheres to the AIDCP rule that all vessels carry an independent observer to ensure compliance, and that the goal of each set on tuna associated with dolphins is to release all captured dolphins alive. To facilitate the latter, and as already required by the AIDCP, all vessels in the fleet use purse-seine nets with Medina Panels, practice backdown, and carry gear for swimmers should it be necessary to assist dolphins over the corkline of the net.

A final condition of MSC certification for PAST was that it formally commit to a sustainability action plan. This plan consisted of five components (**Figure 5**) and included a commitment to provide significant financial investment in an international research program to conduct a fisheries-independent survey to assess the status of dolphin populations in the ETP. This latter condition was associated with public concern regarding the status of dolphin populations impacted by dolphin sets.

Although some steps have been made toward development of a plan to update assessments of the status of dolphin populations (the previous assessment having been conducted in 2006), adoption of a survey plan by the AIDCP has yet to occur and funding sources for such a plan are yet to be identified. Progress toward assessing stock status includes: a review of available methodology for estimating dolphin abundance, including but not limited to ship-based surveys (Johnson et al., 2018); and development of ship-based survey design options (Oedekoven et al., 2018), which include new methodology to explore the possibility of negative bias in the abundance estimates (Barlow, 2015)[11]. In addition, a trial dolphin survey, funded by the government of Mexico and PAST, was conducted in November 2019 with the goal of testing a survey vessel provided by the government of Mexico and new drone-based survey methodology. The results of this trial survey indicate that the

[9]http://www.iattc.org/PDFFiles/AIDCP/_English/AIDCP_Educational-module-on-the-AIDCP.pdf and http://www.iattc.org/PDFFiles/AIDCP/_English/AIDCP_Dolphin-Safe-certification-system.pdf

[10]http://www.iattc.org/PDFFiles/AIDCP/_English/AIDCP_Tuna-Tracking-System.pdf

[11]Negative bias in the estimates of abundance can result if dolphin schools on the survey trackline are not seen by the search team. Should the results of Barlow (2015) be confirmed with a mark-recapture field study (e.g. Borchers, 2012), this would imply that abundance is greater than has been previously estimated, which could have implications for the determination of stock status.

FIGURE 5 | Components of the sustainability action plan of the Pacific Alliance for Sustainability of Tuna. This plan was a condition of the Marine Stewardship Council's certification of tuna caught by setting on dolphins (downloaded from https://www.pacifictunaalliance.org/sustainability/msc-action-plan.html on 15 May 2020).

survey vessel should perform well for marine mammal surveys and that the double-platform survey protocol involving drones is feasible, but that further testing of drone models and camera equipment will be required (Oedekoven et al., 2021). Although Mexico has expressed the desire to move forward in the future with a full survey, the plans for this survey have not been publicly released.

POSSIBLE EFFECTS OF THE FISHERY IN ADDITION TO ENTANGLEMENT MORTALITY

Tuna and dolphins in the ETP naturally co-occur in large aggregations. Presumably, they derive mutual benefits from co-schooling. Data from tagged individuals show that both tuna and dolphins join and leave these multi-species aggregations on a fluid and daily basis (Scott et al., 2012), but the prevalence of this association in the ETP raises the possibility that disrupting it through dolphin sets and tuna capture may have negative consequences. Additionally, dolphin sets involve a high-speed chase, encirclement and confinement in a net, and release during the backdown process, all of which have the potential to create disruption and stress. Increased fetal and/or calf mortality, separation of nursing females and their calves, decreased fecundity, increased predation, disruption of mating and other social systems, and ecological disruption have all been suggested as possible negative effects associated with setting on dolphins (see below; also, Perryman and Foster, 1980; Au, 1991). Causality between dolphin sets and these effects has not been established, and arguments have been made that they would not be expected to be significant[12]. Additionally, studies based on fishery data require some assumptions which may not be possible to validate. Nonetheless, research has revealed correlations that are consistent with

[12]https://iattc.org/PDFFiles/AIDCP/_English/AIDCP_%20Report%20to%20the%20US%20Secretary%20of%20Commerce.pdf

the hypothesis that chase and encirclement result in negative impacts on dolphins.

For example, in many dolphin schools encircled by purse-seine nets, dependent calves were missing (Archer et al., 2001). Based on an analysis of 77,361 individuals from two spotted dolphin stocks killed in 9,397 sets between 1973 and 1990, 75–95% of lactating females did not have their nursing calves with them (Borchers, 2012). The estimated total "calf deficit" ranged from 10 s to 8300 calves per year and, assuming these dependent calves did not survive separation from their mothers, represented a 14% increase above the number of calves killed as reported by fisheries observers. A possible mechanism to explain this calf deficit is that females with dependent calves are separated during the chase prior to the set of the purse-seine net (Noren and Edwards, 2007). In bottlenose dolphins, the normal echelon swimming position of a calf is energetically beneficial to a calf, but costly to a mother (Noren, 2008, 2013; Noren et al., 2008). Because the chase is a fast-moving, chaotic environment (National Research Council, 1992), it may be difficult for mothers and calves to maintain their normal swimming positions. As well, there are multiple points during the fishing process when calves could be separated from the mothers and not recorded as observed mortality, should they die later as a result of the separation (Archer et al., 2001).

Negative relationships between fishing activity and dolphin reproductive rates have also been documented by several studies. Based on tissue samples collected by observers, Perrin and Henderson (1984) compared reproductive rates and ages at sexual maturity in eastern spinner dolphins among areas with different amounts of dolphin fishing, and Barlow (1985) and Chivers and Myrick (1993) did the same for northeastern offshore spotted dolphins. Their expectations were that the more heavily fished, and therefore more depleted, dolphin populations would show density-dependent responses, with higher reproductive rates and younger ages at sexual maturity. In fact, they found the opposite, negative relationships between fishing activity and metrics associated with dolphin reproductive rates. Perrin and Mesnick (2003) quantified that sexual dimorphism was high and testis size low for eastern spinner dolphins relative to other populations of this species, indicating a polygynous mating system. They concluded that social disruption of eastern spinner dolphin schools associated with chase and encirclement in the purse-seine fishery could negatively impact reproductive output, especially if dominant males were removed from schools. In an analysis of photographs of entire schools of spotted and spinner dolphins taken from a research vessel-based helicopter from 1987 through 2003, Cramer et al. (2008) found an inverse correlation between the annual proportion of calves in a school (a proxy for reproductive rate) and the annual number of purse seine sets for spotted dolphins (but not spinners). They also found an inverse correlation between the length of calves at independence (a proxy for duration of nursing) and the annual number of purse seine sets, again for spotted dolphins but not spinners. Finally, Kellar et al. (2013) analyzed hormone levels from 212 skin and blubber biopsy samples from female spotted dolphins collected between 1998 and 2003. They found that the proportion of pregnant females in a school was negatively related to an index of fishing activity nearby in space and time. They also found that recent exposure to purse seine sets was significantly lower for pregnant as compared with non-pregnant females.

The degree to which these effects may have population-level consequences is associated with the degree to which the fishery interacts with dolphins individually, and at the population level. Evidence that individual dolphins experience multiple sets in their lifetimes dates at least to the 1970s. A 1976 research cruise designed to refine fishing methods and gear, also incorporated behavioral studies of dolphins. This research indicated that dolphins may learn from exposure to dolphin sets, as evidenced by apparent hiding underwater in response to an approaching purse-seiner, avoiding encirclement through maneuvers that made it difficult to herd them into the net, and once in the net, congregating away from the vessel and net walls, and moving to the apex of the net before backdown (Pryor and Norris, 1978). More recent research has shown that evasive behavior of dolphins has increased over time and was strongest where fishing was most intense (Lennert-Cody and Scott, 2005). Reilly et al. (2005) used mean values from 1998 to 2000 and estimated that there were over 5,000 sets on northeastern offshore spotted dolphins per year, resulting in 6.8 million dolphins chased and 2.0 million dolphins encircled in purse-seine nets annually. For eastern spinner dolphins, the numbers were about 2,500 sets per year, resulting in 2.5 million dolphins chased, and 300,000 dolphins captured annually. When divided by the mean estimated abundances during the same years, a northeastern offshore spotted dolphin was chased 10.6 times and captured 3.2 times per year on average, and an eastern spinner dolphin chased 5.6 and captured 0.7 times per year.

DISCUSSION: REMAINING CHALLENGES AND LESSONS LEARNED

Remaining Challenges

Absolute abundance of dolphins has featured prominently in the context of evaluating the impact of the fishery and establishing international and U.S. management schemes. In particular, absolute abundance estimates have been used in population dynamics models to evaluate dolphin stock status, and to determine per-stock per-year mortality limits (AIDCP Annex III)[13]. Historically, these estimates have been based on fisheries-independent surveys conducted by NOAA, but the most recent of these was conducted in 2006 (see below). Previous attempts to develop indices of relative abundance from fisheries-dependent observer data (Buckland and Anganuzzi, 1988; Lennert-Cody et al., 2016) have proven problematic because of non-random distribution of search effort relative to dolphin abundance and time-varying biases associated with changes in fisher search behavior. It is unlikely that other methods of assessing stock status, such as close-kin genetics, will be available in the near

[13]The per-stock per-year mortality limit for each stock is based on the lower bound of the confidence interval on abundance (Barlow et al., 1995; Inter-American Tropical Tuna Commission [IATTC], 2006). Should the annual mortality exceed the limit for a stock, all sets on that stock, and on any mixed-species dolphin schools that contain that stock, are prohibited for that year.

future (Johnson et al., 2018). It is, therefore, generally agreed that there is an ongoing need for fisheries-independent surveys to estimate absolute abundance of dolphins.

Conducting fisheries-independent surveys requires significant funding; the lack of such funding largely explains the long time since the most recent survey. The cost of a single (1 year), two-vessel survey comparable to those conducted by NOAA and incorporating drone-related methodological improvements has been estimated at US$11M–$15M (Oedekoven et al., 2018). Recommendations have been made to conduct back-to-back surveys over several years to obtain pooled estimates of abundance with greater precision than that of single-year estimates (Oedekoven et al., 2018), which would greatly increase costs. In-kind contributions, particularly for vessels, could potentially reduce survey costs by roughly US$3M per vessel, but this could leave the survey schedule vulnerable to the fiscal status and internal research priorities of a few countries.

Ultimately, a stable plan for long-term funding of fisheries-independent dolphin surveys is needed. Critical to its development is a thorough review of the benefits of current bycatch mitigation measures relative to their costs. An evaluation of trade-offs associated with maintaining 100% observer coverage on large purse-seine vessels (currently requiring on the order of U.S. $1.6M annually) would be particularly insightful. Quantitative analysis of the level of observer coverage necessary to estimate total fleet bycatch with a specified precision, absent a substantial observer effect (vessels following all protocols to minimize dolphin mortality only when an observer is present), would allow for informed dialogue among all stakeholders regarding tradeoffs between costs, goals, and resources. Related is the possibility of data collection by Electronic Monitoring Systems (EMS) as a means of evaluating bycatch mitigation efforts on vessels/trips that do not carry a human observer. Although bycatch enumeration is not currently possible with EMS, EMS appear capable of collecting data on some operational aspects of dolphin sets, including the start time of chase and backdown, and the presence of net canopies, net collapses and high-mortality sets (Román et al., 2020). Finally, a review should also consider the need for other data types. For example, life history data are important for estimating age distributions and reproductive rates, but these data have not been collected since the mid-1990s (Scott et al., 2018). In the absence of absolute abundance estimates, stock mortality limits have been calculated from projections of absolute abundance obtained from population dynamics models. These models would likely benefit from biological data that represent the current population.

The strong emphasis currently placed on bycatch mitigation, with a goal to reduce it to near zero, presents an interesting philosophical, and perhaps practical issue. This emphasis, and reliance on observers for enforcement, may distract from other, perhaps equally important issues. These include development of a long-term data base of biological information which might be used to help monitor stock status, more deeply investigating potential effects of the fishery other than entanglement mortality, and conducting research on the ecological impact of separating co-schooling tuna and dolphins through the purse-seine setting process.

Lessons Learned

The six decades of catching tuna by setting on dolphins, and the multidisciplinary efforts to mitigate dolphin mortality associated with this fishery have been filled with successes and failures. We find a number of clear themes in these successes and failures that may provide transferable lessons to bycatch mitigation efforts in general.

First, the most significant successes in decreasing bycatch can be attributed to modifications in fishing gear and fishing practices (see Squires et al., 2021). Most of these were implemented by the fishery itself almost as soon as the practice of setting on dolphins began, and were continually improved through time, albeit supported and improved by scientists associated with the U.S. government and IATTC.

Second, placing the fate of a vessel in its own hands has been a powerful incentive to effect reduction of bycatch mortality. In particular, the establishment of DMLs proved to be remarkably successful because these limits were implemented on a vessel-specific basis, thereby rewarding each vessel captain for reducing dolphin mortality with the opportunity to continue to set on dolphins.

Third, impacts of a fishery on non-target species may extend beyond entanglement mortality. Rich data and rigorous science show strong correlations between dolphin sets and increased fetal and/or calf mortality, and decreased fecundity of dolphins associated with these sets.

Fourth, science has been a powerful ally, but also an excuse for inaction. Research has guided modifications to fishing gear and fishing practices that have lowered dolphin mortality; field and analytical methods have provided a means to assess dolphin abundance and trends; data indicating the potential for effects of the fishery in addition to entanglement mortality have provided plausible explanations for the apparent slowed recovery of dolphin populations. Yet the scientific process is not quick and not certain. For example, by the time science provided abundance estimates that resulted in a formal listing of depleted under the MMPA, the most significant reductions in dolphin mortality had long since occurred. And despite unprecedented effort (Kaschner et al., 2012), fishery-independent abundance and trend estimates are still associated with high levels of uncertainty. The lesson is that timely management action could require decisions to be made in the face of indicative, but less-than-conclusive, data. It is in this context that the Precautionary Principle is relevant (e.g., Kriebel et al., 2001).

Fifth, unilateral regulation is often inadequate if it does not reflect, and is not consistent with, a multilateral one. This is obvious in the case of the ETP yellowfin tuna purse-seine fishery that is practiced by multiple nations in multiple jurisdictional regions, and for which the multilateral regulatory framework is that developed and implemented under the AIDCP and, for the IATTC, the 2003 Antigua Convention. Additionally, the global nature of trade means that even in cases where a fishery is more regional, unilateral regulation that would be adequate in that context may still be inadequate in a broader one. Related are differences between nations in culture and institutions that can lead to significant misunderstanding. For example, legal and institutional orders differ between member nations of the IATTC, and in the context of dolphin-safe, many of these nations could not fully understand how the U.S. Congress and courts could override agreements that had been previously negotiated in multilateral settings. For its part, the U.S. did not fully appreciate the subsequent sense of being let down that was felt by their negotiating partners who also considered that a formal international commitment had been breached.

And finally, extraction of marine living resources, and incidental impacts associated with that extraction, occurs even in the most remote parts of the world's oceans. This is a given; sustainability in resource extraction, including maintaining healthy marine ecosystems, should be the goal. This is certainly not a new concept; we choose to emphasize and support it here.

AUTHOR CONTRIBUTIONS

TG synthesized existing data from multiple sources to construct **Figure 2**. CL-C provided fact-checking pertaining to the fishery and international policy. LB wrote the first and synthesized subsequent drafts of the manuscript. All authors contributed to conception of the manuscript, wrote sections, contributed to revision, read, and approved the submitted version.

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REVIEW
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Marine Mammal Interactions With Fisheries: Review of Research and Management Trends Across Commercial and Small-Scale Fisheries

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Marine mammal interactions with fisheries, such as bycatch and depredation, are a common occurrence across commercial and small-scale fisheries. We conducted a systematic review to assess the management responses to marine mammal interactions with fisheries. We analyzed literature between 1995 and 2021 to measure research trends in studies on direct and indirect interactions for: (i) high and low to middle-income countries, (ii) fishery operations (commercial and small-scale), and (iii) taxonomic groups. Management responses were categorized using the framework described previously in peer-reviewed studies. Marine mammal bycatch remains a major conservation concern, followed by marine mammal depredation of fishing gear. A high proportion of studies concentrated on commercial fisheries in high-income countries, with an increase in small-scale fisheries in low to middle-income countries between 1999 and 2020. The insufficient understanding of the social dimensions of interactions and the inevitable uncertainties concerning animal and human behaviors are major challenges to effective management. Despite the key role of human behavior and socioeconomics, we found only eight articles that incorporate human dimensions in the management context. Integrating social dimensions of marine mammal interactions with fisheries could help in setting pragmatic conservation priorities based on enhanced understanding of critical knowledge gaps. An area-specific adaptive management framework could be an effective tool in reducing the risk to marine mammals from fisheries by coupling technical solutions with socio-economic and political interventions. We conclude that despite the vast body of literature on this subject, a "silver bullet" management solution to marine mammal interactions with fisheries does not yet exist.

Keywords: bycatch, depredation, trophic interactions, competition, direct interactions, commercial fisheries, small-scale fisheries, marine mammals

INTRODUCTION

As a result of resource overlap, interactions between marine mammals and fisheries are a common occurrence (Trites et al., 1997; Liu et al., 2019). A wide spectrum of these interactions has been documented, from the mutually beneficial to the detrimental. Marine mammals have taken advantage of human activities, particularly, fishing, for the nutritional gain (Bearzi, 2002; Allen et al., 2014). In several parts of the world, "co-operative" or "associative" fishing between marine mammal species and fishing communities has led to mutually beneficial interactions, in that both the fishermen and the animals find easy target and forage fish, respectively (Northridge and Hofman, 1999; Neil, 2002; Peterson et al., 2008; Kumar et al., 2012; D'Lima et al., 2014). On the other end of the spectrum, interactions such as depredation and bycatch negatively affect and alter marine social-ecological systems (Tixier et al., 2021; **Box 1**). Depredation affects both marine mammal populations and fishery socioeconomics due to gear damage and catch loss, by creating artificial resource provisioning due to the introduction of novel prey resources, leading to higher chances of incidental entanglements (Werner et al., 2015). Bycatch is a major threat to the conservation and recovery of marine mammal population(s) worldwide (Avila et al., 2018) leading to the declines of a wide range of species such as dugongs (*Dugong dugon*) (Marsh et al., 2011), monk seals (*Monachus monachus*) (Woodley and Lavigne, 1991; Guüçluüsoy et al., 2004), and Hector's dolphins (*Cephalorhyncus hectori*) (Slooten and Dawson, 2010), amongst others. The recent "human caused" extinction of the Baiji (*Lipotes vexillifer*) (Turvey et al., 2007), and the expected extinction of the vaquita (*Phocaena sinus*) (Rojas-Bracho et al., 2006; Jaramillo-Legorreta et al., 2007; Morzaria-Luna et al., 2012) have further added to the growing concerns of the impact of fisheries on marine mammals.

The large-scale capture of pantropical spotted (*Stenella attenuata*), spinner (*S. longirostris*) and common (*Delphinus delphis*) dolphins (Gosliner, 1999; Wade et al., 2007) in the Easter Tropical Pacific (ETP) tuna fishery was instrumental in the recognition of bycatch as a global threat to marine mammal populations and the subsequent development and implementation of the United States Marine Mammal Protection Act (MMPA) in 1972 (Gerrodette, 2009). Since then, large-scale commercial operations using gear such as gillnets, purse-seines, hook and line, and trawlers have been extensively reviewed to develop management strategies to reduce their impacts on marine mammals (Gilman et al., 2007; Tixier et al., 2021). Several aspects of these interactions, particularly, marine mammal entanglement in fisheries have been widely studied (Read et al., 2006; Moore et al., 2010; Reeves et al., 2013), especially: (i) well-documented bycatch in gillnets, trawls, longlines, purse-seines, pots and traps, etc.; (ii) poorly documented bycatch in artisanal or small-scale fisheries in developing countries, and (iii) the transition of incidental marine mammal catch in fishing gear from being discarded to gaining market value (Read, 2008). This increased understanding has provided the impetus for the development and implementation of marine mammal population recovery and maintenance measures (Perrin et al., 1994).

Despite this documentation, the management and mitigation of these interactions is a pressing concern (Anderson et al., 2020; Hines et al., 2020). The k-selected life-history traits of marine mammals: long-life spans coupled with relatively late sexual maturity and low reproductive rates, severely limit conservation efforts (Brown et al., 2014; Mannocci et al., 2014).

In their review of a decade of bycatch management in commercial fisheries, Dawson et al. (2013) categorized management measures into three main technical approaches to increase the understanding of their efficacy in attaining management goals: (i) strategies that change human behavior, either mandated or voluntary: e.g., spatial or place-based management, monetary incentives, etc.; (ii) strategies that change the nature of interactions: e.g., technological interventions like bycatch reduction devices in trawl fisheries, and gear modifications such as changes in hook design in longline fisheries; and (iii) strategies that change animal behavior: e.g., acoustic deterrent devices, acoustic harassment devices. These management strategies were developed and evaluated in the context of commercial fishery operations. The high economic and social costs of management, such as, heavy reliance on expensive technology, marginalization of fishing communities, inadequate governance and enforcement, limit their wider implementation in small-scale fisheries (Brotons et al., 2008; Mangel et al., 2013; Brownell et al., 2019) but these aspects were not explicitly considered in the Dawson et al. (2013) typology.

An alternative multidisciplinary framework for bycatch mitigation has recently been proposed by Squires et al. (2021). This "Bycatch mitigation hierarchy framework" offers a systematic order of actions to manage and mitigate bycatch that considers the human elements of bycatch reduction. This framework comprises four basic approaches: (1) private solutions, including voluntary, moral suasion, and intrinsic motivation; (2) direct or "command-and-control" regulation starting from the fishery management authority down to the vessel; (3) incentive- or market-based approach to alter producer and consumer behavior and decision-making; and (4) a hybrid of direct and incentive-based regulation through liability laws.

Managing marine mammal interactions with commercial and small-scale fisheries requires deep understanding of their social, economic, and cultural linkages to fishers' livelihoods (Carvalho et al., 2011). However, the current definitions of fishery operations are relatively simplistic in the context of management (**Box 2**). For small-scale fisheries in particular, these definitions are nuanced, and vary spatially (Teh and Pauly, 2018). Therefore, finding an appropriate balance between the ecological and human dimensions of marine mammal interactions with fisheries is a major challenge (D'Lima et al., 2014).

Despite the acknowledgment of these challenges, the long-term efficacy of management measures implemented so far, is seldom considered by stakeholders in the overall conservation management process. This ambiguity leads to a general lack of focus and consensus on the critical role of place-based management measures to mitigate these interactions. The overall aim of this review, therefore, is to understand the scope of research on marine mammal interactions with fisheries and discuss the implications of the management measures aimed at

BOX 1 | Marine mammal interactions with fisheries: Categories and definitions of different types of interactions with commercial and small-scale fisheries.
Based on their effects on marine mammal species and fishery socioeconomics, the interactions between fishing operations and marine mammals have been categorized as "direct" or "indirect" (adapted from Beverton, 1985; Hall, 1996; DeMaster et al., 2001; Read, 2008).

Direct interactions occur when marine mammals come into direct or close contact with fishing gear (Read, 2008). These interactions include *bycatch,* accidental entanglements in fishing gear and depredation (Silva et al., 2002; Read et al., 2006; Gerrodette, 2009).
Bycatch is defined as the unintended capture of marine biota in fishing gear during an operation targeting a different species (Gray and Kennelly, 2018). Bycatch in fishing gear is a persistent threat for many marine mammal species (Read, 2008).
Depredation is also a direct interaction where marine mammals remove or damage fish (Tixier et al., 2019), leading to catch loss and gear damage for fishers. Depredation is recorded for several coastal and offshore odontocete species, over a range of fishery operations (Bearzi et al., 2019), for example, gillnet depredation by bottlenose dolphins (*Tursiops truncatus*) (Ayadi et al., 2013; Rechimont et al., 2018) and that of longline gear by sperm whales (*Physeter macrocephalus*) and killer whales (*Orcinus orca*) (Werner et al., 2015; Towers et al., 2019).
Indirect interactions arise due to fishery-induced ecological changes and resource competitions (i.e., habitat and prey overlap between fisheries and marine mammals) (Plagányi and Butterworth, 2009). Over the past five decades, marine fishery operations have advanced technologically, and fishery production has increased as a result. These changes have resulted either in gross overfishing or local reductions in the biomass of the target species that marine mammals depend on for prey (DeMaster et al., 2001), therefore influencing species compositions of marine communities, particularly top predators like marine mammals. For example, reduced prey availability due to overfishing has led to local declines in striped dolphin (*Stenella coeruleoalba*) population(s) in the Mediterranean Sea (Aguilar, 2000) and harbor seal (*Phoca vitulina*) population(s) in the Aleutian Islands (Plagányi and Butterworth, 2009).

BOX 2 | Definitions of fishery operations based on scale, and social and economic factors.
Fisheries are classified as "small" or "commercial" (also industrial) based on their overall size, presumed technological differences, capital investment and market, areas of operations, and production output (Béné, 2006; Pauly, 2009).

Commercial fisheries include industrial or large-scale operations with substantial technological and capital investments. These operations generally supply fish to international as well as domestic markets and use specific types of fishing gear, for example, trawlers, longlines, gillnets (drift, bottom-set), and purse seines [Food and Agriculture Organization, 2001–2016,-]. As a result of their high profit margins and international supply chains, commercial fishery statistics are available in national and international reports, particularly the data on catch characteristics, fishing activities and demography (Christensen and Pauly, 2004; Pauly, 2006).
Small-scale fisheries employ about 90% of the human population in low-income countries and provide food security to over 45% of the human population worldwide (Chuenpagdee et al., 2006; Batista et al., 2014; Zeller et al., 2015; Fisher et al., 2018). Based on their market volumes, small-scale fisheries are classified as commercial or non-commercial (also subsistence) small-scale fisheries (Gillett et al., 2001). Commercial small-scale fisheries are those that cater to local or regional markets, whereas non-commercial or subsistence fisheries provide food security locally and to marginalized socio-economic classes (Pauly, 1997; Zeller et al., 2015). Small-scale fisheries are highly diverse, in that they target multiple species using a variety of gears. This diversity severely limits a comprehensive accounting of these operations in national policies (Gillett and Lightfoot, 2001).

reducing the impacts of these interactions, particularly in the context of small-scale fisheries.

MATERIALS AND METHODS

We used two bibliographic databases, SCOPUS, and Web of Science to identify a range of peer-reviewed articles, published and unpublished reports, and conference papers related to primary research on marine mammal interactions with fisheries. We did not use Google Scholar as it is not recommended for systematic or scoping reviews due to its unreliability in identifying articles specific to the keywords (Haddaway et al., 2015).

After initial deletions based on duplicates and relevance to the search terms, the list was reduced to 784 articles. Of these, 489 articles were excluded by screening the titles and abstracts for relevance to key words (**Supplementary Material**). Further stringent inclusion and exclusion criteria were applied to the remaining articles, based on the overall aims of the review, i.e., to understand the overall research trends in marine mammal interactions with fisheries. Review articles, fisheries management *per se*, marine mammal behavior and ecology studies, that were not directly related to interactions, were excluded from the review.

Relevant additional articles and gray literature (e.g., reports, conference, and meeting proceedings, etc.) were mined through the available reference lists of articles and were analyzed based on the above criteria. Limited or poor bibliographic information, however, is a major challenge in the accessibility of gray literature. Several reports and proceedings were inaccessible using search engines like Google Scholar, Elsevier, Scopus, etc., or through university libraries.

Thematic analyses were conducted on the final list of manuscripts (271 articles: published articles 263, gray literature: 8 documents) to understand the general trends in research, based on the year of publication, region where research was conducted, type of fishery operation (commercial or small-scale fishery, **Box 1**), and taxonomic group (cetacean, pinniped, sirenian, etc.). The literature was then segregated into two main emergent themes related to the interactions between marine mammals and fisheries (**Box 2**): (A) Direct interactions: mainly, (i) bycatch, and (ii) depredation, and (B) Indirect interactions: (i) trophic interactions, to compare trends over time across fishery operations and, to understand which of these interactions are the most reported in the literature.

Further, management strategies and the challenges faced to mitigate marine mammal interactions with fisheries were collated and discussed in accordance with the technical categories described by Dawson et al. (2013) (see Introduction) because we considered these categories appropriate for the current literature.

There are two limitations to our study: (1) the literature reviewed is a non-random sample from the available literature driven by the key-words we used in our search; (2) our

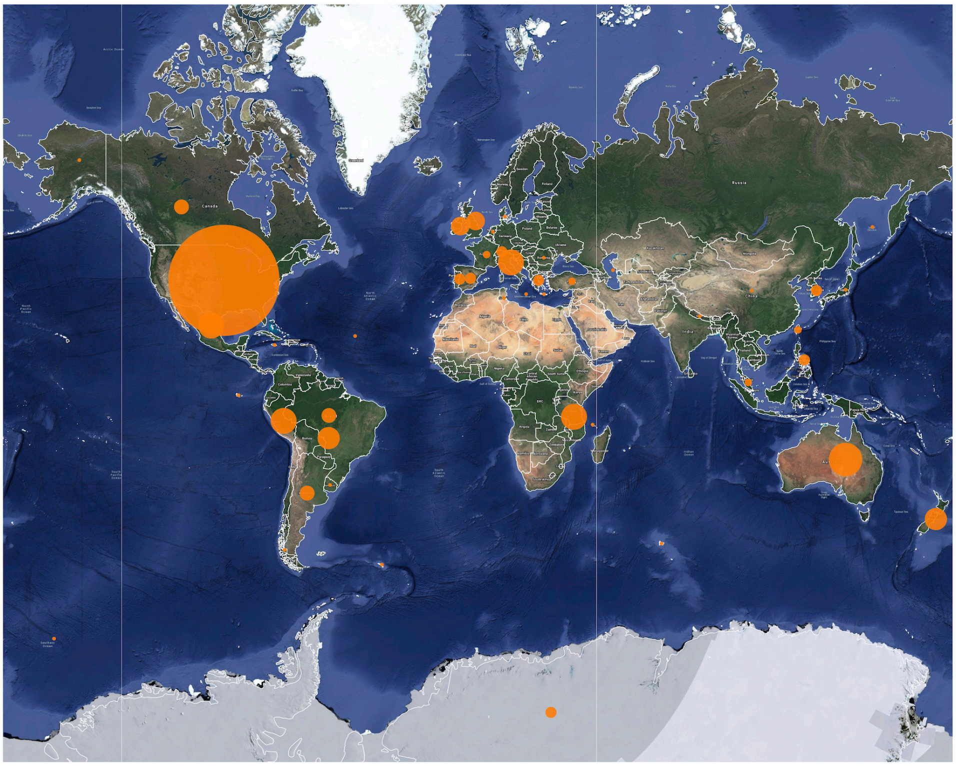

FIGURE 1 | Geographic distribution of the publications reviewed in this study. The size of the orange circles across the map depicts the number of publications in the given field site where studies were conducted, and/or from regions from where data for those studies originated.

analyses are based on the results described in this literature and are not a statistical comparison between experimental or control studies.

RESULTS

We first document the global trends in research on marine mammal interactions across commercial and small-scale fisheries, and various fishing gears. We then present the thematic analyses of research on direct and indirect interactions, followed by a discussion of the observed management responses and challenges across fishery operations and interactions.

Global Research Trends Across Commercial and Small-Scale Fisheries

There was a high degree of consistency (90%) between the bibliographic databases, SCOPUS and Web of Science. In the final analyses (271 articles), research was mainly focused on marine mammal interactions with commercial fisheries particularly in high income countries (108 articles). This trend is more pronounced in the early 1990's, in high-income countries such as United States, New Zealand, Great Britain, Ireland, Spain, France, Greece, and Australia. Studies on commercial fisheries in low-middle income countries were not so common (15 articles). The proportion of studies on small-scale fisheries in low to middle-income countries has increased since the mid-2000's (124 articles: high-income countries: 25, low to middle-income countries: 99) (**Figures 1**, **2**). Cetaceans remain the prime focus of research followed by pinnipeds and sirenians (263 articles where taxa were recorded: 189 cetaceans, 66 pinnipeds, 8 sirenians, 8 marine megafauna, 14 marine mammals in general) (**Figure 3**).

Research Trends Across Fishing Gears

Marine mammal interactions with fisheries also vary across gear types in both commercial and small-scale fisheries. For both commercial and small-scale fisheries, marine mammal interactions were the reported most commonly in gillnets (27 and 79 published studies respectively). These variations are further described for each interaction type below in section "Thematic Analyses of Marine Mammal Interactions With Fisheries."

Thematic Analyses of Marine Mammal Interactions With Fisheries

These analyses reflect the number of studies and trends in these studies across both commercial and small-scale fisheries. Bycatch remains the main area of research, with 187 studies, followed by depredation with 56 studies. Trophic interactions between marine mammal and fisheries have attracted research attention since the late 2000's, particularly, studies on prey consumption by marine mammals and their dietary overlaps with fisheries, as an aid to understand the potential risks from fisheries to

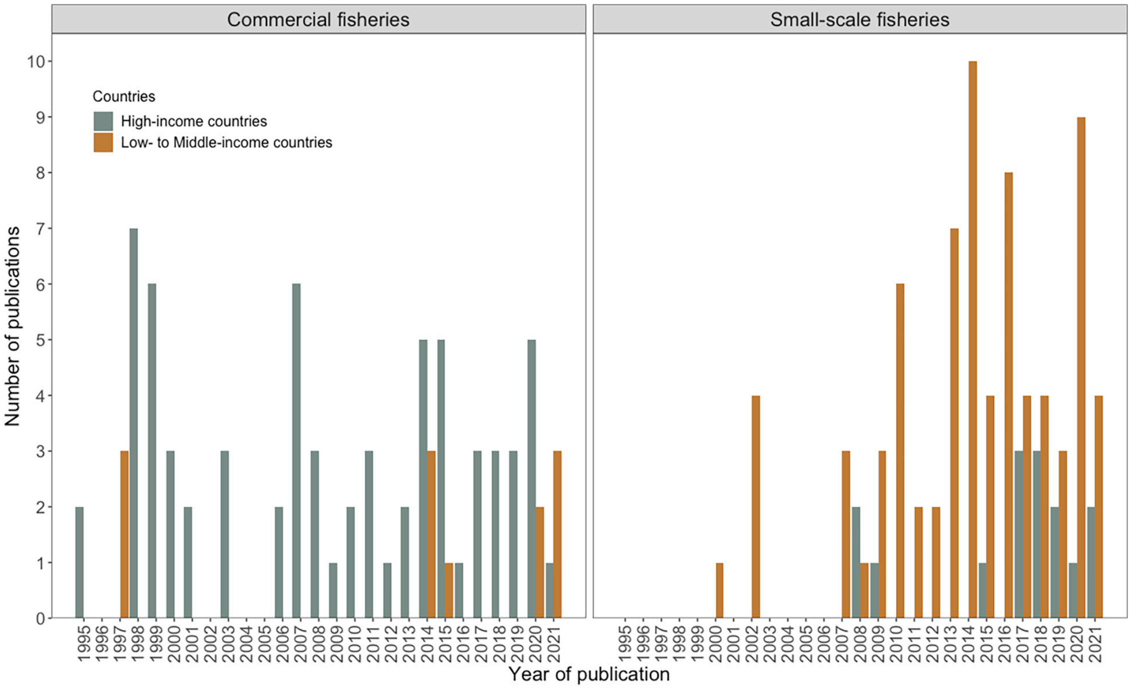

FIGURE 2 | Trends in number of published studies over time (1995–2020) across commercial and small-scale fishery operations in high-income and low to middle income countries. These data are segregated by commercial and small-scale fishery operations, *gray* signifying high income countries and orange, low to middle income countries.

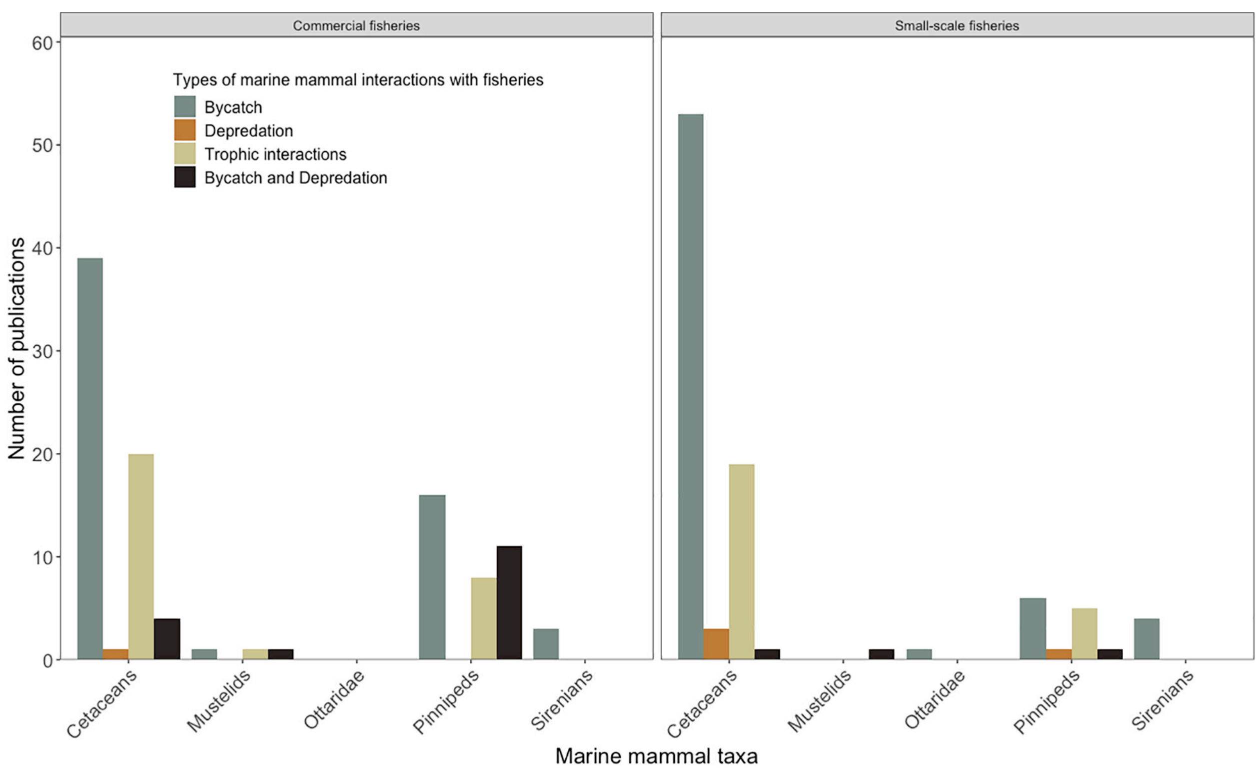

FIGURE 3 | Trends in number of published studies of marine mammal interactions with fisheries across marine mammal taxa (cetaceans, mustelids, pinnipeds and sirenians) across commercial and small-scale fisheries, and, bycatch, depredation, and trophic interactions).

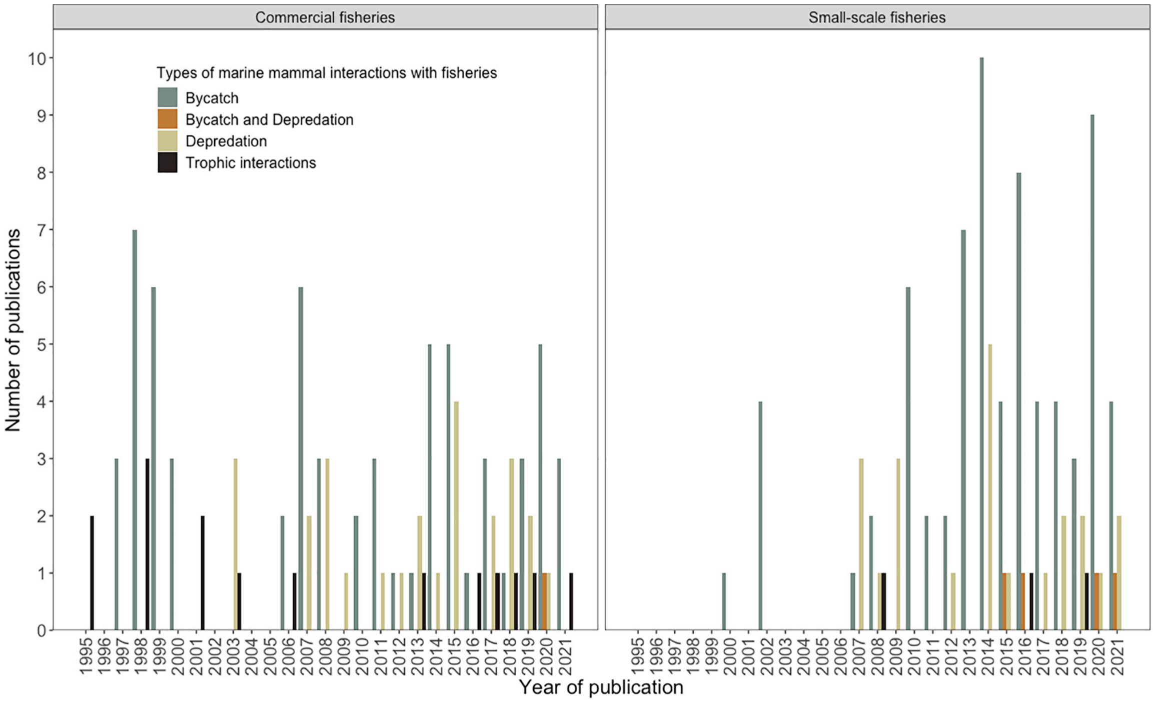

FIGURE 4 | Trends over time (1995–2020) in number of published studies on bycatch, depredation, and trophic interactions, across commercial and small-scale fisheries.

mammal population(s) and vice versa (22 studies) (**Figure 4**). These themes are described in detail below.

Direct Interactions: Bycatch

Between 1997 and 2021, 164 publications documented the scale of marine mammal bycatch, including fishery and gear-specific bycatch estimates (Julian and Beeson, 1998; Allen, 2000; D'Agrosa et al., 2000; Ortega-Argueta et al., 2005). Marine mammal bycatch is spatially variable, with larger bycatch estimates coinciding with areas of higher marine mammals or fish (prey species) abundance (Lewison et al., 2004, 2014; de Godoy et al., 2020; Baird et al., 2021), reflecting the increasing resource overlap between marine mammals and fishery operations. In many regions, spatial, temporal, and oceanographic factors affect bycatch. For example, management area and season were not associated with bycatch in Australian trawl fisheries (Hamer et al., 2012; Allen et al., 2014), whereas, in the gillnet fisheries of Peru, bycatch was higher in certain geographic locations, but was not associated with seasons (Majluf et al., 2002; Mangel et al., 2010; Ayala et al., 2019). Bycatch estimates also vary across different gear types. Bycatch estimates for gillnets are the highest in both commercial and small-scale fisheries (29 and 63 published studies, respectively), compared with other gear types such as longlines, trawlers, and purse seines (**Figure 5**).

Direct Interactions: Depredation

Fifty-six studies examined depredation, in both commercial (28 studies) and small-scale fisheries (28 studies). The impacts of depredation vary across fisheries and marine mammal populations. For some gear, depredation commonly leads to both gear damage and catch loss, in other gears, economic losses to fisheries are uncommon. For example, for commercial longline fisheries, significant reduction in catch rates occurred due to sperm whale and killer whale depredation (Tixier et al., 2017). In areas where overfishing has caused stock declines, these losses may compound social-economic costs, leading to the implementation of retaliatory measures like intentional shooting or hunting marine mammals, allegedly to protect livelihoods (Gilman et al., 2007; Lauriano et al., 2009).

For commercial fisheries, longlines have the highest proportion of reported marine mammal depredation with gear damage and catch loss (19 studies) (Hernandez-Milian et al., 2008; Huang, 2011; Towers et al., 2019). Depredation of other commercial gear such as purse seines, gillnets, and trawlers, is relatively less common (Goldsworthy et al., 2001; Hall et al., 2013). Gillnets are a still a cause for concern in small-scale fisheries (20 studies), with a higher proportion of reported marine mammal depredation than any other gear (Bordino et al., 2002; Rechimont et al., 2018). In small-scale fisheries, gear damage and catch loss were reported for most gear types, irrespective of the level of reported depredation (Bearzi et al., 2011; Monaco et al., 2019; **Figure 4**).

Cetacean depredation of commercial fishing gear has been reported from species such as bottlenose dolphins (*Tursiops spp.*) (Paudel et al., 2016; Bayless et al., 2017; Wild et al., 2017; Rechimont et al., 2018; Revuelta et al., 2018), killer whales (*Orcinus orca*) and sperm whales (*Physeter macrocephalus*) (Peterson et al., 2013, 2014;

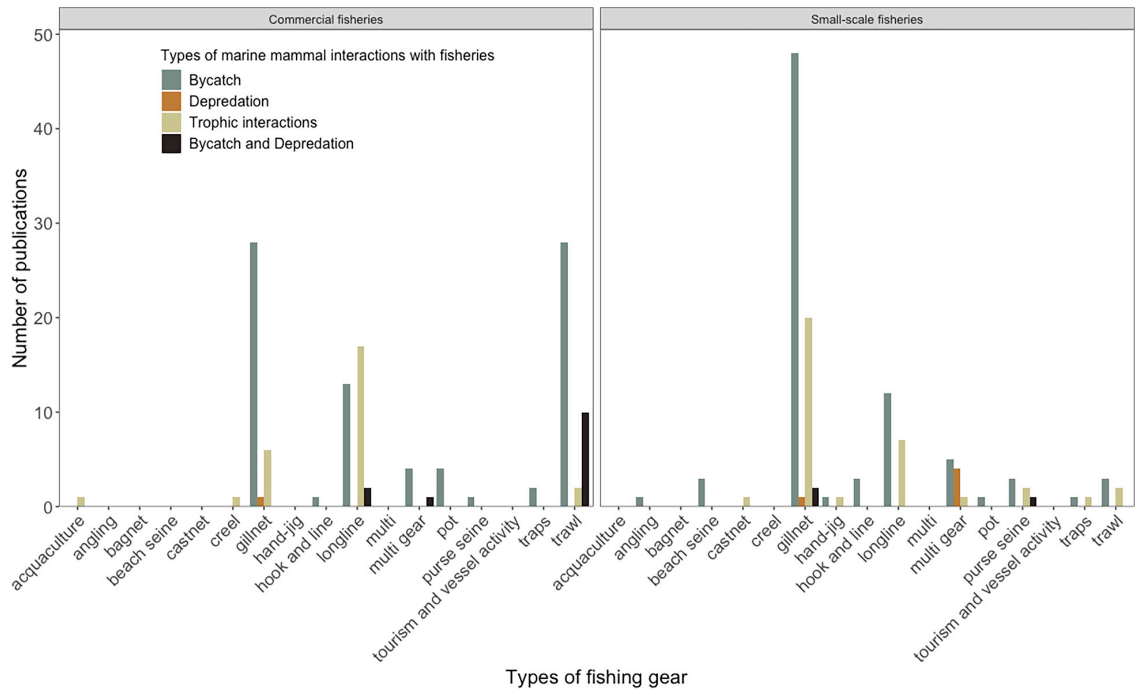

FIGURE 5 | Trends in the number of published articles on marine mammal interactions with fisheries (Bycatch, Depredation and Trophic interactions), across commercial and small-scale fishing gears.

O'Connell et al., 2015; Thode et al., 2015; Bayless et al., 2017; Hanselman et al., 2018; Richard et al., 2018; Tixier et al., 2019; Towers et al., 2019), short-finned pilot whales (*Globicephala macrorhyncus*) and false killer whales (*Pseudorca crassidens*) (Hernandez-Milian et al., 2008; Rabearisoa et al., 2015; Bayless et al., 2017). Pinniped depredation has been documented for gray (*Halichoerus grypus*) (Moore, 2003), fur (*Arctocephalus gazelle, A. australis*) (Croll and Tershy, 1998; Goldsworthy et al., 2001; Bombau and Szteren, 2017), harbor (*Phoca vitulina*) (Moore, 2003; Rafferty et al., 2012) and elephant seals (*Mirounga spp.*) (Green et al., 1998; van den Hoff et al., 2017), and South American sea lions (*Otaria flavescens*) (Sepulveda et al., 2007; de la Torriente et al., 2010; de Maria et al., 2014).

Species such as bottlenose dolphins, Boto (*Inia geoffrensis*), Risso's dolphin (*Grampus griseus*), striped (*Stenella coeruleoalba*), spinner (*Stenella longirostris*) and common dolphins (*Delphinus spp.*) have been reported depredating small-scale gear (Lauriano et al., 2009; Cruz et al., 2014; Mintzer et al., 2015). Pinniped depredation of small-scale fisheries is less frequently reported than for commercial fisheries, possibly reflecting the variability in the distributions of small-scale fisheries and pinnipeds (Panagopoulou et al., 2017; Sepulveda et al., 2018). Small-scale fisheries occur mainly in the tropics whereas pinnipeds mostly occur at higher latitudes (Chuenpagdee et al., 2006).

Indirect Interactions: Trophic Interactions Between Marine Mammals and Fisheries

Twenty-two studies discussed predator overlap with fishery operations and prey species to understand the prey requirements of top predators and the potential threats posed to both fisheries and animals (Kellert et al., 1995; Croll and Tershy, 1998). The overlap between the diet of the alleged marine mammal predator and the target species of commercial fisheries, such as longlines, gillnets and trawlers was described in 14 articles. Two articles quantified this overlap, highlighting marine mammal predation pressures on fishery resources and competition with fisheries (Li et al., 2010; Reinaldo et al., 2016). Diet overlap for both pinnipeds and cetaceans with small-scale fisheries were documented but not quantified.

The pressures exerted by fisheries on marine mammal energy intake and predation are orders of magnitude higher than the effect of marine mammal predation on the fish catch of commercial and small-scale fisheries (Croll and Tershy, 1998; Weinstein et al., 2017). An indirect effect of this trophic competition also leads to the shift of marine mammal diet from higher to lower trophic level species. Little is known about how this shift manifests across marine mammal populations. Nonetheless, it is clear that overfishing, rather than marine mammal predation has resulted in fishery stock declines that in turn affect marine mammal population(s) (Goldsworthy et al., 2001; Etnier and Fowler, 2010).

Management Response to Marine Mammal Interactions With Fisheries

Of the reviewed literature, 129 studies described and discussed management responses to various types of interactions observed (commercial fisheries: 63 studies, small-scale fisheries: 56 studies, both commercial and small-scale fisheries: 10 studies) (**Table 1**). The management responses and recommendations observed are categorized below using the framework described by Dawson et al. (2013). This framework mainly demonstrates the variations in management responses in commercial fisheries. We adapted this framework in this literature review to highlight the general challenges and opportunities in management response, in both commercial and small-scale fisheries.

Management Response Categories

Measures That Aim to Change Human Behavior

This strategy includes spatial or place-based conservation measures and, modifications in fishing practices, or a combination of both, usually to mitigate marine mammal bycatch and incidental injuries or entanglements.

Place-based conservation measures, such as, Marine Protected Areas, have been implemented since the 1970's (Hoyt, 2011). Permanent or temporary fishing closures (di Sciara et al., 2016) are a common conservation approach for habitats and species impacted by fishing activity. In response to the spatial and temporal variations in marine mammal bycatch and depredation events, selective fishing practices, particularly, in recognized sensitive habitats, and/or seasonal variations in fishing practices have also been implemented (Lauriano et al., 2009; Hanselman et al., 2018). The efficacy of these attempted solutions has rarely been rigorously tested.

Monetary compensation (Güçlüsoy, 2008) and subsidies on more selective gear and mitigation devices (Monaco et al., 2019) have also been implemented to offset the cost of gear damage and catch loss to fishers caused by depredation or incidental entanglements. Levies and subsidies on commercial fisheries, based on allotted quotas, have been introduced to encourage adequate reporting, and reduction of marine mammal bycatch (Bisack and Das, 2015). Developing alternative sources of income to deter the use of fishing gear(s) most prone to interactions, have also been suggested (Majluf et al., 2002). Alternative livelihoods such as eco-tourism have shown partial success in mitigating these interactions through poverty alleviation in certain regions of the world (Ermolin and Svolkinas, 2018; Berninsone et al., 2020).

Technological Interventions to Change the Nature of Interactions

Technological interventions, mainly for commercial fisheries, include bycatch reduction devices and modifications to gear that reduce bycatch rates, facilitating the safe escape of animals caught in fishing gear, or to reduce the instances of depredation (Allen et al., 2014). Gear modifications such as bottom-set gillnets illuminated with LED lights (Bielli et al., 2020), and depredation mitigation devices (Rabearisoa et al., 2015) can reduce cetacean, sea turtle, and sea-bird bycatch. Modifications to longlines have been extensively trialed. Hook modifications (McLellan et al., 2015; Hamilton and Baker, 2019), gangions (O'Connell et al., 2015) and decoys (Wild et al., 2017), have

TABLE 1 | Number of publications describing management strategies across marine mammal interactions with fisheries for both direct (bycatch, depredation) and indirect (trophic) interactions and fishery operations (commercial and small-scale fisheries); as per the management categories described by Dawson et al. (2013).

Management strategies	Marine mammal interactions with fisheries							
	Bycatch			Depredation		Trophic interactions		
	Commercial fisheries	Small-scale fisheries	Fisheries in general	Commercial fisheries	Small-scale fisheries	Commercial fisheries	Small-scale fisheries	Fisheries in general
Changes in human behavior	26	27	10	21	18	8	1	0
Changes in animal behavior	2	1	0	1	4	0	0	0
Changes in the nature of interactions	1	0	0	0	0	0	0	0
Combination of above categories	6	5	2	3	4	0	0	1

significantly reduced the rate of killer whale and sperm whale depredation in longline gear.

Several other technological modifications to fishing gear, particularly gillnets, have been developed, and reviewed (Harwood and Hembree, 1987; Trippel et al., 2003; Uhlmann and Broadhurst, 2015; Hamilton and Baker, 2019). However, these modifications are generalist in their approach and often of limited value to small-scale fisheries because of the spatial, temporal, and cultural variations in fishery operations (Teh et al., 2015; Davies et al., 2018).

Measures That Aim to Change Animal Behavior

This approach includes technological interventions that help to deter marine mammals from fishing gear. Active sound emitting devices, mainly acoustic deterrent devices, are used for bycatch mitigation, and acoustic harassment devices have been tested to reduce the instances of depredation (Reeves et al., 2001) on both commercial and small-scale fisheries.

Acoustic deterrent devices or pingers, actively emit mid to high frequency signals (2.5 to 10 kHz) at a low intensity ($<$ 150 dB, 1 μPa at 1 m) that "deter" marine mammals from approaching fishing gear. Pingers have been shown to reduce the bycatch of bottlenose dolphins (Cox et al., 2004), harbor porpoises (*Phocaena phocaeana*) and Franciscana (*Pontoporia blainvillei*) (Dawson et al., 2013; Mangel et al., 2013; Chladek et al., 2020) in gillnet fisheries. Pingers have also proven successful in reducing pinniped interactions with aquaculture operations and have reduced the bycatch of some (but not all) species of cetaceans in gillnets (Clay et al., 2019).

Acoustic harassment devices are relatively high output sound emitters ($>$185 dB) primarily used to deter pinnipeds from mariculture or aquaculture operations by causing discomfort to the animals (Quick et al., 2004). Concerns about the effect of depredation on catch rates have led to their widespread use in commercial fisheries and aquaculture (Dawson et al., 2013).

CHALLENGES, UNCERTAINTIES, AND OPPORTUNITIES FOR MANAGEMENT ACTIONS

Based on the intensity of interactions, or their effects on marine mammal populations or fishery operations, management approaches aim to achieve set goals or conservation priorities. The success or failure of these goals can be recognized by certain indicators, for example, increased recruitment in and/or mortality reduction of the target population(s), and improved usage of the area by stakeholders, etc. (Hoyt, 2011; di Sciara et al., 2016). All efficacy metrics are dependent on prior information about the system under consideration, such as: (i) baseline population or abundance estimates, (ii) mortality records of the target species, (iii) stakeholder usage of the area, and (iv) the scale of the fishery operation. However, there are inevitable uncertainties regarding data on marine mammal population(s), fishery socioeconomics, and both human and animal behavioral ecology that limit the assessment of management efficacy in several ways (**Table 2**).

First, evaluating the success of place-based conservation measures and technological interventions is challenging because of the uncertainties associated with prior information on marine mammal abundance and identification of strategic habitats. In many instances, information on marine mammal abundance is inadequate, and data on most species are restricted to stranding records or from fisheries landing centers (IUCN, 2019). For example, for coastal species like the Indian Ocean humpback dolphin (*Sousa plumbea*), which is widely distributed around the peninsular Indian coastline, population estimates are fragmented, and most of the information on bycatch comes from incidental strandings and is thus mainly anecdotal (Sutaria et al., 2015; Braulik et al., 2017).

Second, robust and long-term population estimates require significant financial investment over extended periods. Such information is difficult to obtain, especially in developing countries (Moore et al., 2010; Lewison et al., 2014). In these scenarios without prior information, designing place-based measures poses a challenge to setting realistic goals and evaluating their success in the long run (di Sciara et al., 2016).

Third, bycatch projections for commercial fishery operations are based on existing bycatch estimates which are derived from fishery-observer surveys, and compiling and collating logbook data (Julian and Beeson, 1998; Morizur et al., 1999; Norman, 2000; Majluf et al., 2002; Underwood et al., 2008; Kindt-Larsen et al., 2016). While observer surveys have been an invaluable source of data (Edwards and Perrin, 1993; Morgan et al., 2002), accurately projecting these estimates over entire fishing fleets is not a straightforward process since observer coverage is often inadequate and the presence of an observer can influence fishing practices (Curtis and Carretta, 2020).

Small-scale fisheries present additional challenges with respect to bycatch monitoring, because of their spatial and operational variability, and unregulated, unstructured working environment (Hines et al., 2020). Methods such as rapid survey interviews, spatial risk assessments and monitoring catch at landing centers have provided baseline bycatch estimates for small-scale fisheries in developing countries (Pilcher et al., 2017; Temple et al., 2019; Hines et al., 2020; Verutes et al., 2020). However, fragmented monitoring efforts in small scale fisheries and the resultant inconsistencies in or the lack of bycatch data hamper the successful implementation of mitigation measures in small-scale fisheries (Gilman et al., 2010; Teh et al., 2015).

Data on marine mammal space use and behavioral observations, in the context of fishery operations, are being used for spatial risk assessments, understanding the drivers of bycatch, and assessing the effectiveness of implemented management measures (Grech et al., 2008; Marsh et al., 2011; Cerutti-Pereyra et al., 2020). Studies on acoustic activity and behavior of marine mammals have also assisted in understanding the precursors and the intensities of these interactions (Iriarte and Marmontel, 2013; Lewison et al., 2014; Kindt-Larsen et al., 2016; Lopes et al., 2016; Clay et al., 2019). Over the past decade, researchers have stressed the importance of ecological studies to understand the underlying causes of bycatch (Northridge et al., 2017).

TABLE 2 | Management actions and recommendations for direct (bycatch, depredation) and indirect (trophic) interactions, with commercial and small-scale fisheries. [Commercial fisheries (CF), Small-scale fisheries (SSF); No management action (N), as per the management categories described by Dawson et al. (2013)].

Management actions and other recommendations for reducing marine mammal interactions with fisheries			
	Types of interactions		
Management categories	**Bycatch**	**Depredation**	**Trophic interactions**
Changes in human behavior	• Spatial management: gear bans, protected areas, seasonal changes in fishing and gear use (CF, SSF) • Levies and subsidies (CF, SSF) • Monitoring bycatch (CF, SSF) • Alternate livelihoods (SSF)	• Compensations and subsidies (SSF) • Spatial management Marine Protected Areas (CF, SSF) • Fishery management (CF)	• Food Web Modeling • Ecological data • Ecosystem Based Fishery Management • Animal and prey abundance studies • Diet studies
Changes in animal behavior	• Visual Deterrent Devices (CF) • Acoustic Deterrent Devices (Pingers) (CF, SSF)	• Acoustic Deterrent Devices (Pingers) (CF, SSF) • Acoustic Harassment Devices (CF)	N
Changes in the nature of interactions	• Spatial management: gear modifications (CF, SSF)	N	N
Combination of two or more of the above three categories	Spatial management + Acoustic Deterrent Devices (CF) • Remote Electronic monitoring (CF, SSF) • Penalties for non-compliance with spatial management or acoustic deterrent devices (CF)	• Bycatch reduction devices, • gear modifications, acoustic deterrent devices (CF) • Gear modifications, Acoustic deterrent devices (SSF)	• Compensations, multi-species management strategies, incorporating human dimensions

Marine debris and the transition of bycatch from discard to commodity are two emerging concerns documented in the available literature. Since the mid to late-2010, data from stranding records and carcass examinations has been used to augment bycatch estimates to evaluate the risks due to all anthropogenic activities, mainly fisheries, and to understand the fate of bycaught animals. Analyses of injuries on carcasses have revealed significant number of entanglements of marine mammals in marine debris from small-scale fisheries, indicating interactions with fishing gear or ghost nets (Kaiser et al., 1996; Franco-Trecu et al., 2017).

The transition of bycatch from discard to commodity is also a growing concern in the light of fishery resource depletions (Ermolin and Svolkinas, 2018). Incidental catches create opportunities for food procurement, as declining fish catches widen the gap between supply and demand for food and income, particularly in developing countries (Robards and Reeves, 2011; Leeney et al., 2015; de Boer et al., 2016). Recent studies have highlighted the urgency of the issue. Marine mammals have been targeted for bait (Campbell et al., 2020; Briceno et al., 2021) or to attract fish to modified fish aggregating devices (Castro et al., 2020).

This process increases the likelihood of a fishery transitioning from bycatch to unregulated and directed harvest of marine mammals (DeMaster et al., 2001), exacerbating the threats to marine mammal populations. These factors should also be included in existing management strategies.

Human behavior is another major challenge to the uptake and application of management. Some measures are unlikely to be adopted, unless mandated and enforced. Even with mandates and regulations in place, the compliance levels depend upon the local social and economic conditions (Whitty, 2018). For example, fishing restrictions, quotas, or blanket bans on fisheries, can adversely affect community livelihoods and introduce human-human conflict within social-ecological systems, making management of interactions even more complicated (Dickman, 2010). Furthermore, in areas where marine mammals are protected by law, fishers are often unwilling to report interactions altogether for fear of persecution (Torres et al., 2018). For interactions such as marine mammal depredation of fisheries, the observed disparity between reported and actual depredation levels, particularly for small-scale fisheries, is high (Bearzi et al., 2011). Therefore, although the frequency of depredation may be lower than reported, the perceived economic damages due to depredation are likely to be higher than they actually are (Sepulveda et al., 2018). In certain cases, there may not be a link between depredation events and the involvement of marine mammals (Bearzi et al., 2008). Several other factors may cause gear damage and catch loss, for example, fish or other invertebrate species, or marine debris (Lauriano et al., 2009), which need to be considered in management strategies.

Incentive-based management measures, such as alternative livelihoods have been advocated as a practical and successful measure to manage interactions. However, such approaches may have limited success in poverty alleviation or be unsuitable for certain areas, particularly remote areas in low income countries, and for small-scale fisheries (Marsh et al., 2011; Squires et al., 2021). If implemented at all, such activities need to be carried out in accordance with proper guidelines (Armitage et al., 2009) and regulations, if unintended consequences to local communities, are to be avoided.

The wider implementation and effective enforcement of conservation measures can be severely limited by social-economic factors (Mendoza-Portillo et al., 2020). For example, small-scale fishers in the developing world are unlikely to have the financial resources to implement technological interventions. Even with subsidies to purchase these devices and adequate participant training and awareness, their voluntary long-term usage and maintenance and ultimately, their efficacy is highly unlikely (Brotons et al., 2008). For example, the vaquita population is dangerously low, despite the considerable investments in the gillnet ban by the Mexican government. This population decline is largely fueled by the high demand for and high market value of the totoaba (*Totoaba macdonaldi*) coupled with corruption (Brownell et al., 2019).

We were concerned to find that, despite the key role of human behavior and socioeconomics in the management context, only eight of the articles we considered propose the incorporation of human dimensions, mainly, socioeconomics of fisheries, cultural significance of livelihoods, and stakeholder belief systems (Szteren and Lezama, 2006; Bearzi et al., 2011; Morteo et al., 2012; Iriarte and Marmontel, 2013; Cook et al., 2015; Leeney et al., 2015; Snape et al., 2018; Mustika et al., 2021).

In addition to human behavior, the behavioral ecology and life-history traits of marine mammals are also major challenges to evaluating the long-term success of management measures. Marine mammal behavior plays an important role in the design and testing of technological interventions. For example, acoustic deterrent devices, or pingers have been shown to be unsuccessful in the long term in deterring marine mammals that exhibit behavioral plasticity. Especially for species with high site fidelity, pingers usually cause "habituation" or act as a "dinner bell," facilitating higher interaction rates (Brownell et al., 2019). For instance, pingers successfully reduced Franciscana bycatch in bottom-set gillnet fisheries, and common dolphins and beaked whale bycatch in drift gillnet operations. However, Cox et al. (2004) concluded that after the initial success in bottlenose dolphin bycatch reduction, these devices were unlikely to further reduce their bycatch rates in gillnets.

Integrating data on marine mammal life-history traits is also vital to setting pragmatic goals for the management of interactions, particularly for bycatch mitigation. Fisheries bycatch models generally assume relatively a large population sizes of the by-caught species and seldom consider the impacts of the Allee effect (Wade and Slooten, 2020), i.e., reduced population growth rate at small population sizes. The Allee effect can have a significant influence on small, restricted populations, for example, coastal small cetaceans (Allee et al., 1949; Berec et al., 2007). Furthermore, the k-selected life-history traits of marine mammals, i.e., low reproductive rate and relatively slow rate of population growth, further limit population recovery (Mannocci et al., 2014). In such scenarios, even if fisheries-induced risks remain constant, bycatch could be a rare event with very deleterious population impacts, especially on small declining populations (di Sciara et al., 2016).

To effectively manage the resource overlap between marine mammals and fisheries, management guidelines should include social and economic dimensions of fisheries and marine mammal ecology. These measures could help in setting pragmatic and achievable conservation priorities based on enhanced understanding of critical knowledge gaps. The heterogeneity in the cultural, social and political nature of small-scale fisheries warrants a holistic understanding of these knowledge systems to implement temporal and spatial management measures. Bycatch is a major conservation concern in marine ecosystems, that requires a multidisciplinary approach to mitigation (Lent and Squires, 2017; Squires et al., 2021).

In addition to the occurrence, abundance and behavior of marine mammals, information on the effects of artificial resource provisioning on predators can shed light on how trophic interactions affect other functional groups in the ecosystem, including prey species (mainly fish stocks), and therefore fisheries (Tixier et al., 2019). However, there is also a need to develop social and economic perspectives of these interactions for the application of any mitigation measures to be contextually successful, both temporally and spatially. An area-specific adaptive management framework could therefore be an effective tool in reducing the risk to marine mammals from fisheries by coupling technical solutions with socio-economic and political interventions.

CONCLUSION

Marine mammal interactions with fisheries occur frequently. Marine mammal bycatch in fisheries is widely studied in literature and remains a major conservation concern due to its detrimental effects on marine mammal populations. However, despite its occurrence in many fishery operations, the effects of depredation on marine mammal ecology and fishery operations have rarely been quantified (Mooney et al., 2009). Depredation poses short-term benefits for marine mammals, creating new foraging opportunities directly facilitated by fishing operations (Tixier et al., 2015, 2020; Esteban et al., 2016). Such interactions could also result in increased chances of entanglement, injuries, and bycatch (Lewison et al., 2014; Guinet et al., 2015; Werner et al., 2015) due to the proximity of marine mammals to active fishing gear.

The biggest management challenge is our inadequate understanding of the social dimensions of these interactions and the inevitable uncertainties concerning both animal and human behaviors (Panagopoulou et al., 2017; Whitty, 2018; Mendoza-Portillo et al., 2020). Human behavior and fishery socioeconomics, in particular, play a key role in effective management of interactions and very few studies address these limitations (Northridge et al., 2017; Lewison et al., 2018).

An adaptive management response, i.e., (i) identifying management challenges and (ii) addressing the related data gaps in both the ecological and human dimensions should be used to set context-specific research and management priorities. Such a multidisciplinary approach could help in underlining the best possible measures for mitigating bycatch based on the local social-ecological conditions (Lent and Squires, 2017). For example, for a given region, with prior information on the fishery operations, and available data on marine mammal interactions with fisheries, the challenges could be evaluated against any existing management actions. This exercise could be then used to identify area-specific research gaps and to set pragmatic targets for research, management and social well-being in that area.

Clear conservation policies coupled with political will and capacity are important for the meaningful implementation of mitigation measures (Kuiper et al., 2018). With the limitations of technological solutions, coupled with socio-economic and political challenges, an adaptive management framework can be an effective tool in reducing the risk to marine mammals from fisheries. Nonetheless, we conclude that despite the vast body of literature on this subject, a “silver bullet” management solution to marine mammal interactions with fisheries does not yet exist (Bearzi et al., 2011; Snape et al., 2018).

AUTHOR CONTRIBUTIONS

KJ, HM, and DS contributed to the conception of the study. KJ conducted the review, analyzed the data, and wrote the first draft of the manuscript. All authors contributed to the editorial and review process and approved the final version of the manuscript.

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