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Ecological risks of a data-limited fishery using an ensemble of approaches

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ABSTRACT

Overexploitation is currently the main cause of marine defaunation. Vulnerability to overexploitation varies across populations. Determining which populations are of highest ecological risk from fishing mortality guides management. Because no single approach is optimal across taxonomic groups, a multi-model ensemble of relative risk estimates for a data-poor Pacific Ocean tuna longline fishery was obtained from two semi-quantitative Productivity-Susceptibility Analyses (PSAs) and from a quantitative approach that estimates instantaneous fishing mortality to compare to reference points of yield-per-recruit models. Individual estimates were combined to produce a pooled mean relative risk rank order. The study identified stocks below biological limits for which the contribution from this fishery to cumulative anthropogenic mortality may warrant intervention. Relative risks in descending order were for populations of albatrosses, cetaceans, mesopelagic sharks, rays, marine turtles, epipelagic sharks and teleosts. The fishery's contribution to cumulative fishing mortality of western central north Pacific Ocean striped marlin warrants a more rigorous assessment to determine absolute risks. The study identified the disparate factors explaining relative risk from an individual fishery versus absolute risk from cumulative anthropogenic mortality sources. Improved risk assessments are possible by addressing identified deficits with PSAs, obtaining information on variables that explain catch and post-capture survival risks that was unavailable for this assessment, improving electronic monitoring data quality and filling gaps in life history traits. Findings support stakeholders to design integrated bycatch management frameworks that mitigate fishing mortality of the most vulnerable taxa and account for multispecies conflicts that result from some bycatch mitigation methods.

1. Introduction

Overexploitation, including from fishing mortality, is a primary threat to marine species, risking protracted or irreparable harm and permanent loss of populations and species, with consequences across manifestations of biodiversity and ecosystem services, including fishery yields [1–3]. Relative vulnerability to overexploitation and other anthropogenic mortality sources varies across populations of marine species. Determining which populations are of highest ecological risk provides critically important information to guide natural resources management [4,5].

Direct fishing mortality by pelagic marine fisheries is a main driver of reductions in the size and abundance of many pelagic apex predator populations, including of targeted and incidentally caught species [6–9]. Fishing mortality can reduce genetic diversity and fitness of affected populations, and alter the state of ecosystem balance that produces targeted services, including fishery yields [10–12]. Fishing mortality of large, highly migratory pelagic predators of high trophic levels indirectly modifies trophic food web structure and processes and functionally-linked systems [6,13–16]. Fisheries targeting relatively fecund species can have profound impacts on incidentally caught species that, due to their lower reproduction rates and other life history traits, are relatively vulnerable to increased mortality. These vulnerable groups include seabirds, marine turtles, sea snakes, marine mammals, chondrichthyans (sharks, rays and chimaeras) and some teleosts [17–21]. Their populations can decline quickly and are slow to recover once depleted.

Ecological risk assessments (ERAs) evaluate the magnitudes and likelihood of adverse ecological consequences of anthropogenic and natural stressors [22]. Methods for ERAs of the effects of fishing have been developed, somewhat recently, for the continuum of data-poor to data-rich fisheries. ERA methods include rapid, first order, qualitative

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evaluations, semi-quantitative assessments, and model-based quantitative assessments [5,23–25]. Most semi-quantitative fisheries ERAs are designed to determine population- and species-level relative risks from fishing mortality of taxonomic groups especially vulnerable to overexploitation for data-limited fisheries through productivity-susceptibility analyses (PSAs) [26–28]. PSAs can also identify which regional fishing gear types are the highest risk to a vulnerable stock [28]. ERAs can holistically assess relative risks from fishing operations across affected taxonomic groups or risks at other levels of marine biodiversity, including effects on habitat, on genetic diversity and evolutionary processes resulting from selective fishery removals, and on broad changes in components of the state of ecosystem functions and structure [5,29–31].

Findings from semi-quantitative PSAs, and a relatively new model-based ERA that uses many of the same attributes as PSAs [25], that are suitable for data-poor settings can rapidly identify highest-risk biodiversity units (e.g., populations and stocks, species, habitats) so that precautionary management responses can be quickly implemented [5]. Managers could respond to relative risk findings by identifying interventions that provide precautionary reductions in risk for these most vulnerable units. Some management measures designed to increase fishing gear selectivity to reduce the bycatch of one species exacerbate the catch risk of other species of conservation concern, including hook shape, bait type, time-of-day of fishing and fishing depth [32]. Findings on relative risks from semi-quantitative ERAs enable managers to account for these cross-taxa conflicts.

The relatively lower-certainty PSAs are useful for assessing large numbers of biodiversity units to triage those deemed to be relatively vulnerable or of uncertain risk to undergo more rigorous analysis [5]. Findings from the relatively data-light but low-certainty PSAs can thus guide the use of often limited resources to conduct more data-intensive, higher-certainty, model-based assessments to estimate and manage absolute risk for these most vulnerable populations and stocks [5,33]. For highly vulnerable populations with insufficient data quality to support more rigorous risk assessments, semi-quantitative ERA findings enable prioritizing research to fill those data gaps.

Numerous methods are now available that effectively mitigate pelagic longline bycatch rates and mortality of species of conservation concern that are also economically viable, practical, safe and enable compliance monitoring, although there has been mixed progress in their uptake [30,34]. Findings from ERAs support stakeholders to design and implement an integrated bycatch management system that mitigates fishing mortality of the most vulnerable taxa, where tradeoffs may be required when multispecies conflicts are unavoidable [32].

This study conducted an ERA of a data-limited, distant-water tuna longline fishery to determine the relative vulnerability of populations and stocks that make up the catch. Because no single approach for assessing relative ecological risks is optimal across taxonomic groups with variable life histories, an ensemble of approaches to estimate relative risks was employed. Two common PSA approaches [35–40] were used. A new quantitative ERA approach that estimates instantaneous fishing mortality to compare to reference points of yield-per-recruit models, which employs many of the same attribute values as conventional PSAs [25], was also used. Estimates from this ensemble of ERA models were combined to produce a pooled mean relative risk rank order. Available estimates of absolute status were compiled for stocks susceptible to capture in the fishery to identify those that may be below limit biological thresholds, where the contribution from this fishery to cumulative anthropogenic mortality may warrant reductions. Findings inform stakeholders to adopt responses that provide precautionary protection for the most vulnerable populations and stocks, select management measures with acceptable tradeoffs when multispecies conflicts are unavoidable, and identify taxa warranting more rigorous, quantitative risk assessments to determine absolute risks.

2. Methods

2.1. Fishery

An ecological risk assessment was conducted for a fishery comprised of 10 distant-water pelagic longline vessels that fish across the Pacific Ocean. The vessels are flagged to the Pacific Island country of Vanuatu and owned by the Taiwanese Tunago Fishery Company. Vessels have 30 crew and transship catch at sea. The fishery is in a Fisheries Improvement Project with plans to pursue certification against the Marine Stewardship Council (MSC) fisheries standard [41].

The 10 Tunago vessels that were assessed in this study are a subset of the Vanuatu-flagged longline fleet. During the five-year period between 2014 and 2018, there were between 49 and 82 Vanuatu-flagged pelagic longline vessels operating in the Pacific Ocean, with albacore (*Thunnus alalunga*) and bigeye (*T. obesus*) tunas making up the largest proportion of the retained catch [42].

2.2. Stocks and populations assessed for relative risks

An initial step in identifying candidate populations and stocks for inclusion in the risk assessment was to determine which species are captured in the Tunago fishery, and identify species used for bait. The species list was compiled using: electronic monitoring (EM) data from a single trip by a Tunago vessel (Supplemental Material Table S1); amalgamated catch data for the full Vanuatu-flagged albacore and bigeye tuna longline fishery during calendar years 2018 and 2019 [42]; and information on bait provided by the vessel owner. Within each taxonomic group of teleosts, sharks, rays, turtles, seabirds, marine mammals, we then selected stocks and populations for inclusion in the relative risk assessment based on which species made up the largest proportion of the catch within groups, and which had the worst stock status based on B/B_{MSY} or proxy, for teleosts and elasmobranchs and poorest species- or population-level conservation status, for marine turtles, seabirds and marine mammals [43–47]. Western and Central Pacific Ocean (WCPO) skipjack tuna (*Katsuwonus pelamis*), a relatively abundant stock despite being heavily fished [48], was also included in the relative risk assessment, along with two other fish stocks that also have conclusive stock assessment findings (Pacific bluefin tuna *Thunnus orientalis*, WCPO oceanic whitetip shark *Carcharhinus longimanus*) but are considered to be in relatively poor condition [49,50], in order to enable comparing absolute stock status to estimated relative risk from the Tunago fishery, and to demonstrate the different approaches of the PSAs and a quantitative, model-based ecological risk assessment method, referred to as the Ecological Assessment of the Sustainable Impacts of Fisheries or EASI-Fish [25]. We excluded stocks of target species from the ERAs because, even if determined to be relatively high risk, this would not elicit a management response of increasing fishing selectivity to reduce their catch rates, and because WCPO skipjack provided a better example of a relatively abundant teleost.

Seabirds, however, were not included in the EASI-Fish assessment (described in Section 2.3.3) because no estimated values were available for the a and b parameters of the length-weight equation $W = a * L^b$ [51]. This length-weight relationship is needed because the EASI-Fish yield per recruit and spawning biomass per recruit models operate in units of biomass [25]. Additionally, a von Bertalanffy growth function is likely not relevant for seabird species [52], and anthropogenic mortality sources in addition to fishing would need to be accounted for in the model as conducted for marine turtles.

2.3. Ecological risk assessment approaches

2.3.1. Marine Stewardship Council PSA

The study used the MSC Risk-Based Framework (RBF) [35–39]. The method is based on the Commonwealth Scientific and Industrial Research Organization (CSIRO) PSA approach [5,24,53]. Of the numerous adaptations to the CSIRO PSA approach, we employed MSC's because it is globally and broadly employed, the approach is standardized, it is relatively simple to implement given the availability of readily accessible tools and low data quality requirements, and MSC's methodology and Excel-based software are open source [35–39]. The MSC PSA approach can therefore be easily replicated and findings validated.

Supplemental Material Section S2 summarizes the MSC RBF approach. When assessing fisheries against the MSC fisheries standard, the MSC RBF is intended to be used only for components that are data-deficient, determined based on criteria identified in MSC's Fisheries Certification Process [37]. The MSC PSA approach allows for grouping species of similar taxonomies when assessing a large number ($>$ 15) of species. Here, however, we applied MSC's PSA approach individually to each assessed stock/population. We excluded the productivity attribute density dependence, which is intended to be used only to assess invertebrates.

For the areal overlap attribute, for stocks and populations where native distribution maps from AquaMaps [54] were available, we estimated the proportion of half-degree cells of the stock/population distribution that overlap with the fishing grounds of the Vanuatu-flagged tuna longline fishery. We included AquaMaps species distribution cells with a probability of occurrence of $\geq$ 0.5. For the two albatross species, we used a shapefile from BirdLife International and Handbook of the Birds of the World [55] of maximum non-breeding distributions (no estimates of the probability of occurrence) that was converted to rasterized points and further rasterized to plot the points within one-degree cells. To define the geospatial distribution of the fishing grounds, we used the WCPFC public domain database for positional data, aggregated into 5° x 5° cells, available from 2000 to 2017, for all Vanuatu-flagged longline vessels [56] and pooled this with Tunago EM positional data from two trips conducted in 2018–2019. Fig. 1 exemplifies this approach to estimate geo-spatial overlap, showing the distribution of Pacific Ocean populations of the leatherback marine turtle (*Dermochelys coriacea*) with a probability of occurrence scale, and the location of Vanuatu longline fishing effort. Supplemental Material Section S3 contains areal overlap figures for the other assessed stocks and populations.

In addition to the areal overlap attribute, we modified the MSC PSA approach to fill gaps in MSC guidance and to make the method more suitable for application to species susceptible to capture in pelagic longline fishing gear. This resulted in modifications or clarifications to the MSC attributes reproductive strategy, average maximum size and, average size at maturity, encounterability and selectivity of gear type, described in Section S2.2.

The study compiled productivity and susceptibility attribute values for each assessed stock and population. No MSC full assessments of longline fisheries have employed the MSC RBF for the stocks and populations included in this PSA [57–63], so it was not possible to harmonize scores for susceptibility attributes. The attribute values were used to produce individual and overall productivity and susceptibility attribute scores, an overall risk score *R* (referred to as V, vulnerability, in the CSIRO PSA method, [5]), and overall relative risk category of low, medium or high [36–38].

2.3.2. Patrick PSA

The study used a second PSA approach of Patrick et al. [40], summarized in Supplemental Material Section S4. For the areal overlap attribute, we employed the approach described for the MSC PSA. We excluded the susceptibility attribute assessing the impact of fisheries on habitat. In this study only a single gear type was assessed, and there are likely small differences in relative risks to assessed stocks and populations from habitat effects of this pelagic longline fishery. Furthermore, pelagic longline gear is understood to have minimal risk of direct habitat impacts (e.g., Patrick et al. [64] assigned low risk scores to all pelagic longline fisheries), although some pelagic longline fisheries may contact the seabed when fishing at shallow submerged features such as seamounts [65].

Each attribute was assigned the same weight (except for the excluded habitat attribute which was assigned a weight of 0), as conducted by Patrick et al. [64] for an assessment of Hawaii's pelagic longline fisheries. When information is not sufficient to assess an attribute, then the attribute is not scored (it is assigned a weight of 0) [40]. To enable comparisons of findings from the MSC PSA, we reversed the productivity scale of Patrick et al. [40] so that an individual productivity score of 1 = low risk, 3 = high risk, and overall risk scores and overall relative

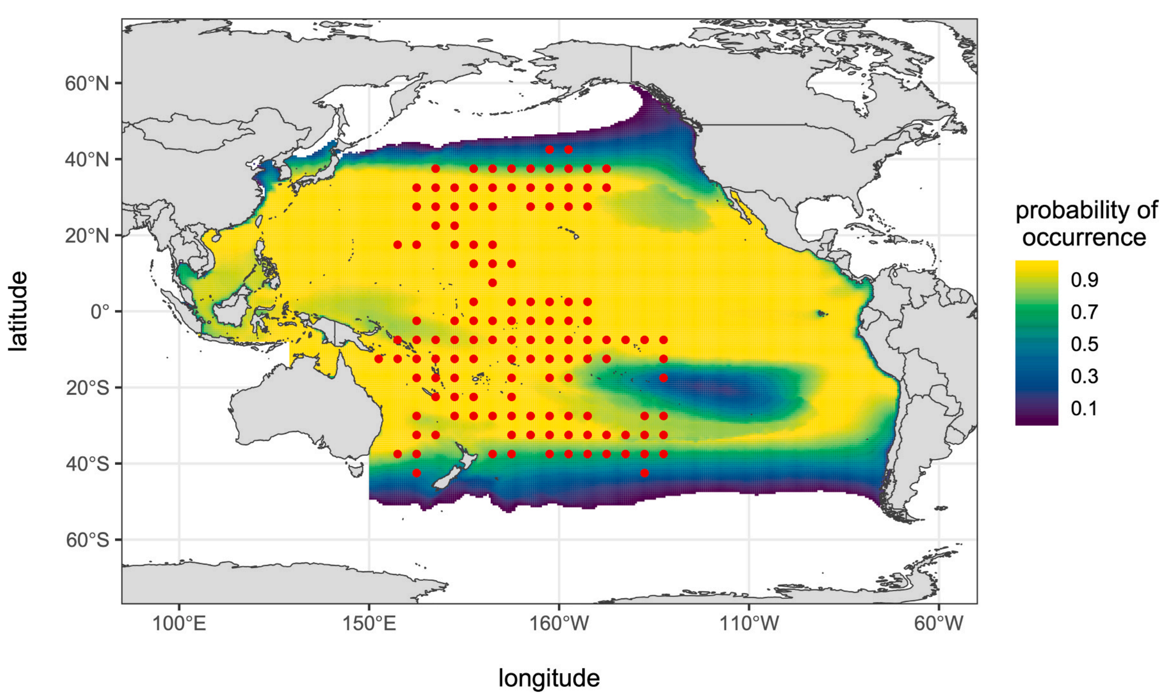

Fig. 1. Spatial distribution of Pacific Ocean populations of the leatherback sea turtle and probability of occurrence by half-degree cell from data from AquaMaps [54], and five-degree cells of fishing grounds of the Vanuatu-flagged tuna longline fishery from WCPFC [56] and from EM data from Tunago vessels (solid dots, marking centroids of 5 × 5 cells).

risk categories were assigned following the MSC PSA approach [36–38].

As with the MSC PSA approach, the Patrick et al. [40] PSA method was selected because the method is standardized, enabling studies to be replicated and findings validated. To assess fish stocks captured in U.S. fisheries, Patrick et al. [40] adapted the CSIRO approach, including by employing substantially expanded suites of both productivity and susceptibility attributes. The Patrick et al. [40] method has subsequently been used for assessments of other longline fisheries (e.g., Taiwan pelagic longline fisheries, [66]). The broader suite of attributes may provide more robust estimates of relative risks if the additional attributes are measuring aspects of ecological risk that are substantially different and not redundant with the original MSC PSA attributes. As with the MSC PSA, attribute values were compiled and used to produce individual and overall productivity and susceptibility attribute scores, an overall risk score *R*, and overall relative risk category of low, medium or high [40].

2.3.3. EASI-Fish ERA

EASI-Fish, a quantitative, model-based ecological risk assessment method [25], was a third relative risk ERA method included in the study. Instead of employing arbitrary reference points for productivity and susceptibility to assess relative risk as employed in the CSIRO PSA and adaptations, EASI-Fish uses susceptibility attributes to approximate instantaneous fishing mortality (*F*), based on the proportion of individual length classes of a stock that are susceptible to capture and mortality annually. *F* is then compared to reference points used in length-structured yield-per-recruit models, the productivity component of the ERA, to determine the vulnerability status of each assessed stock/population [25]. Results can be displayed in Kobe-style plots to demonstrate whether spawning stock biomass per recruit (SBR) and *F* for each assessed stock/population is above or below selected reference points. Supplemental Material Section S6 summarizes the EASI-Fish productivity and susceptibility attribute definitions. We adapted the attribute on the proportion of a year during which fishing is permitted to use the proportion of the year during which the fishery occurs, which for the Tunago fishery was estimated to be 64.4% (235 days of the year). The approach described for the MSC PSA for the areal overlap attribute was also employed for the EASI-Fish ERA.

To achieve the study objective of assessing relative risks of assessed populations/stocks both within and across species groups, we adopted different F- and B-ratios for teleosts, elasmobranchs, sea turtles and marine mammals in an attempt to use comparable reference point thresholds for each group that accounts for differences in their stock-recruitment relationship [25,67–70]. For teleosts we used $F_{current}/F_{MSY}$ and $SBR_{current}/SBR_{MSY}$, where F = fishing mortality rate, SBR=spawning stock biomass per recruit, and MSY=maximum sustainable yield. As proxies for these MSY-based reference points, for elasmobranchs we used $F_{current}/F_{40\%}$ and $SBR_{current}/SBR_{40\%}$, and for marine turtles and marine mammals we used $F_{current}/F_{80\%}$ and $SBR_{current}/SBR_{80\%}$ (for definitions of these biological reference points, see [25,70–73]). We selected these reference points because they approximate common benchmarks to enable an assessment of relative vulnerability. These reference points were not selected for use as suitable stock-specific or ecosystem-based limit reference points (a minimum threshold representing a risk of protracted or irrevocable harm) or target reference points (a desired state of a fishery and resource based on socioeconomic and biological considerations that managers aim to be near) [71,73–75].

As with the two PSA approaches, EASI-Fish was selected because it is standardized, facilitating replication of studies and findings to be validated. However, the model is not currently open source. The approach has been recently used for a growing number of regional assessments, including the initial assessment of eastern Pacific Ocean (EPO) tuna fisheries [25,69,76].

Because this study assessed the risks from an individual fishery and thus a small subset of regional fishing mortality, the study did not estimate *absolute* risk of assessed stocks and populations, i.e., whether a stock's current regional fishing mortality and biomass exceed a reference point, as conducted in previous EASI-Fish applications [25,69,76]. EASI-Fish is designed to present the absolute vulnerability status of assessed stocks and populations by assessing most or all of the fisheries contributing to total fishing mortality for that stock/population, and for species susceptible to other anthropogenic mortality sources, by accounting for these other non-fishery mortality sources (e.g., marine turtle egg collection, [69]). EASI-Fish, however, can also be used to estimate relative risks, including for a single fishery. For this study, a Kobe plot display of EASI-Fish outcomes, while not estimating effects of total anthropogenic mortality on stock yields, indicates the relative impact of the individual assessed fishery on stock-specific spawning per recruit. While EASI-Fish and PSA results are not directly comparable [25], the relative risk rank order of assessed stocks, however, are relatable as different indices of relative risk.

2.3.4. Vulnerability categorizations using stock assessment findings

The study identified the subset of the fish stocks susceptible to capture by the Tunago fishery for which stock assessments have been conducted and produced certain or otherwise indicative estimates of absolute biomass and exploitation rates and MSY-based or otherwise proxy reference points. North Pacific Ocean (NPO) stocks of Pacific saury (*Cololabis saira*), which is used for bait by the Tunago fleet, was also included. Of these stocks, if the B-ratio suggested that the stock biomass may be below a limit biological threshold, we then estimated the Tunago fishery contribution to total annual estimated catch as an indicator of the risk that the Tunago fishery poses to the stock.

2.3.5. Multi-model ensemble of relative risk estimates

The individual estimates of stock-specific relative vulnerability from this ensemble of three ERA models were combined to produce a pooled mean relative risk rank order. First, to obtain rank order values on the same scale for the PSAs and EASI-Fish, the rank order values from each risk assessment were standardized by the mean and two standard deviations [77]. This was necessary because a different number of stocks were assessed by the two PSAs and EASI-Fish. The overall rank order was then calculated as the mean of (i) the standardized EASI-Fish rank value and (ii) the mean of the two PSA standardized rank values, except for the two albatross species whose overall risk score was calculated as the mean of the standardized rank order values from the two PSAs only. For the three stocks with stock assessment findings that were included in the relative risk assessments, we compared the individual and pooled relative risk findings to the stock assessment findings on F- and B- ratios.

3. Results

3.1. Individual and pooled relative risk estimates

Fig. 2 is a PSA plot from the MSC PSA, summarizing the overall scores for productivity and susceptibility attributes. The key to the data labels in the figure caption lists the assessed stocks and populations in rank order from highest to lowest overall risk score *R*. Fig. 3 is a PSA plot using the Patrick et al. [40] approach, but with the x- and y-axis scales adapted to enable comparing with the MSC PSA results, summarizing the overall scores for productivity and susceptibility attributes. As with Fig. 2, the key to the data labels in the caption of Fig. 3 lists the assessed stocks and populations in rank order from highest to lowest overall risk score *R*.

Fig. 4 summarizes the F- and B-ratios for assessed stocks and populations using the EASI-Fish [25] risk assessment method in a Kobe-style plot. The rank order from lowest B-ratio (i.e., highest to lowest relative risk) is identified in the caption of Fig. 4.

Supplemental Material Sections S3 and S5 contain the compiled productivity and susceptibility values and individual attribute scores for the two PSAs. Section S7 contains the attribute values for the EASI-Fish relative risk assessment.

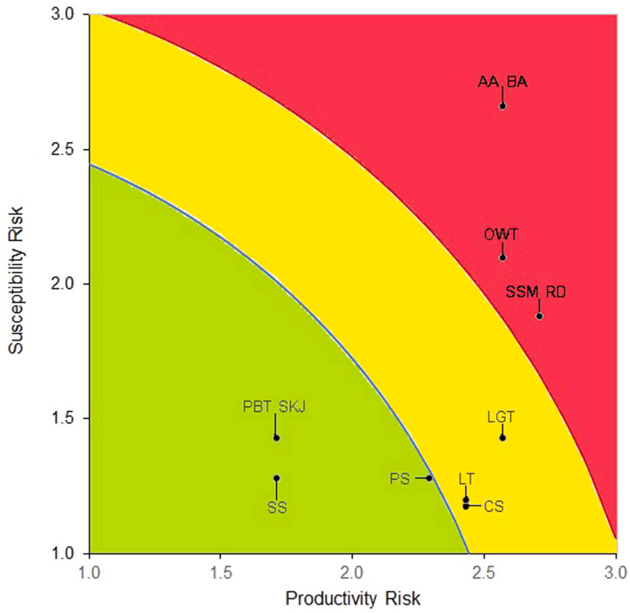

Fig. 2. PSA plot using the MSC RBF [37,38] for assessed populations, stocks and species. Red and blue lines divide overall risk *R* values of the PSA plot (calculated as the Euclidean distance from the origin of the x-y scatter plot) into three equal areas, where *R* values < 2.64 (shaded green) are categorized as low relative risk, between 2.64 and 3.18 as medium relative risk (shaded yellow), and > 3.18 as high relative risk (shaded red) [5,36,38]. Data labels, from highest to lowest *R*: (1 and 2) AA = antipodean albatross *Diomedea antipodensis*, Pacific Ocean populations and BA = black-footed albatross *Phoebastria nigripes* – same *R* values; (3 and 4) RD = rough-toothed dolphin *Steno bredanensis* and SSM = shortfin mako shark, SPO stock – same *R* values; (5) OWT = oceanic whitetip shark, WCPO stock; (6) LGT = loggerhead marine turtle *Caretta caretta*, Pacific Ocean stocks; (7) LT = leatherback sea turtle, Pacific Ocean stocks; (8) CS = crocodile shark *Pseudocarcharias kamoharai*, Pacific Ocean stocks; (9) PS = pelagic stingray *Pteroplatytrygon violacea*, Pacific Ocean stocks; Pacific Ocean populations; (10 and 11) PBT = Pacific bluefin tuna and SKJ = skipjack tuna, WCPO stock – same *R* values; (12) SS = shortbill spearfish, NPO and SPO stocks.

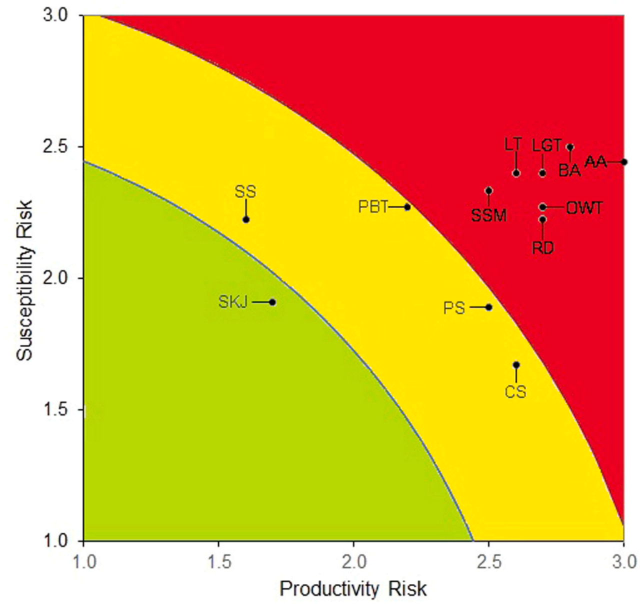

Fig. 3. PSA plot using the Patrick et al. [40] approach for assessed populations, stocks and species. Red and blue lines divide overall risk *R* values of the PSA plot (calculated as the Euclidean distance from the origin of the x-y scatter plot) into three equal areas, where *R* values < 2.64 (shaded green) are categorized as low relative risk, between 2.64 and 3.18 as medium relative risk (shaded yellow), and > 3.18 as high relative risk (shaded red) [5,36,38]. Data labels, from highest to lowest *R*: (1) AA = antipodean albatross, (2) BA = black-footed albatross; (3) LGT = loggerhead sea turtle, Pacific Ocean stocks; (4) LT = leatherback sea turtle, Pacific Ocean stocks; (5) OWT = oceanic whitetip shark, WCPO stock; (6) RD = rough-toothed dolphin, Pacific Ocean populations; (7) SSM = shortfin mako shark, SPO stock; (8) PBT = Pacific bluefin tuna; (9) PS = pelagic stingray, Pacific Ocean stocks; (10) CS = crocodile shark; Pacific Ocean stocks; Pacific Ocean populations; (11) SS = shortbill spearfish, NPO and SPO stocks; (12) SKJ = skipjack tuna, WCPO stock.

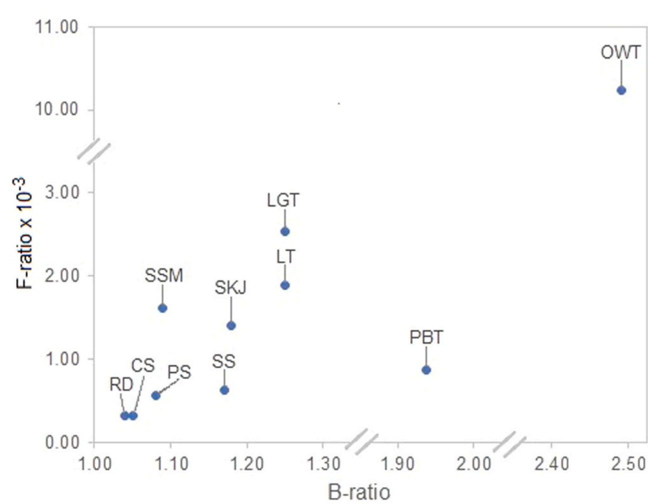

Fig. 4. Kobe-style plot using the EASI-Fish [25] approach for assessed populations, stocks and species. Reference points for teleosts: F/F_{MSY}, B/B_{MSY}; elasmobranchs: $F/F_{40\%}$, $SBR/SBR_{40\%}$; marine turtles and cetaceans: $F/F_{80\%}$, $SBR/SBR_{80\%}$. Data labels, from highest to lowest B-ratio: (1) RD = rough-toothed dolphin, Pacific Ocean populations; (2) CS = crocodile shark, Pacific Ocean stocks; (3) PS = pelagic stingray, Pacific Ocean stocks; (4) SSM = shortfin mako shark, SPO stock; (5) SS = shortbill spearfish, NPO and SPO stocks; (6) SKJ = skipjack tuna, WCPO stock; (7) LT = leatherback sea turtle, Pacific Ocean stocks; (8) LGT = loggerhead sea turtle, Pacific Ocean stocks; (9) PBT = Pacific bluefin tuna; (10) OWT = oceanic whitetip shark, WCPO stock.

Fig. 5 shows the relative risk rank order from the three individual risk assessments and an overall or pooled relative risk rank order, all using the same scale, and with populations/stocks ordered by the overall rank.

3.2. Stock assessment findings

There were 18 fish stocks captured in the Tunago fishery that have undergone stock assessments that produced conclusive or indicative estimates of absolute levels of abundance, exploitation rates, and MSY-based or otherwise proxy reference points:

- WCPO and EPO bigeye tuna [78–80];
- WCPO and EPO yellowfin tuna (*T. albacares*) [81–83];
- WCPO skipjack tuna (*Katsuwonus pelamis*) [48];
- NPO and south Pacific Ocean (SPO) albacore tuna [84,85];
- Pacific bluefin tuna [49];
- Western and central north Pacific Ocean (WCNPO) and southwest Pacific Ocean (SWPO) striped marlin (*Kajikia audax*) [86–88];
- NPO shortfin mako shark (*Isurus oxyrinchus*) [89];
- WCNPO and SWPO swordfish (*Xiphias gladius*) [90,91];
- Pacific Ocean blue marlin (*Makaira nigricans*) [92];
- NPO blue shark (*Prionace glauca*) [93];
- WCPO oceanic whitetip shark [50];
- WCPO silky shark (*C. falciformis*) [94]; and

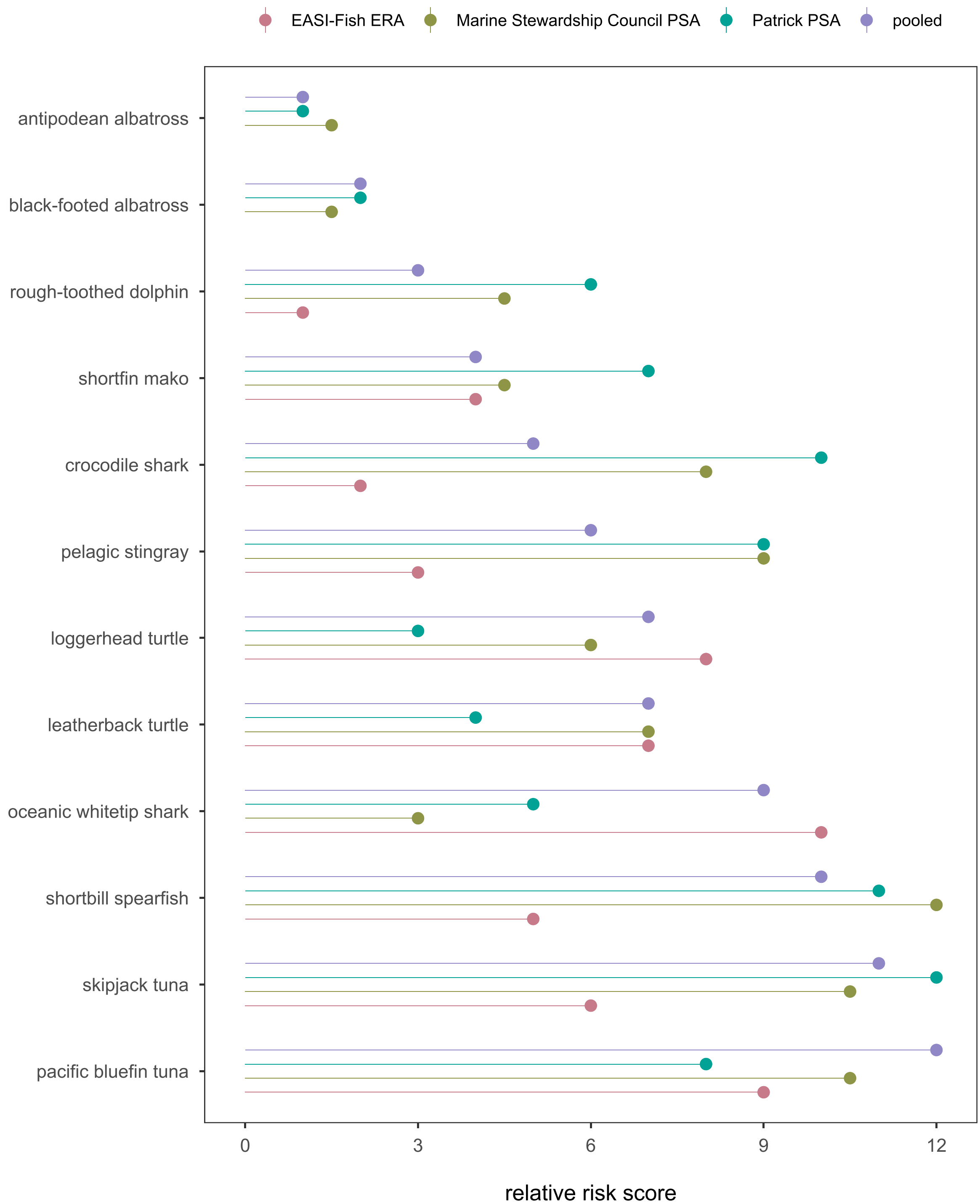

Fig. 5. Individual ERA relative risk rank-orders for EASI-Fish [25], Marine Stewardship Council [35–39] and Patrick et al. [40], and overall, pooled relative risk rank order. A rank of 1 indicates highest risk (most vulnerable), and 12 lowest risk. Stocks/populations are ordered by the overall, pooled rank.

- Combined stocks of Pacific saury [95].

Of these, six had B/B_{MSY} or proxy < 1, for which the rank order, from lowest to highest B-ratio, was: (1) Pacific bluefin tuna, (2) WCPO oceanic whitetip shark, (3) WCNPO striped marlin, (4) SWPO striped marlin, and (5) Pacific Ocean stocks of saury, with (6) EPO bigeye tuna borderline ($SSB_{current}/SSB_{MSY}$ ~0.92). Of these six stocks, SWPO striped marlin and Pacific saury have F-ratios < 1, EPO bigeye tuna's F-ratio ~1, and the remainder are experiencing overfishing with F-ratios > 1.

Three of these fish stocks are likely below biomass limit reference points. Pacific bluefin tuna is heavily depleted ($SSB/SSB_{F=0}$ = 0.045) [49]. This depletion level is substantially below the biomass depletion-based limit reference point of 20% of the unfished stock biomass ($20\%SSB_{F=0}$) adopted by the tuna regional fisheries management organization for other key tuna stocks. The current fishing mortality rate is above the rebuilding target threshold (i.e., overfishing is occurring), but the fishing mortality rate has been declining and the spawning and juvenile biomass have both been experiencing moderate increasing trends over the past decade [49]. The absolute stock-level effect of the individual Tunago fishery is likely negligible. Based on catch data for the full Vanuatu longline fishery, the Tunago fishery catches ca. 0.75 t of Pacific bluefin tuna per year (about 4 individual Pacific bluefin per year), which is $< 0.01\%$ of the 2018 estimated total weight of the catch [42,49,96]. The available EM data did not have any records of captured bluefin, but the EM analyst was unable to identify to the species level over 30% of the tuna catch (Table S1).

The absolute stock-level effects of the Tunago fishery on WCPO oceanic whitetip shark may also be negligible, based on the available, very limited, catch data. The stock is likely below a biological limit threshold with median $SB_{recent}/SB_{F=0} = 0.04$, median $SB_{recent}/SB_{MSY} = 0.09$ and subject to substantial overfishing with median $F_{recent}/F_{MSY} = 3.92$ [50]. Based on logbook data for the full Vanuatu longline fishery [42], the Tunago fleet is estimated to annually capture 13 kg, which is $< 0.0005\%$ of the total annual estimated weight of the catch [50]. As with bluefin, the Tunago EM data did not contain records of captured oceanic whitetip sharks, but the EM analyst was unable to identify to the species level a large proportion of the shark catch (Table S1).

Extrapolating from the Tunago EM data and weight-at-length equation from Sun et al. [97], the fishery annually captures about 29.7 t of WCNPO striped marlin, which is 1.4% of the weight of the total mean annual estimated catch from the most recent available five years [87]. The stock may be below a biological limit threshold ($SSB/SSB_{F=0} = 0.054$, $SSB_{recent}/SSB_{MSY} = 0.38$) and not rebuilding [87].

4. Discussion and conclusions

4.1. Most vulnerable catch estimated using an ensemble of approaches

The pooled findings from the three ERAs indicated that relative risks to populations and stocks from the Tunago fishery, from highest to lowest relative risk, were for: albatrosses, cetaceans, mesopelagic sharks, rays, marine turtles, epipelagic sharks and teleosts (Fig. 5). Of teleost stocks with certain assessment findings, the Tunago fishery contribution to cumulative fishing mortality of WCNPO striped marlin warrants attention.

Because no single approach for assessing relative ecological risks is optimal across taxonomic groups, we employed an ensemble of methods. This multi-model approach may better approximate common benchmarks so that results are comparable within and across assessed taxonomic groups. Different benchmarks, attributes, attribute definitions and cutoffs, weighting of attributes and methods for scoring are needed to account for varied life histories of assessed stocks and populations [23,25,38,69]. This included PSAs using arbitrarily-defined benchmarks and EASI-Fish using biological reference points intended to be equivalent across assessed units with variable life histories to reflect effects of a fishery in reducing stock yield. For the latter, however, the implications for population-level effects from the same reduction in stock yield for highly productive species versus species with low fecundity and delayed maturation are complex, discussed below.

Whereas PSAs use arbitrarily-defined benchmarks, EASI-Fish estimates quantitative effects on stock yields [25]. EASI-Fish conducts a yield per recruit stock assessment, employing commonly used reference points that are intended to be comparable across assessed biodiversity units, so that findings identify the relative effect of a fishery on stocks/populations against these common thresholds. EASI-Fish findings therefore reflect the effect of a fishery in reducing stock yield, where life history attributes (e.g., the relationship between adult biomass and recruitment) dictate how an individual stock responds to fishery removals. With PSAs, teleosts will typically be ranked as lower risk than non-teleosts due to their disparate life histories, as was the case here. The rank orders of relative risk from the two PSAs were intuitive when considering only productivity attributes: species of seabirds, sharks, marine mammals and turtles, with low fecundity, delayed maturation and other life history traits that make them vulnerable to anthropogenic mortality, were ranked as higher risk than relatively productive teleosts. With EASI-Fish, however, teleost populations may have the same or higher estimated relative vulnerability than non-teleost populations when assessed against equivalent exploitation rate and biomass reference points.

EASI-Fish and PSAs both produced indicators of relative risk. Because this study assessed risks from an individual fishery and thus a small subset of regional fishing mortality, it did not estimate *absolute* risk of whether a stock's current regional exploitation rate and biomass exceed a biological reference point, as conducted in previous applications of EASI-Fish [25,69,76]. Instead, the EASI-Fish results here identified reduced lifetime yield of assessed stocks and populations due only to the individual Tunago fishery. As a result, due to the relatively small fishing mortality rate of the Tunago fishery, the exploitation rates have nominal effect on the yield per recruit of the assessed stocks/populations (all F-ratios were close to 0, Fig. 4). The reference points used in the EASI-Fish assessment may not be biological reference point limits, that when exceeded result in protracted or irreparable harm, or biological extinction. For example, a stock that is below B_{MSY} is not producing maximum long-term sustainable yields, but the yield from this biomass level can be sustained, perhaps in perpetuity, and above limits where risk of extended or permanent impairment or loss is predicted to occur [71,72,75]. This explains why a variety of definitions are used to define when a stock is overfished or when overfishing is occurring [71–73]. But the study's objective was to assess relative and not absolute risk of exceeding a limit threshold, and the selected reference points meet this objective by providing a commensurate indication of the effect of the Tunago fishery across assessed stocks and populations.

Because EASI-Fish attempts to uses the same benchmark reference point for all assessed stocks and populations, results theoretically identify the Tunago fishery's effect on biomass yield across species groups with variable life histories. However, if populations of a long-lived, late-maturing species and short-lived species with maturation at a young age have similar B-ratio values, it is unclear whether this means that the fishery has the same effect on the viability of the two populations. Does being near a biological reference point for a marine turtle population mean the same as being near the same biological reference point for a tuna stock? For example, the EASI-Fish estimated B-ratio for WCPO skipjack tuna was lower than for the two assessed marine turtles. This indicates that the Tunago fishery's reduction in stock yield for the skipjack stock was relatively higher than for the turtle stocks. Small changes in mortality, as well as truncated age structure, Allee effects (depensation, negative population growth at small population sizes), and other population-level as well as ecosystem-level regime shift responses to anthropogenic pressures, have disproportionate effects on species with low fecundity, delayed maturation and other life history characteristics that make them particularly vulnerable to anthropogenic sources of mortality relative to highly productive species [98–101]. This makes it unclear how to estimate the relative population-level effects from a given level of a B-ratio. For instance, marine turtle population dynamics are far more sensitive to changes in reproductive output (number of clutches laid per season, number of skipped breeding seasons) than to mortality for immatures and to a lesser extent for adults (especially adult males) [43,44,102]. Furthermore, the risk of harm to non-market bycatch species as compared to stocks of principal market species from the same level of reduction in spawning biomass or yield per recruit may also differ due to effects of management measures and market responses as a resource becomes increasingly scarce [103,104]. Nonetheless, the biological reference points used in EASI-Fish are pragmatically useful for species with variable life histories, at least for an ERA of a data-deficient fishery as in this study.

The absolute status of a stock does not necessarily reflect the relative risk from an individual fishery for two reasons. First, productivity attributes are only one factor affecting stock status. Market value, gear-specific selectivity and other anthropogenic stressors, for example, may have larger effects on stock/population status than life history traits (i.e., even highly productive and resilient populations with broad distributions can be depleted and extirpated, such as Pacific bluefin tuna, and see Gaston and Fuller [105]). Second, the relative ecological risk from an individual fishery can be independent of absolute stock status. For instance, a highly depleted stock may have an extremely low catch risk and fishing mortality from a particular fishery due to, for example, limited overlap between fishing depth and vertical habitat distribution, or low contact selectivity due to hook size, bait type or mesh size. This was the case for Pacific bluefin tuna. While the stock is heavily depleted [49], it had the lowest pooled rank-order relative risk of assessed stocks and populations. And, the absolute stock-level effect of the Tunago fishery on this stock may be negligible (< 0.01% of the weight of annual estimated catch, based on available, highly uncertain data). Similarly, the Tunago fishery is estimated to not pose a risk to the WCPO oceanic whitetip shark stock (< 0.0005% of the total annual estimated catch), which was estimated to be of high relative risk by the two PSAs but had the highest B-ratio (lowest risk) in the EASI-Fish assessment. However, this again is based on highly uncertain data, including relying on only logbook records, which are notoriously unreliable [106,107]. Furthermore, discussed below, the extremely poor data quality on discards produced by the EM system, and the overwhelming impact of regional longline fisheries on this oceanic whitetip stock [50] provide a strong rationale for prioritizing substantially improved data quality to more robustly assess the effect of the fishery on this stock. The Tunago contribution to annual catch of WCNPO striped marlin, which may also be below a biological limit reference point and not rebuilding [87], was estimated to be 1.4% of the weight of the total annual estimated catch. For stocks such as this one that have exceeded a limit threshold, if sources of anthropogenic mortality individually contribute small proportions of total removals, but cumulatively are substantial, then interventions are needed to manage these individual, small mortality sources [25,108]. This is particularly relevant for taxa with small population sizes and broad distributions where fishery captures are extremely rare events that are distributed across numerous individual fisheries.

Cumulative effects of regional longline fishing mortality of Pacific bluefin tuna are small, where < 10% of total catch is by regional longline fisheries [49], and discussed above, available, limited data suggest that the Tunago fishery has only a nominal effect on this stock. However, mortality in longline fisheries has accounted for almost all WCNPO striped marlin fishing mortality since the early 1990s [87]. Given the importance of longline gear to anthropogenic mortality, combined with the estimated Tunago fishery contribution to regional fishing mortality, strengthens a precautionary recommendation for interventions to reduce catch risk of this stock in the Tunago and other longline fisheries regionally. Related, there is a need for improvements in guidance defining thresholds when individual fishery-level contributions to cumulative fishing mortality of a stock/population warrants management interventions.

4.2. Improving PSA approaches

One issue with obtaining certain outcomes from PSAs is which attributes to include. The selection of attributes should ensure that all main aspects of productivity and susceptibility are covered but without introducing redundancy. If attributes are redundant, then their inclusion would not substantially improve accuracy while making the analysis more onerous, time consuming and expensive [24]. The selection of cutoffs for scoring each attribute and what weights to assign to individual attributes to calculate an overall relative risk value are additional considerations that affect the rigor of PSA findings.

The two PSA approaches had large differences in relative vulnerability outcomes for some populations due to employing different attributes and cutoffs, and different methods for calculating overall productivity and susceptibility scores. For example, the largest differences in relative risk outcomes between the two PSA approaches were for the two marine turtle stocks. The MSC PSA categorized both stocks as medium risk, Patrick as high risk, and these stocks had the largest differences in vulnerability rank order by the two PSAs (Figs. 2, 3, 5). Some attributes included in both approaches, including L_{max} and t_{max}, due to different cutoffs, resulted in higher risk scores by Patrick et al. [40] (Tables S4, S8, S16, S20). Some attributes unique to the Patrick approach assigned high risk scores, such as intrinsic growth rate, natural mortality, contact selectivity, biomass of breeding age classes and management strategy attributes, while some attributes unique to the MSC PSA approach assigned low or medium risk scores for the marine turtle stocks, such as size at maturity and selectivity. The PSAs also had large differences in relative risk estimates for Pacific bluefin tuna, with MSC assigning a low risk, Patrick medium risk (Figs. 2, 3, 5). Again, different cutoffs for shared attributes contributed to the different results, such as for age at maturity (Tables S13, S25). And scores of attributes unique to each approach also contributed to the inconsistent PSA relative risk outcomes for this stock, where, for instance, MSC's reproductive strategy and selectivity attributes assigned low risk scores, and Patrick's growth coefficient, morphological characteristics (selectivity), and breeding strategy attributes assigned high, high and medium risk scores, respectively.

Theoretically, if PSAs include attributes with multiple definitions designed correctly to account for different life histories of assessed groups, as the original PSAs such as by Stobutzki et al. [23] were designed, they will provide estimates of relative vulnerability that are comparable within and across taxonomic groups. The two PSA approaches included here, however, are not designed in this manner. For instance, we encountered several problems with applying the MSC PSA approach due to attribute definitions that were unsuitable for this cross-taxa assessment of relative risks for species captured in pelagic longline gear, as well as due to a lack of explicit guidelines on how certain attributes should be applied. Section S.2 describes modifications made to MSC PSA attributes to address some of these problems. Section S2.3 identifies additional issues identified with the MSC PSA approach that were not adapted for this study. Amending the MSC RBF to rectify these deficits would improve consistency in application and augment the ability to distinguish between relative risk of species susceptible to capture in pelagic longline fisheries. As MSC has acknowledged, the cutoffs for some of the PSA attributes may require adjustment for some taxonomic groups (GPF4.3.2 in MSC [38]). The definitions of some of the attributes may also require modifications to make them suitable for assessing relative risks between as well as within taxonomic groups (Section S2.2).

4.3. Improving data quality

Addressing substantial deficits in data quality would support more robust semi-quantitative as well as quantitative risk assessments of the Tunago fishery, and more broadly, would enable assessing the effects of regional, cumulative anthropogenic mortality sources. Locally, a substantially larger monitoring dataset from the Tunago fleet is needed. Due to monitoring data being available from only a single trip by a Tunago vessel, the study also used data from the full Vanuatu-flagged longline fishery as a proxy. This likely substantially overestimated areal overlap between tropical and sub-tropical species and the Tunago fishery, which targets albacore and bigeye tunas at higher latitude, temperate zones.

In addition, by basing the risk assessment on extremely limited monitoring data, it is very likely that rare species susceptible to capture in pelagic longline gear, some of which may be relatively vulnerable, were not identified and included in this assessment. Species richness and other species-level biodiversity indices are extremely sensitive to sample size and species abundance distribution (evenness). The less even the relative abundance of species in a community is, the larger the proportion of relatively rarer species within that system will be detected with more sampling effort [109–111].

Furthermore, substantially improved data quantity and quality are needed to support effective integrated bycatch management and to develop and apply population-specific, multispecies or ecosystem-level harvest strategies [31,112]. For example, to identify where and when vulnerable bycatch occurs so that area-based bycatch management tools could be considered, data requirements to develop a spatially-explicit model with standardized effort would need to be met [31]. This would enable identifying how temporally and spatially predictable dynamic as well as static sites of vulnerable bycatch hotspots could be avoided [113–115]. And, for some species, information on the provenance of bycatch is required to determine which populations are affected by the fishery [116].

A more robust risk assessment as well as effective bycatch management framework also requires filling information gaps on contemporary fishing practices and gear designs. Information was available for a small subset of potentially significant explanatory factors for catch risk and probability of post-capture survival. For example, information is needed on gear characteristics (e.g., branchline weighting design, length of branchlines and floatlines), fishing methods (lazy lines, bait thaw condition, management of offal and spent bait, proximity to shallow submerged features), and handling and release practices [117,118]. An assessment of compliance with voluntary company policies on leader material and shark lines (branchlines attached to floats or floatlines to target epipelagic sharks), monitoring data on hook and bait types, and more robust assessment of compliance with the national fisheries management authorities' required use of tori lines and night setting in areas where seabird bycatch measures are required to be employed [42] are additional priorities. These data quality improvements could be achieved through a combination of a dockside inventory, conventional at-sea human observer coverage and expanded EM coverage [31].

However, the Tunago EM system requires improvements. When properly designed, EM systems have several advantages over conventional human observer programs, including overcoming main sources of statistical sampling bias, enabling multiple areas of vessels to be monitored simultaneously and near-continuously and allowing questionable data to be audited [119,120]. Consistent with findings from an assessment of the same EM system being used in a different longline fishery [118], the EM system of the Tunago fishery needs to be modified to improve the quality of discards data on fields for the species, length, at-vessel condition and release condition. For instance, the EM analyst was unable to identify over 49% of discarded catch to the species level (Table S1). Dockside collection of data fields that are challenging for EM analysts to collect and that significantly explain catch and mortality risk (e.g., mass of branchline weights, leader length) is an additional improvement priority.

There are gaps in fundamental biological and ecological information for some of populations susceptible to capture in pelagic longline fisheries. For instance, information on fecundity in some populations of principal market tunas and other scombrids is lacking [121]. Basic demographic information, such as adult and juvenile survival rates, are lacking for most seabird species [122] and for several marine turtle populations [123]. Over 46% of elasmobranch species are considered data deficient [9]. Filling gaps in accurate life history attributes is a priority to support robust quantitative as well as semi-quantitative ERAs.

Findings from semi-quantitative ERAs guide decisions on which stocks and populations warrant more rigorous, quantitative risk assessments to determine absolute risks from the Tunago fishery. Data requirements for these more robust, model-based, quantitative ERAs, however, are often not able to be met. Due to large gaps in key information for many marine species, it would be challenging to construct robust population models, especially for the long-lived species exposed to this fishery. In addition to filling gaps in life history traits, substantially more robust estimates of total anthropogenic mortality of individual populations are needed. This includes estimating the age-class and sex of total anthropogenic removals; estimating indirect, collateral sources of anthropogenic mortality such as from reducing optimal fish school sizes and outcomes of human-induced climate change; and population-level effects from broad community- and ecosystem-level responses to individual and interacting anthropogenic stressors [124–126]. These deficits highlight the tremendous capacity for improved certainty of estimates of relative and absolute ecological risks of the effects of fishing.

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CRediT authorship contribution statement

Eric Gilman: Conceptualization, Formal analysis, Methodology, Project administration, Writing – original draft, Writing – review & editing. **Milani Chaloupka:** Formal analysis, Methodology, Writing – review & editing. **Chrissie Sieben:** Formal analysis, Methodology, Writing – review & editing.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.marpol.2021.104752.

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