INTER-AMERICAN TROPICAL TUNA COMMISSION

SCIENTIFIC ADVISORY COMMITTEE

NINTH MEETING

La Jolla, California (USA)
14-18 May 2018

DOCUMENT SAC-09-12

DEVELOPMENT OF A FLEXIBLE ECOLOGICAL RISK ASSESSMENT (ERA) APPROACH FOR QUANTIFYING THE CUMULATIVE IMPACTS OF FISHERIES ON BYCATCH SPECIES IN THE EASTERN PACIFIC OCEAN

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ABSTRACT

Ecological Risk Assessment (ERA) methods have been an increasingly popular alternative to traditional stock assessments for rapidly and cost-effectively assessing the relative vulnerability of non-target species in resource- and data-limited fisheries. The widely-used Productivity-Susceptibility Analysis (PSA) requires detailed fishery susceptibility and biological information for a large number of parameters, and cannot definitely determine species vulnerability or quantify cumulative impacts from multiple fisheries. This

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paper introduces a flexible quantitative approach that uses less input parameters than PSA to quantify the cumulative impacts of multiple fisheries on data-poor bycatch species. The method first produces a proxy of the instantaneous fishing mortality rate (F) of each species based on the 'volumetric overlap' of each fishery with the stock's distribution. F is then used in length-structured per-recruit models to assess the vulnerability of each species using conventional biological reference points (*e.g*. F_{MSY}, $F_{0.1}$ and $\mathrm{SSB}_{40\%}$). The method is illustrated with data from the eastern Pacific Ocean (EPO) tuna longline and purse-seine fisheries. Application of the method to 14 pelagic and mesopelagic teleost and elasmobranch non-target species, and classification of the vulnerability status of each species using a phase plot, is demonstrated. This approach may allow fisheries managers to more confidently identify the most vulnerable species to which resources can be directed to either implement mitigation measures, apply more detailed analysis, or collect further data to facilitate a formal stock assessment in the future.

INTRODUCTION

The Inter-American Tropical Tuna Commission (IATTC) has adopted an ecosystem-based approach to the management of tuna fisheries in the eastern Pacific Ocean (EPO) through its commitment to ensuring the long-term sustainability of the stocks of tuna and tuna-like species, associated non-target species and the supporting ecosystems through the adoption of the Antigua Convention, in particular Article VII 1(f) "*adopt, as necessary, conservation and management measures and recommendations for species belonging to the same ecosystem and that are affected by fishing for, or dependent on or associated with, the fish stocks covered by this Convention…*". Furthermore, in 2018 the IATTC has developed a Strategic Science Plan (SSP) with an explicit goal to "*evaluate the ecological impacts of tuna fisheries",* primarily by identifying species at risk and prioritizing them for data collection, research and management. However, ecological sustainability can be difficult and expensive to demonstrate in practice, due to the common paucity of reliable biological and catch information for non-target species, especially those of little or no commercial value. Therefore, assessing all impacted species using traditional stock assessment approaches is both cost-prohibitive and impractical.

An alternative approach, particularly for data-limited fisheries, is ecological risk assessment (ERA), a range of qualitative to quantitative methods that are now used by organizations worldwide as a rapid and cost-effective approach for assessing the ecological impacts of fishing (Gallagher *et al.* 2012). In contrast to conventional stock assessments, which aim to precisely determine the status of a fished stock, the primary objective of ERA is to rapidly identify potentially vulnerable species and prioritize them for further data collection, to subsequently facilitate more rigorous quantitative assessment and develop specific management measures (*e.g*. gear modifications, time-area closures, *etc*.) that can reduce a species' vulnerability. Therefore, incurring false positives (*i.e.* classifying as "most vulnerable" when another classification is true) is preferable to incurring false negatives (*i.e*. classifying a species as something other than "most vulnerable" when the latter is true) (Hobday *et al.* 2011).

The semi-quantitative Productivity-Susceptibility Analysis (PSA) (Milton 2001, Stobutzki *et al.* 2001) has been a particularly popular ERA method, applied to a range of species groups and fisheries worldwide, (Lucena-Frédou *et al.* 2016) (Milton *et al.* 2008) (Cortés *et al.* 2010, Arrizabalaga *et al.* 2011) (Waugh *et al.* 2008). PSA estimates the relative vulnerability of each impacted species by using a categorical score (1-3) for attributes relating to a species' susceptibility to being caught (*e.g.* gear selectivity) and its biological productivity (*e.g.* natural mortality rate). The scores for susceptibility and productivity attributes for each species are averaged, and then combined to produce an overall vulnerability (v) score from 1 (least vulnerable) to 3 (most vulnerable). An arbitrary threshold score (*e.g*. $v > 2.0$; see Cope *et al.* 2011) is then used to classify species as "high risk".

Unfortunately, these thresholds have no biological significance, nor have they been statistically derived. This is because input data—even highly precise parameter estimates—are reduced to categorical scores.

As a result, PSA scores provide only a *relative measure of vulnerability* for a single fishery, as the scores for separate fisheries cannot be summed to assess the cumulative impacts of multiple fisheries. This presents a conundrum for fisheries managers, who may wish to establish formal PSA reference points (*e.g.* v = 2.0) to initiate a management response. Such arbitrary thresholds can therefore cost a fishery valuable resources by triggering mitigation measures for species that have only been classified as vulnerable due to artifacts of the assessment method. This highlights the need for improved assessment methods and biologically meaningful reference points to definitively determine the vulnerability of data-poor species.

To address this issue, and the objective of the IATTC SSP to "*develop analytical tools to identify and prioritize species at risk*", this paper introduces a flexible spatially-explicit quantitative ecological risk assessment approach—**E**cological **A**ssessment of **S**ustainable **I**mpacts of **Fish**eries (EASI-Fish)—that quantifies the cumulative impacts of multiple fisheries on data-poor bycatch species. EASI-Fish first provides a proxy of fishing mortality from the "volumetric overlap" of multiple fisheries on a species' three-dimensional spatial distribution, which is then used in length-structured per-recruit models to determine the species' vulnerability status using conventional and precautionary biological reference points. Using the longline and purse-seine tuna fisheries in the EPO as a case study, this paper demonstrates the application of the method to a range of pelagic and mesopelagic teleost and elasmobranch non-target species caught in these fisheries for the purpose of identifying and prioritizing species for management, or for the collection of further information that will fill key data gaps and allow for more formal population assessments to be undertaken in the future.

1. METHODS

1.1. Spatial extent of the assessment region and definition of included fisheries

Although EASI-Fish was designed to be used in data-limited fisheries, it has sufficient flexibility to utilize a range of data types of varying quality. For demonstration purposes, the approach is applied to a reasonably data-rich setting to assess the vulnerability of a representative group of species (target, retained non-target, and discarded non-target) caught in the purse-seine fishery and in the fishery by large-scale longline tuna fishing vessels (LSTLFVs) (herein called the "longline fishery") in the EPO (defined as the region from the coast of the Americas to 150°W between 50°S and 50°N).

The analyses presented in this paper draw upon data obtained from vessel logbooks or collected by on-board scientific observers. or submitted to the IATTC by its Members under Resolutions C-03-05 and C-11-08 and described in Document SAC-08-07b. Specifically, the longline fishery data were derived from vessels over >24 m length overall (LOA) included in the IATTC Regional Vessel Register that are authorized to fish for tuna and tuna-like species, which provide monthly reports of catch and fishing effort at a resolution of at least 5°x5°, and from scientific observer programs that monitor at least 5% of the fishing effort by longline vessels over 20 m LOA under Resolution C-11-08.

The purse-seine fishery data were collected by the on-board observer program of the Agreement on the International Dolphin Conservation Program (AIDCP), which covers 100% of the fishing effort by Class-6 (carrying capacity >363 t) purse-seine vessels. This fishery was disaggregated into three separate fisheries based on set type: i) sets associated with floating objects (OBJ), ii) sets associated with dolphins (DEL), and ii) sets on unassociated schools of tuna (NOA).

1.2. Assessing susceptibility as a proxy for instantaneous fishing mortality (*F*)

Similar to most other ERA approaches such as PSA, EASI-Fish is comprised of separate susceptibility and productivity components. The susceptibility component in EASI-Fish is used to approximate the finite mortality rate (f), which is then converted to an approximation of the instantaneous fishing mortality rate

(*F*) and is compared to biological reference points (RPs) used in the productivity component—length-structured yield and biomass per-recruit models. Some (or even all) of the fisheries that contribute to the species' cumulative mortality effect across fisheries may be data-limited and lack species-specific catch data to estimate *F* for data-poor bycatch species. In such cases, EASI-Fish estimates the proportion of the population that is susceptible to being captured from the extent of the overlap of a species' horizontal and vertical distribution with each fishery.

EASI-Fish uses a length-based approach with similar susceptibility parameters as previous risk assessment approaches (which treat a species' population as a single biomass pool) to estimate the proportion of a length class (*j*) of a species' population that is susceptible to incurring mortality by fishery *x* (S_{xj}) in a given year, which can be represented as:

$$S_{xj} = \frac{G_x}{G}\left(D_x A_{xj} N_{xj} C_{xj} P_{xj}\right)$$

(Eq. 1)

where *G* is the total number of grid cells occupied by a species (*i.e*. the "stock"), and G_x is the number of occupied grid cells containing at least one unit of fishing effort by fishery *x* during a specified year. For the purposes of this study, the stock definition for each species, within which its distribution is estimated, was assumed to be defined by the boundaries of the EPO.

In this study, *G* was estimated from environment envelope models developed for each species at 0.5° x 0.5° resolution, using the method of Kaschner *et al.* (2006) and a knife-edge probability-of-occupation threshold of 60% for each cell.

Fishing effort for each fishery in 2016 was overlaid on each species' distribution map to calculate G_x, and the percentage overlap of each fishery was calculated by dividing G_x by *G.* Effort data for purse-seine vessels were reported at <0.5° resolution, so each set location was allocated to the corresponding 0.5° grid cell. In contrast, longline data for 2016 were reported at 5° x 5° resolution, so the longline grid conservatively assumes that there was at least one unit of effort in each occupied 0.5° x 0.5° cell contained in a 5° x 5° cell with effort.

The first four parameters in the parentheses of Equation 1 (D_x, A_{xj}, N_{xj}, and C_{xj}) comprise what is generically regarded as "selectivity" in fisheries stock assessments, which combines—often implicitly— "population availability" (the relative probability that a fish of length class *j* is located in the area and time where the fishery is operating) and "contact selectivity" (the relative probability that a fish of length class *j* will be retained once it comes in contact with the gear) (Millar and Fryer 1999). Because selectivity curves are unlikely to be available for data-poor bycatch species, it was considered important to disaggregate selectivity components as far as practicable. This also allows the individual components to be parameterized if information is available, or the default assumption of full selection to be implemented as a precautionary measure in the absence of reliable information.

Fishing season duration (D_x) is the proportion of the population that is available to fishery *x* given the proportion of a year when fishing is permitted, expressed as the number of fishing days divided by 365. In the EPO, Resolution C-13-01 mandated a 62-day closure of the purse-seine fishery in 2016, meaning that the species was potentially exposed to purse-seine fishing for 0.83 (365-62/365) of the year. Fishing effort is assumed to be evenly distributed throughout the year, since fishing mortality would be different in a fishery in which all the fish could be caught in 100 days, for example, rather than the full 365 days. The default precautionary value is 1.0 for fishery *x*, to assume that the species is available to fishery *x* for the entire year. In the case of the EPO, the value of 0.83 is actually precautionary, because we assume that during the period of fishery operation the species is fully exposed.

Seasonal availability (A_{xj}) is the proportion of length class *j* that is available to capture by fishery *x*, given that some species undertake extensive intra-annual migrations outside the boundaries of the fishery, where they are unavailable for fishery interactions (Rowat and Brooks 2012). This is expressed as the proportion of the year that the species is available to the fishery. In cases where migrations are known to occur, but are not adequately quantified, broad categories may be used (*e.g*. 1-3 months = 0.25, 4-6 = 0.5, 7-9 = 0.75, 10-12 = 1.0). The default precautionary value is 1.0 for fishery *x*, to assume that no seasonal movement outside of fishery *x* occurs for length class *j*.

Encounterability (N_{xj}) is the proportion of length class *j* that may potentially encounter the gear used by fishery *x* based on the species' distribution in the water column relative to the normal fishing depth range of the gear. In the EPO, we defined the effective fishing depth range for all purse-seine set types as 0-200 m (Hall and Roman 2013) and 0-300 m for 'deep sets' by longlines (see Griffiths *et al.* (2017a). Minimum, maximum, and preferred depths of each species were defined using the results of published studies using electronic tags (*e.g*., Schaefer and Fuller 2010), longline fishing experiments using time-depth recorders (TDRs) (Boggs 1992), or relating catch to estimated maximum hook depths (Nakano *et al.* 1997, Ward and Myers 2005, Zhu *et al.* 2012). Available depth ranges are often independent of length or age classes (but see Kitagawa *et al.* 2007, Childers *et al.* 2011), so the proportional overlap of the species' distribution with the fishery could be assumed to be constant across all length classes. The default precautionary value is 1.0 for fishery *x*, thus assuming that all of length class *j* encounters the gear for fishery *x*. A graphical representation of the encounterability concept is shown in Figure 1.

Although a species may encounter the gear due to overlapping vertical and spatial distribution with the fishery, this does not mean all fish will be caught and incur mortality. Contact selectivity (C_{xj}) describes the proportion of length class *j* that is retained once it encounters the gear used by fishery *x*. Typical gear selectivity curves can be used where available; for example, dome-shaped or logistic ogives to represent net and longline fisheries, respectively. However, reliable gear selectivity curves will not be available for the majority of bycatch species. In such cases, full knife-edge selectivity (C_{xj}=1.0) may be assumed from the smallest length class fish observed in fishery *x*. However, the default precautionary value is 1.0 for fishery *x*, to assume full selectivity for all size classes in fishery *x*.

Although the proportion of length class *j* that is impacted by fishery *x* may be very high, given its geographic and vertical distribution relative to the fishing gear of fishery *x*, the species may be discarded, either due to its low market value, or for conservation reasons. For example, the IATTC mandates the release of oceanic whitetip sharks (Resolution C-11-10) and Mobulid rays (C-15-04) in all fisheries, and the release of silky sharks in the purse-seine fishery (C-16-06). Therefore, fishing mortality would be overestimated unless the component of the catch that survives release is accounted for. This is introduced in the model as post-release mortality (P_{xj}), the proportion of length class *j* that is caught by fishery *x* and dies before, during, or soon after release. Where species-specific data is not available, the default precautionary value is 1.0 for fishery *x*, to assume a post-release mortality rate of 100%.

Following the estimation of the overall susceptibility of length class *j* to incurring mortality from fishery *x* (S_{xj}), a proxy for the instantaneous fishing mortality rate (*F*), which is required to assess the vulnerability of a species against RPs in the per-recruit models, can be estimated from the annual finite fishing mortality rate (*f*)—or exploitation rate—for the species caught by all fisheries in a specified year as:

$$F = -\ln\left[1 - \sum_{x=1} q_x\, E_x \left(\frac{\sum_{j=1}^{n} S_{xj}}{n}\right)\right]$$

(Eq. 2)

Here, *n* is the number of length classes defined for the species, fishing effort (E_x) is the total effort, scaled to a maximum of 1, of fishery *x* applied in area G_x, while the catchability coefficient (q_x) is the fraction of the stock that is caught by one unit of effort (E_x) in fishery *x*. In many data-limited fisheries *q* and *E* will not be known, so a precautionary approach is to assume both are equal to 1, so that at some level all the fish are caught. However, this assumes that all fisheries have the same impact. For example, a low-effort fishery with a wide distribution has a greater impact than a high-effort fishery with a more concentrated distribution, ignoring all other factors. Therefore, some consideration should be given to the relative magnitude of q_xE_x where possible. For example, if q_xE_x is an order of magnitude less than another fishery, then its impact may be expected to be much smaller, and could be down-weighted, or potentially excluded from the calculation.

The *F* value for the assessment year, in this case 2016 (F_{2016}) is then compared with values for *F* for the various RPs derived from the per-recruit model (described below). However, it needs to be emphasized that, because of the assumptions and likely uncertainty in the parameters used in deriving the *F* estimate, it should only be considered a proxy of *F* (and probably a conservatively high one). It is for this reason that the results from EASI-Fish should not be used to definitively define the status of a species' population, *sensu* a stock assessment. EASI-Fish is a quantitative prioritization tool to identify the most vulnerable species that should then be considered for data collection, further detailed analysis, research and management.

1.3. Characterizing species productivity using per-recruit models

One of the major impediments for quantitative assessments of the population status of data-poor species is the lack of species-specific time series of catch data required for fully-integrated stock assessment models, such as those used for the EPO yellowfin and bigeye assessments. However, the yield-per-recruit (Y/R) model of Beverton and Holt (1957) is widely used in developing or data-limited fisheries due to the relatively few parameters that need to be estimated, most of which also need to be estimated for ERA methods such as PSA. However, Y/R models are generally age-structured, which poses complications with most data-limited species and fisheries, since most biological and fishery processes are more precisely represented in terms of length rather than age (*e.g*. length-weight relationships, length-at-maturity, gear selectivity) (Chen and Gordon 1997). Many bycatch species also lack reliable biological studies that describe their growth or population dynamics in terms of age. Furthermore, many age-specific parameters are often estimated from length-based conversions, such as estimated length-at-age in von Bertalanffy growth models. Therefore, it is more practical to construct per-recruit models based on length, rather than age.

Y/R is used to characterise the biological dynamics of each species using the generic Ricker (1975) model, which Chen and Gordon (1997) adapted for lengths as:

$$\frac{Y}{R} = \sum_{j=1}^{n} \frac{W_j b_j F}{b_j F + M} \left[1 - e^{-(b_j F + M)\Delta T_j}\right] e^{-\sum_{k=1}^{j-1}(b_k F + M)\Delta T_k}$$

(Eq. 3)

Here, new recruits and fully-recruited length classes are denoted by the subscripts *j* and *k*, respectively. W_j is the mean weight of a fish in length class *j*, while selectivity (b_j) is the proportion of the population in length class *j* that is caught across all fisheries, represented as:

$$b_j = \sum_{x=1}^{n} S_{xj}$$

(Eq. 4)

The instantaneous natural mortality rate (*M*) is assumed to be constant across all length classes, but can be length-specific if sufficient data are available. *F* is disaggregated into increments of 0.01, from zero to a biologically-realistic maximum for a species. The parameter ΔT represents the time taken for a fish to grow from one length class to the next, represented as:

$$\Delta T_j = \frac{1}{K} ln \frac{L_\infty - L_j}{L_\infty - L_j - d_j}$$

(Eq. 5)

where K and L_∞ are parameters from the von Bertalanffy growth function, and d is the width of the length class, calculated as $L_{j+1} - L_j$. In the absence of direct estimates of K and L_∞, L_∞ may be estimated from maximum recorded length (L_{max}) (Froese and Binohlan 2000) and K values may be used from studies of closely related species, as is the procedure in PSA.

The spawning stock biomass-per-recruit (SSB/R) model of Quinn and Deriso (1999) is complementary to Y/R, and can be modified to suit the analysis of length rather than age classes and be represented as:

$$\frac{SSB}{R} = \sum_{j=1}^{n} W_j m_j \prod_{x=r}^{j-1} e^{-(b_j F + M)}$$

(Eq. 6)

where W_j is the mean weight of fish in length class j (L_j), m_j is the proportion of mature females at the mean length of length class j, and the product operator describes the number of fish surviving from the length at recruitment (L_r) to L_j. Because the number of spawners is unlikely to be known for most bycatch species, and the model estimates the relative SSB/R, the initial number of spawners is set to a value of one. The value for m_j is taken from a female maturity ogive for the species, represented in the generic logistic form:

$$m_j = \frac{1}{1 + e^{\left(-r(L_j - L_{50})\right)}}$$

(Eq. 10)

where L_j is the mean length of a fish in length class j, L_{50} is the length at which 50% of the population is mature, and r is the curvature parameter. Alternatively, knife-edge maturity can be assumed from the length at first maturity (L_{m}). If a direct or reliable published estimate of L_{50} or L_{m} is unavailable, L_{m} can be estimated (with error) using the von Bertalanffy parameter L_∞ in the empirical equation of Froese and Binohlan (2000):

$$L_m = 10^{(0.898\, log_{10} L_\infty - 0.0781)}$$

(Eq. 11)

Where the corresponding 95% confidence interval is:

$$95\%\ CI = L_\infty\ {}^{+}/{-} \left(1.965 \cdot 0.358 \cdot \sqrt{\left(0.00214 + (0.0135 \cdot (log_{10}(L_\infty) \cdot 1.689)^2)\right)}\right)$$

1.4. Estimating natural mortality

The instantaneous natural mortality rate (*M*) is one of the most influential parameters in stock assessment models, but is notoriously difficult to estimate directly (Kenchington 2014, Then *et al.* 2015). Therefore, it is commonplace to run stock assessment models using a range of *M* values derived from multiple estimators based on life history invariants. Therefore, *M* was calculated for each species using six estimators recommended by Kenchington (2014) and Then *et al.* (2015) (Table 1). Priority was given to *M* values that were estimated directly (*e.g.* from tagging), followed by t_{max}-based estimators ($\text{Hoenig}_{\text{nls}}$ and

Hoenig$_{t\mathrm{max}}$), and finally K-based estimators (Jensen, Pauly$_{\mathrm{nls\text{-}T}}$, Pauly$_{\mathrm{LKT}}$ and Pauly$_{\mathrm{KT}}$). Where species lacked information on longevity and L_∞, L_∞ was estimated from maximum recorded length (L_{max}), using the method of Froese and Binohlan (2000), and used in the Pauly$_{\mathrm{LT}}$ estimator with a mean annual water temperature of 25˚C for the EPO (Fiedler and Talley 2006) (Table 1).

2. BIOLOGICAL REFERENCE POINTS

A number of reference points (RPs) can be used in stock assessment models to assess the status of a population relative to a fishing mortality rate. EASI-Fish uses a similar approach, but it is important to reiterate that RPs here are used to quantify the relative vulnerability of a population to decline, rather than to evaluate stock status.

The instantaneous fishing mortality rate at maximum sustainable yield (F_{MSY}) is used, because it is a commonly-used RP in stock assessment. In Y/R models, it is, more specifically, the fishing mortality at which yield is maximized (F_{MAX}), since there is no stock-recruitment relationship. However, $\mathrm{F}_{\mathrm{MAX}}$-based RPs can be overly optimistic, and so $F_{0.1}$ was used as a precautionary RP, as has been recommended for data-limited species or fisheries (Gabriel and Mace 1999). A second precautionary RP used was $F_{40\%}$, which is the F value corresponding to 40% of the spawning potential ratio (SPR; the SSB/R at a given fishing mortality divided by the SSB/R where F=0). The corresponding $\mathrm{SSB}_{40\%}$ is also used, which is the relative SSB/R at $F_{40\%}$.

The final assessment of the vulnerability of each species was determined using F_{2016} and the corresponding SSB-per-recruit value (SSB_{2016}), each expressed relative to the MSY or precautionary RPs ($F_{0.1}$ and $\mathrm{SSB}_{40\%}$). To aid in the interpretation of the results, RP values are represented on the four-quadrant phase (or Kobe) plot that is widely used to display the results of traditional stock assessments, and therefore easy to interpret for most fisheries managers and researchers. The traditional definitions of these quadrants relate to the status of a stock as being “overfished” or “undergoing overfishing”. In contrast, EASI-Fish uses generic definitions of vulnerability to define each quadrant, in order to reflect the uncertainty in model parameters for data-poor species. Figure 5 shows the vulnerability definitions of each quadrant in the EASI-Fish phase plot: i) “Least vulnerable” (green; $F_{2016}/F_{\mathrm{MSY}}<1$ and $\mathrm{SSB}_{2016}/\mathrm{SSB}_{\mathrm{MSY}}>1$), ii) “Increasingly vulnerable” (orange; $F_{2016}/F_{\mathrm{MSY}}>1$ and $\mathrm{SSB}_{2016}/\mathrm{SSB}_{\mathrm{MSY}}>1$), iii) “Most vulnerable” (red; $F_{2016}/F_{\mathrm{MSY}}>1$ and $\mathrm{SSB}_{2016}/\mathrm{SSB}_{\mathrm{MSY}}<1$), and iv) “Decreasingly vulnerable” (yellow; $F_{2016}/F_{\mathrm{MSY}}<1$ and $\mathrm{SSB}_{2016}/\mathrm{SSB}_{\mathrm{MSY}}<1$).

Two simple alternative RPs were proposed by Zhou and Griffiths (2008) for data-limited bycatch species using the Sustainability Assessment for Fishing Effects (SAFE) approach, and later refined by Zhou *et al.* (2012) using life history invariants, take on the form: $F_{\mathrm{msm}}=\omega M$ and $F_{\mathrm{crash}}=2\omega M$, where ω is 0.41 and 0.87 for chondrichthyans and teleosts, respectively. Zhou and Griffiths (2008) showed that F_{msm} is the minimum sustainable fishing mortality, and may be a proxy for F_{MSY}, while F_{crash} is the minimum mortality rate that would eventually render the population extinct. Since the yield-per-recruit model used in EASI-Fish produces actual estimates of F_{MSY} (=F_{MAX}), F_{msm} and F_{crash} were also included in the results as a comparison of their performance.

2.1. Implementation of the model

In addition to providing a routine means for the IATTC staff to prioritize vulnerable species for data collection, further detailed analysis, research and management, EASI-Fish was designed to be an inexpensive and user-friendly tool that can be used by researchers and managers with minimal quantitative modelling experience, but who understand the principles of population dynamics models. Therefore, it was built in Microsoft Excel, with add-ins to perform Monte Carlo simulations to generate uncertainty estimates for each model parameter given specified prior distributions (*e.g.*, normal, triangular, or uniform). Once the parameter distributions were defined, the Y/R and SBB/R models were run 10,000 times using Monte Carlo simulations, each time using a random sample from the distributions of each parameter. The mean, standard error (SE), and 95% confidence intervals (95% CI) were derived for RPs F_{MSY}, $F_{0.1}$, $F_{40\%}$, SSB_{2016}, $\mathrm{SSB}_{\mathrm{MSY}}$, $\mathrm{SSB}_{0.1}$, and $\mathrm{SSB}_{40\%}$. IATTC biological staff undertook extensive literature reviews for each species impacted by EPO fisheries to pre-fill EASI-Fish with the best available model parameters. However, any parameter can easily be overridden with a value deemed most appropriate by the user in the main model graphical user interface (Figure 2).

2.2. Qualitative scoring of parameter data source quality

Although parameter uncertainty is incorporated into the EASI-Fish model, this does not necessarily indicate the precision, reliability, or relevance of the value to the fishery in which it is applied. For example, the population dynamics of many fish species differ on various spatial scales, from ocean basins (Griffiths *et al.* 2010) to region (Williams *et al.* 2012), so the application of biological parameters derived from one region, regardless of the quality of the study, may not be appropriate in a model of the same species in a different region. Of course, in the absence of local information, a common situation for bycatch species, the use of non-local studies may be required. However, a measure of the relevance and quality of parameter values is required to quickly determine the reliability of the model results, which will be important in situations where ERAs may contain a large number of species that are classified as "most vulnerable".

A parameter quality index was developed to score the relevance of the data to the assessed fisheries and species by using a matrix of data quality by ocean basin and taxonomic resolution (Table 2). The parameter quality scores are represented in a single radar plot for each species, aiding in the easy interpretation of a large number of model parameters.

3. RESULTS

3.1. Estimates of susceptibility (*S*)

Habitat models were developed for 14 species caught as target species (4 species; Figure 3) or non-target shark (6 species; Figure 4) or teleost species (4 species; Figure 5) in EPO tuna fisheries. A prominent feature of the distributions of most of these species is the low-to-zero probability of occupancy in the central South Pacific Subtropical Gyre at around 20-30°S, which is characterized by a deep (>200 m) pycnocline (Fiedler and Talley 2006).

For each species, the value for each parameter contributing to the overall susceptibility (S_{xj}) estimate and a description of its derivation is given in Table 3. The horizontal overlap of the longline fishery with the distribution of the species assessed was high, ranging between 48% (*Lepidocybium flavobrunneum*) and 79% (*Kajikia audax*). It is important to note that this may also be due to the longline fishery reporting effort at 5° x 5° resolution, and thus encompassing more 0.5° x 0.5° grids that would be the case if effort were reported at a higher resolution. In contrast, the highest species overlap for the purse-seine fishery was from DEL sets on sharks (*Carcharhinus falciformis*, 30%; *Alopias superciliosus*, 28%), and target tuna species (*Thunnus albacares* and *T. obesus*, 19%).

3.2. Vulnerability of selected species in the EPO

The biological parameter values used in empirical equations, and the per-recruit models for the 14 species and their sources, are shown in Tables 4 and 5, respectively, while EASI-Fish estimates for the fishing mortality and spawning stock biomass estimators and RPs are provided in Table 6.

F_{2016} and SSB_{2016} for the 8 teleost species did not exceed the F_{MSY} and $\mathrm{SSB}_{\mathrm{MSY}}$ RPs, respectively, and they were therefore classified as "least vulnerable". However, five species (*T. albacares, T. obesus*, *K. audax, Coryphaena hippurus,* and *Acanthocybium solandri*) were classified as "most vulnerable" using the precautionary $F_{0.1}$ RP for the first three species, and the precautionary $\mathrm{SSB}_{40\%}$ RP for the latter two. Two species (*Lepidocybium flavobrunneum* and *Lampris guttatus*) were classified as "least vulnerable" by every RP (Table 6).

In contrast, F_{2016} and SSB_{2016} for all shark species exceeded the F_{MSY} and precautionary $F_{0.1}$ and $SSB_{40\%}$ RPs and were classified as "most vulnerable", with the exception of *Prionace glauca*, which was classified as "least vulnerable" under the F_{MSY} and SSB_{MSY} RPs (Table 6).

The SAFE RP F_{msm}—a proxy for F_{MSY}—was generally less than half that of the estimates of F_{MSY} from EASI-Fish, which would have resulted in six of the species (*T. albacares, T. obesus*, *K. audax, C. hippurus, A. solandri* and *P. glauca*) classified as "least vulnerable" by EASI-Fish being regarded as "most vulnerable". Using the less conservative F_{msm} RP, F_{crash}, four species (*T. albacares, T. obesus*, *K. audax,* and *P. glauca*) would be classified as "most vulnerable" (Table 6).

When viewing the EASI-Fish results as a catch assemblage on a phase plot for the MSY RPs (Figure 7a), it is immediately obvious that the shark species, excluding *P. glauca*, are classified as "most vulnerable" and should invoke the most immediate management attention. If the more precautionary RPs are used, the next least productive species (*e.g.*, *P. glauca*) or species experiencing high fishing mortality (*e.g.*, *T. albacares, T. obsesus*, *K. audax*) move from being "least vulnerable" to "decreasingly vulnerable" or "increasingly vulnerable" for the $F_{0.1}/SSB_{0.1}$ (Figure 7b) and $F_{40\%}/SSB_{40\%}$ RPs (Figure 7c).

The radar plots in Figure 8 show that, of the five most vulnerable species with respect to the F_{MSY}/SSB_{MSY} RPs (Figure 7a), the four most vulnerable species have data reliability scores of 8 or more for each parameter, and can therefore be regarded as legitimate "most vulnerable" species. In contrast, the fifth most vulnerable species (*Sphyrna zygaena*) has scores of 0-4 for reproductive parameters and a low-quality estimate of natural mortality, which together may have overestimated the vulnerability of this species. On the other hand, *L. flavobrunneum* and *L. guttatus* were among the least vulnerable species *prima facie* (Figure 7a), but the former lacked reliable data for all growth and reproductive parameters, while the latter lacked a reliable length-at-maturity ogive to estimate L_{50} (Figure 8), which would likely increase the length at maturity, and thus likely reduce the values for SSB RPs.

4. DISCUSSION

The primary objective of ecological risk assessment (ERA) in fisheries is to identify, rapidly and cost-effectively, species most vulnerable to fishing impacts, in order to guide the development of mitigation measures to reduce or eliminate those impacts, or to collect further information to facilitate more formal stock assessment (Hobday *et al.* 2011). Therefore, outcomes from ERA models do not need to be precise, but the models need to be reasonably accurate at defining relative vulnerability among species, repeatable, and transferable between species with different life histories (*e.g.*, teleosts to marine mammals)—although the choice of reference point may vary. To this point, most ERA methods produce only a relative indicator of risk based on categorical values for attributes describing a species' susceptibility and productivity, without biologically meaningful reference points with which to definitively determine the vulnerability of a population. Such attributes are often added or removed, and scores weighted, in an *ad hoc* manner, with little statistical demonstration of the impacts on overall vulnerability scores due to biases from autocorrelated attributes (Duffy and Griffiths 2017). Furthermore, most ERA methods, in their current form, are capable only of fishery-by-fishery assessments, and do not allow the cumulative effects of multiple fisheries impacts to be quantified, thereby underestimating a species' vulnerability. EASI-Fish overcomes these significant shortcomings, while using fewer data inputs than the widely-used PSA method (Table 7), and quantifies the cumulative impacts of fisheries using conventional and scientifically defensible fishing mortality and spawning biomass RPs that are familiar to most fisheries researchers and managers.

As in all ERA approaches, assumptions must be made to overcome deficiencies in knowledge about the species and fisheries being analyzed. EASI-Fish makes several assumptions related to the derivation of susceptibility input parameters (*e.g*., encounterability) additional to those of per-recruit models, and these should be carefully considered when interpreting the results. For example, both length-structured per-recruit models and simple surplus production models used in single-species stock assessments (Chen and Gordon 1997, Ye 1998) assume that the density of the species within the stock boundaries is homogenously distributed in space and time. This is an overly simplistic assumption that ignores the typical heterogeneity in fish densities, especially in tuna fisheries, where the increasing use of fish-aggregating devices (FADs) (Hall and Roman 2013) aggregate the biomass of species that would normally be more homogenously distributed (Hallier and Gaertner 2008, Dagorn *et al.* 2013). However, in an equilibrium state (which assumes no immigration or emigration, that the biomass of a species is pooled over the year, and that the species is exposed to instantaneous fishing mortality), such variability in local abundance may be tempered at the stock level.

The definition of stock boundaries is a particularly important consideration in the assessment of any species impacted by fishing (Cadrin and Secor 2009). Unfortunately, stock boundaries can be difficult to determine, even for a commercially-important species such as Atlantic bluefin tuna (*Thunnus thynnus*) that has been the subject of extensive tagging studies (Block *et al.* 2005) and genetic analyses (Carlsson *et al.* 2006). Therefore, it can be assumed that the stock boundaries of most bycatch species will be poorly understood, and so a precautionary approach is to define the smallest feasible stock boundary relative to the fishery being assessed. Pelagic species such as a tuna may have an assumed stock boundary, at least at the spatial scale of the smallest management unit. For example, for a country assessing a species within its EEZ, the stock boundary would also be the EEZ. However, for large management areas such as the EPO, it may be precautionary to assume smaller stocks based on oceanographic features or the movements of well-studied species. For example, a tagging study of bigeye tuna in the EPO indicated that there is little mixing across the Equator, suggesting separate northern and southern stocks (Schaefer *et al.* 2015).

One of the most important assumptions of the susceptibility component of EASI-Fish is that the presence of any level of fishing by any gear in a 0.5° x 0.5° grid cell where a species is predicted to be present is evenly distributed and has the potential to remove all fish within that cell, providing all other susceptibility attributes (*e.g*. encounterability, selectivity, *etc*.) are fully realized. This assumption is required to estimate fishing mortality that is independent of catch estimates, since catch data are often unavailable, or unreliable, for many bycatch species. Obviously, this is an overly conservative assumption. However, since the goal of the analysis is to prioritize species for data collection, further analysis, and/or mitigation, it can be considered reasonable, and it does take the lack of information into consideration. However, even if all susceptibility parameters were fully realized, EASI-Fish would likely overestimate fishing mortality, and thus represent a precautionary 'worst case' estimate of fishing impacts. It should be emphasized once again that, because of this assumption, the analysis should not be used for determining the status of a stock. Further research is needed to determine the impact of this assumption, and in what types of fish and effort distributions the assumption is optimistic or pessimistic. In cases where spatial information is available on fish density (*e.g*. environmental covariates are correlated with fish density, and effort distributions are known) development of more sophisticated methods may improve the approach.

A similar precautionary approach was also applied for other susceptibility parameters. For example, the encounterability parameter (E) assumed that the efficiency of a specific fishing gear was constant over its specified depth range, which is often not the case for longlines due to environmental factors such as currents and wind, and differences in gear configuration, such as the number of branch lines between floats, that affect the shoaling of the gear and its ultimate fishing depth (Bigelow *et al.* 2006). Therefore, the encounterability of the gear for species with depth ranges greater than 300 m (*e.g*. escolar) may be slightly overestimated.

In the current version of the EASI-Fish model it was assumed that fish were available for capture throughout their specified depth range at all times of day and night, in spite of the well-documented nocturnal vertical migrations to the mixed layer by many pelagic fish included in this study (Schaefer and Fuller 2007, Kerstetter *et al.* 2008, Polovina *et al.* 2008, Abascal *et al.* 2010, Schaefer and Fuller 2010, Hoolihan *et al.* 2011). Again, this was a precautionary approach to account for the lack of differentiation between deep and shallow sets in reported data (see Griffiths *et al.* 2017b) that would allow the definition of two longline 'fleets' in the model, as was done for the three purse-seine set types, which would in turn allow a better approximation of the potential for each longline 'fleet' to interact with fish while they occupied different depth ranges by day and by night. However, such precautionary assumptions are justified until the necessary operational-level longline data become available.

Of the susceptibility components, encounterability and contact selectivity—together generally referred to as "selectivity" in stock assessment—are two of the most important parameters influencing the outcomes of stock assessments and the subsequent management advice (Maunder *et al.* 2014), and therefore need careful consideration for data-poor bycatch species., Selectivity-at-length ogives were available for some economically-important species from stock assessments that included fleets or gear types that were comparable to the purse-seine and longline fleets specified in the EASI-Fish models. However, selectivity ogives are unlikely to be available for bycatch species. The most precautionary approach for these species is to assume that selectivity is fully realized for all length classes, as was done for *Alopias superciliosus* in the three purse-seine fisheries, because it was assessed as overexploited in other fisheries outside the EPO (Liu *et al.* 2006, Fu *et al.* 2016). However, if limited length-frequency or reliable anecdotal information is available (from scientific observers or fishers, for instance) to determine the smallest length caught by a particular gear (L_c), knife-edge selectivity can be assumed where selectivity is fully realized for all length classes greater than L_c, thereby removing any unrealistic fishing mortality on these length classes.

Post-release mortality is an important parameter when assessing a fishery. Handling and release practices that allow a significant proportion of captured fish to survive the sub-lethal effects of capture and release are much simpler to implement than measures such as gear modifications or spatial and temporal closures to reduce the capture of a particular non-target species. This can be accounted for in EASI-Fish to reduce the overall fishing mortality of a susceptible species. In the present study, post-release survival was assumed to be zero for all species assessed, since many species were either target or marketable non-target species. However, the available data on post-release survival-at-length for species that are commonly deliberately discarded, due to their low economic value or because it is mandated by a specific management measure (*e.g.* oceanic whitetip shark in the EPO), were insufficient to justify their inclusion in the analysis. However, tagging experiments of a few discarded bycatch species, albeit with small sample sizes, have been (Musyl *et al.* 2011), or are being, undertaken in the EPO.

4.1. Assessing cumulative impacts of fisheries

A key feature of EASI-Fish is that it allows the cumulative impacts of multiple fisheries on each species to be assessed. Adding fisheries impacts implies that the fisheries do not catch the same fish. However, the assumption that all fish in a grid can be caught—where fisheries happen to overlap exactly and have fully-realized selectivity—can violate the additive assumption. For example, if one fishery fishes in half of an area, its effect, ignoring all the other components, is 50%, but if another fishery fishes in the same half of the area, then its effect, ignoring all the other components, is also 50%. The total of both fisheries effects is thus 100% even though only half of the total area is fished. However, it might be the case that the fish caught by the different fisheries do not overlap because of one of the other components of the calculation. As a hypothetical example, if the first fishery fishes only in the top half of the water column and the second fishery fishes only in the bottom half, then each fishery will be fishing on a different component of the population and catch 25% of the fish (0.5 x 0.5), for a combined impact of 50%, which is then the total area covered by the two fisheries. Of course, an actual application will be much more complex, and fisheries are not able to catch all fish as assumed. The main point is that, when fisheries are combined, they should be evaluated to ensure that they do not substantially overlap in the fish that they catch, given the information available. If they do, then some adjustments should be made. For example, the spatial overlap could be calculated by combining the data for the fisheries that overlap. In some cases, it will not be possible to do this evaluation, and the conservative assumption of no overlap might have to be made.

4.2. Biological reference points

Selecting appropriate RPs to assess fish stocks is an ongoing topic of debate among stock assessment modelers, fisheries managers and policy-makers. Whilst many modern stock assessments use MSY as a target reference point (Smith and Punt 2001), it has been suggested that MSY should be regarded as an upper limit reference point, because the difficulty of estimating it with precision (Caddy and McGarvey 1996, Die and Caddy 1997) means that overfishing can occur before fishing mortality reaches the MSY level. There is the added potential for overestimating the vulnerability of a stock when using yield-per-recruit models, since the stock-recruitment relationship is assumed to be time-invariant and recruitment is independent of stock size, equivalent to a steepness (h) value of 1 (Gabriel and Mace 1999). Conversely, an assessment using Y/R for species that do have a stock-recruitment relationship (*i.e.* $h<1$) would be overly optimistic. For these reasons, Walters *et al.* (2005) argued that MSY has the potential to deteriorate the structure of ecosystems supporting fisheries, and advocated for the use of alternative, more conservative RPs for target species. For instance, Dichmont *et al.* (2010) demonstrated that using maximum economic yield (F_{MEY}) can optimize fishery profits whilst simultaneously satisfying biological conservation objectives, since F_{MEY} is generally attained at fishing mortalities less than F_{MSY}. Other authors recommended the use of $F_{0.1}$ as a precautionary RP for data-limited species or fisheries (Gabriel and Mace 1999), but subsequent work has suggested that low-productivity species, such as elasmobranchs and long-lived teleost fish, are likely to be over-exploited before $F_{0.1}$ is reached (Punt 2000, Campana *et al.* 2002). The problem is exacerbated for these less-productive species and for data-poor species whose biological parameter estimates have large uncertainties, because F-based reference points (F_{MSY} and $F_{0.1}$) can be too sensitive to these uncertainties to provide reliable estimates of vulnerability (Tsai *et al.* 2011), and are therefore liable to incur false negatives—giving a species that should be classified as “most vulnerable” a different classification. Using mako shark as an example, Tsai *et al.* (2011) advocated for the use of RPs that relate to the spawning stock in such data-limited settings, and recommended a target reference point based on a spawning potential ratio of 35% ($\text{SPR}_{35\%}$).

The stock-recruitment relationship is a highly influential component of the productivity of a stock, and can differ substantially among species: compare a highly-fecund pelagic spawner (Maunder and Deriso 2013) with a low-fecundity live-bearing shark (Taylor *et al.* 2013) or a marine mammal (Punt 2017). Therefore, it is important to take the stock-recruitment relationship into consideration when deriving reference points. The Y/R analysis could be extended to include the stock-recruitment relationship to estimate true MSY-based reference points, rather than relying on F_{MAX} or other proxies. The stock-recruitment relationship is difficult to estimate (*e.g.* Magnusson and Hilborn 2007, Conn *et al.* 2010, Lee *et al.* 2012), but species or taxonomic group values can be obtained from meta-analyses (*e.g*. Myers *et al.* 1999). Alternatively, taxonomic group-based proxies could be used: for example, the Pacific Management Council used the $F_{40\%}$ as a proxy for *Sebastes*, and $F_{35\%}$ for all other stocks (Ralston 2002).

Given the likely uncertainty in the input data used for deriving both susceptibility and productivity estimates for data-poor bycatch species, F_{MSY} may not be an appropriate RP, not only because of parameter uncertainties, but also because a fishery manager’s objective is not to optimize yield of bycatch species. Instead, managers may simply wish to have a buffer between fishing mortality and an RP sufficient to ensure that there is room for management intervention should bycatch populations begin to show signs of becoming unsustainable. Therefore, RPs developed for data-poor species used in association with the SAFE ERA approach may not be appropriate either. This is because F_{msm} approximates F_{MSY}, while F_{crash} is the lowest fishing mortality that renders a species extinct, and certainly is not conservative enough, as the population may have reached an unsustainable level for a significant period prior to reaching this RP. Our results showed that $\text{SSB}_{40\%}$ (equivalent to $\text{SPR}_{40\%}$) is the most precautionary RP of the three implemented, and is slightly more precautionary than the $\text{SPR}_{35\%}$ RP recommended by Tsai *et al.* (2011). However, further work is required to determine the most appropriate RPs for different taxonomic groups. For example, less-productive species such as sharks and turtles may be best assessed using biomass-based RPs, while fishing mortality-based RPs may be more appropriate for more productive species (e.g. tunas).

5. CONCLUSIONS AND DIRECTIONS FOR FUTURE WORK

In assessing the potential vulnerability of data-poor bycatch species there is an inherent trade-off between the accuracy and the precision of assessment outcomes. However, EASI-Fish—like the majority of ERA methods, such as PSA—is not intended to provide highly-precise stock status estimates for data-poor species, but rather a transparent and repeatable methodology that is (i) capable of quantitatively estimating species-specific vulnerability for the purposes of prioritizing species for data collection, further detailed analysis, research and management, and (ii) transferable between species with different life histories (*e.g*. teleosts to marine mammals). The latter is a major advantage of the EASI-Fish approach, in that species groups with different life histories that are impacted by fisheries can be assessed using the same model. This is not the case with PSA, where productivity and susceptibility attributes ideally need to be adapted to particular species groups (see Milton 2001, Stobutzki *et al.* 2001, Stobutzki *et al.* 2002) given the differences in the magnitude of their productivity parameters. For example, the reproductive biology of elasmobranchs and marine mammals differs significantly from that of teleosts, particularly in that they generally produce numbers of offspring that are many orders of magnitude smaller and exhibit some degree of parental care (Walker 1998). Therefore, when assessed together with teleosts, elasmobranchs and marine mammals are often clearly the most vulnerable species (see Kirby 2006, Arrizabalaga *et al.* 2011), but only because of the magnitude of the scales required for biological attributes to accommodate all species. This increases the potential for creating false negatives for more productive species that may in fact be at risk, which would be apparent if productivity scales were used that are relevant to these species. However, separating species groups for analysis using PSA also creates other issues for determining which species are at risk, owing to the arbitrary overall vulnerability scores based on categorical attribute scores. For instance, a vulnerability score of 2.3 for teleosts, for example, does not necessarily have the same biological meaning as a score of 2.3 for marine mammals, and this creates difficulties in determining which species are truly vulnerable to fishing.

The EASI-Fish approach is designed for data-poor species, but could be adapted for species with additional data. For many fisheries and species in the EPO that need prioritization, more data are available than are needed for EASI-Fish. Therefore, the EASI-Fish approach, which explicitly deals with spatial overlap and the cumulative effect of multiple fisheries, which are highly desirable qualities, should be further developed to improve estimates that use the available data. For example, the spatial overlap could be improved by determining relationships between fish density and environmental covariates, and matching that with purse-seine and longline effort data to get a better idea of the impact of fishing. The concepts in EASI-Fish could also be used in a less quantitative way to eliminate species from management focus. For example, the simple observation that no significant fisheries operate in areas where a substantial proportion of the fish reside would indicate that that species is not at risk and does not need management regulation.

Although the approach for implementing EASI-Fish described in this paper, using EPO tuna fisheries as a case study, marks an important 'proof of concept', there is scope for further development of the model and its application to other fisheries, such as demersal fisheries that may have unique issues to consider in the estimation of susceptibility parameters. In view of the growing worldwide demand for fisheries to demonstrate that they are ecologically responsible, through both formal avenues (*e.g*. legislation and policies) and the influence of the public perception of commercial fishing, EASI-Fish was designed with the end-user in mind to assist fisheries stakeholders to meet these demands cost-effectively. The flexibility of the EASI-Fish user controls in the Microsoft Excel environment, and the pre-filled best available biological parameters for species caught in EPO tuna fisheries, allow researchers and managers with limited modelling or statistical expertise to quickly and easily explore the potential outcomes of the implementation of specific management measures such as spatial and/or temporal closures, improved post-release survival of discarded species due to improved handling practices (Poisson *et al.* 2014), minimum retention lengths (Griffiths *et al.* 2006), changes in gear selectivity, or the potential effects of climate change, such as a species' distribution moving relative to fishing grounds or political boundaries (*e.g*. Marine Protected Areas, or a nation's EEZ) (Perry *et al.* 2005). Variations in such factors can be implemented independently or in unison, and the change in a species' status can be easily assessed using a range of pre-loaded conventional RPs, or specific RPs that can be added by the user.

The EASI-Fish model presents a significant evolutionary step in the development of rapid and cost-effective ecological risk assessment approaches that are capable of quantitatively assessing vulnerability of data-poor bycatch species to the cumulative impacts of multiple fisheries by using conventional RPs that have been validated in species-rich settings and are widely understood by fisheries researchers and managers. As a result, EASI-Fish may afford fisheries managers more confidence in identifying the most vulnerable species, in order to direct resources to either implementing mitigation measures, or prioritizing research or data collection that can fill key data gaps and subsequently allow more formal and precise stock assessments.

ACKNOWLEDGMENTS

The authors wish to thank Mark Maunder, Rick Deriso, Alex Aires-da-Silva and Cleridy Lennert-Cody for providing valuable advice on the method development and critically reviewing drafts of the manuscript. Nick Webb is thanked for editorial comments that greatly improve the manuscript.

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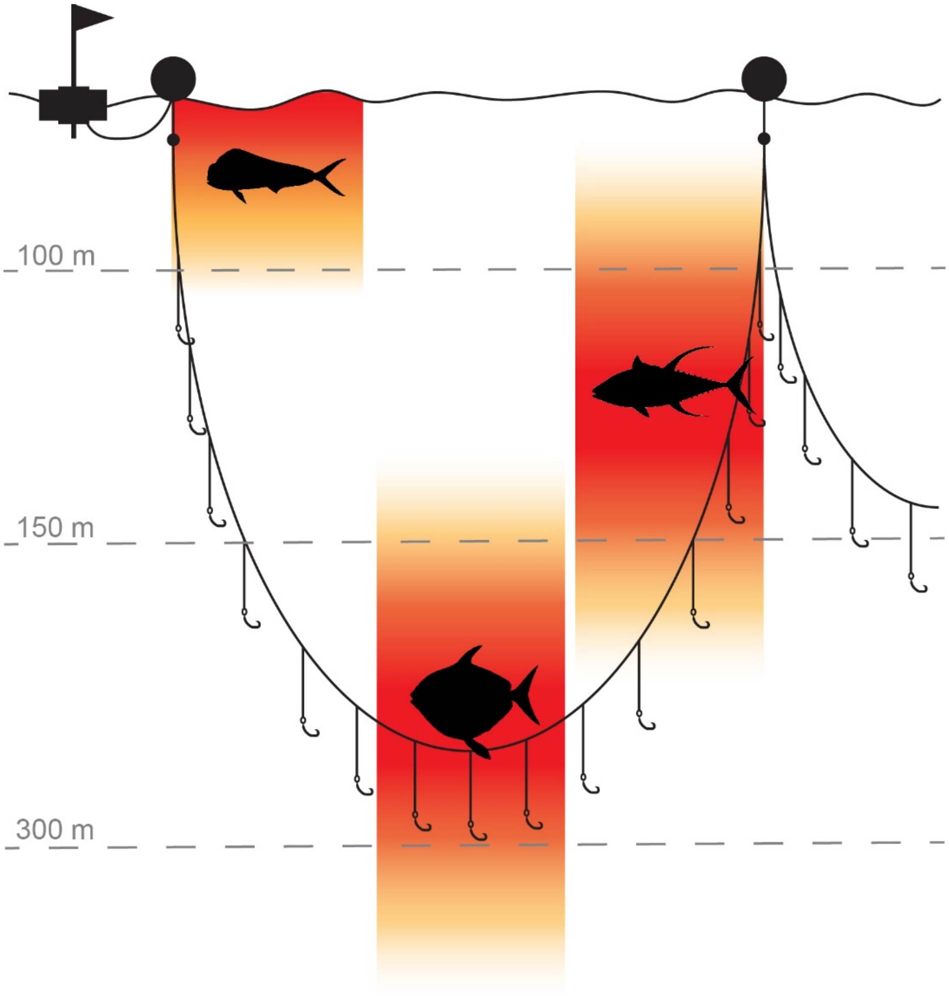
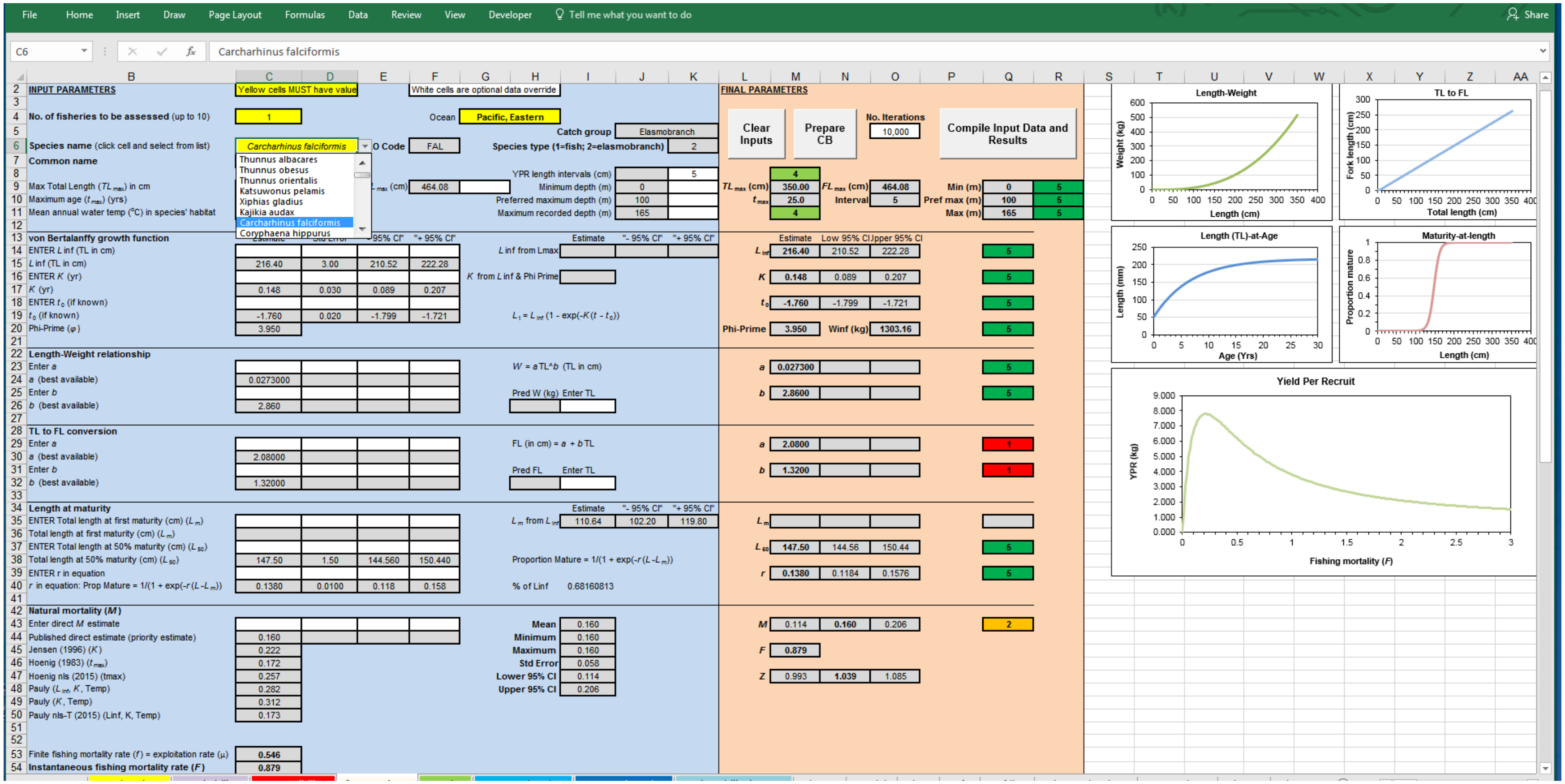


FIGURE 1. Schematic diagram illustrating the concept of encounterability of longline gear by three species (dorado, opah, and yellowfin tuna), given their typical depth preferences during the day relative to the longline gear that is assumed to fish depths of 0-300m during daytime “deep sets”.



Carcharhinus falciformis

INPUT PARAMETERS Yellow cells MUST have value White cells are optional data override

No. of fisheries to be assessed (up to 10): 1

Ocean: Pacific, Eastern

Catch group: Elasmobranch

Species name (click cell and select from list): *Carcharhinus falciformis*

Thunnus albacares
Thunnus obesus
Thunnus orientalis
Katsuwonus pelamis
Xiphias gladius
Kajikia audax
Carcharhinus falciformis
Coryphaena hippurus

O Code: FAL

Species type (1=fish; 2=elasmobranch): 2

Common name

Max Total Length (TL_{max}) in cm

Maximum age (t_{max}) (yrs)

Mean annual water temp (°C) in species' habitat

L_{max} (cm): 464.08

YPR length intervals (cm): 5

Minimum depth (m): 0

Preferred maximum depth (m): 100

Maximum recorded depth (m): 165

	Estimate	Std Error	"- 95% CI"	"+ 95% CI"
von Bertalanffy growth function				
ENTER *L* inf (TL in cm)				
L inf (TL in cm)	216.40	3.00	210.52	222.28
ENTER *K* (yr)				
K (yr)	0.148	0.030	0.089	0.207
ENTER t_0 (if known)				
t_0 (if known)	-1.760	0.020	-1.799	-1.721
Phi-Prime (φ)	3.950			
Length-Weight relationship				
Enter *a*				
a (best available)	0.0273000			
Enter *b*				
b (best available)	2.860			
TL to FL conversion				
Enter *a*				
a (best available)	2.08000			
Enter *b*				
b (best available)	1.32000			
Length at maturity				
ENTER Total length at first maturity (cm) (L_m)				
Total length at first maturity (cm) (L_m)				
ENTER Total length at 50% maturity (cm) (L_{50})				
Total length at 50% maturity (cm) (L_{50})	147.50	1.50	144.560	150.440
ENTER r in equation				
r in equation: Prop Mature = 1/(1 + exp(-*r* (*L* - L_m))	0.1380	0.0100	0.118	0.158

	Estimate	"- 95% CI"	"+ 95% CI"
L inf from Lmax			
K from *L* inf & Phi Prime			
L_m from L_{inf}	110.64	102.20	119.80

$L_t = L_{inf} (1 - \exp(-K(t - t_0)))$

$W = a TL^b$ (TL in cm)

Pred W (kg) Enter TL

FL (in cm) = $a + b$TL

Pred FL Enter TL

Proportion Mature = 1/(1 + exp(-*r* (*L* - L_m))

% of Linf 0.68160813

Natural mortality (*M*)

Enter direct *M* estimate	
Published direct estimate (priority estimate)	0.160
Jensen (1996) (*K*)	0.222
Hoenig (1983) (t_{max})	0.172
Hoenig nls (2015) (tmax)	0.257
Pauly (L_{inf}, *K*, Temp)	0.282
Pauly (*K*, Temp)	0.312
Pauly nls-T (2015) (Linf, K, Temp)	0.173

Mean	0.160
Minimum	0.160
Maximum	0.160
Std Error	0.058
Lower 95% CI	0.114
Upper 95% CI	0.206

Finite fishing mortality rate (*f*) = exploitation rate (μ): **0.546**

Instantaneous fishing mortality rate (*F*): **0.879**

FINAL PARAMETERS

No. Iterations: 10,000

	4		
TL_{max} (cm)	350.00	FL_{max} (cm)	464.08
t_{max}	25.0	Interval	5
	4		

Min (m)	0	5
Pref max (m)	100	5
Max (m)	165	5

	Estimate	Low 95% CI	Upper 95% CI	
L_{inf}	216.40	210.52	222.28	5
K	0.148	0.089	0.207	5
t_0	-1.760	-1.799	-1.721	5
Phi-Prime	3.950	Winf (kg)	1303.16	5
a	0.027300			5
b	2.8600			5
a	2.0800			1
b	1.3200			1
L_m				
L_{50}	147.50	144.56	150.44	5
r	0.1380	0.1184	0.1576	5
M	0.114	0.160	0.206	2
F	0.879			
Z	0.993	1.039	1.085	

FIGURE 2. Screenshot of the parameter input graphical interface for the Ecological Assessment of the Sustainable Impacts of Fisheries (EASI-Fish) model. The model is pre-filled with the best available parameter values (grey cells) for species caught in EPO tuna fisheries (including links to data sources), which can be overridden by the user if required.

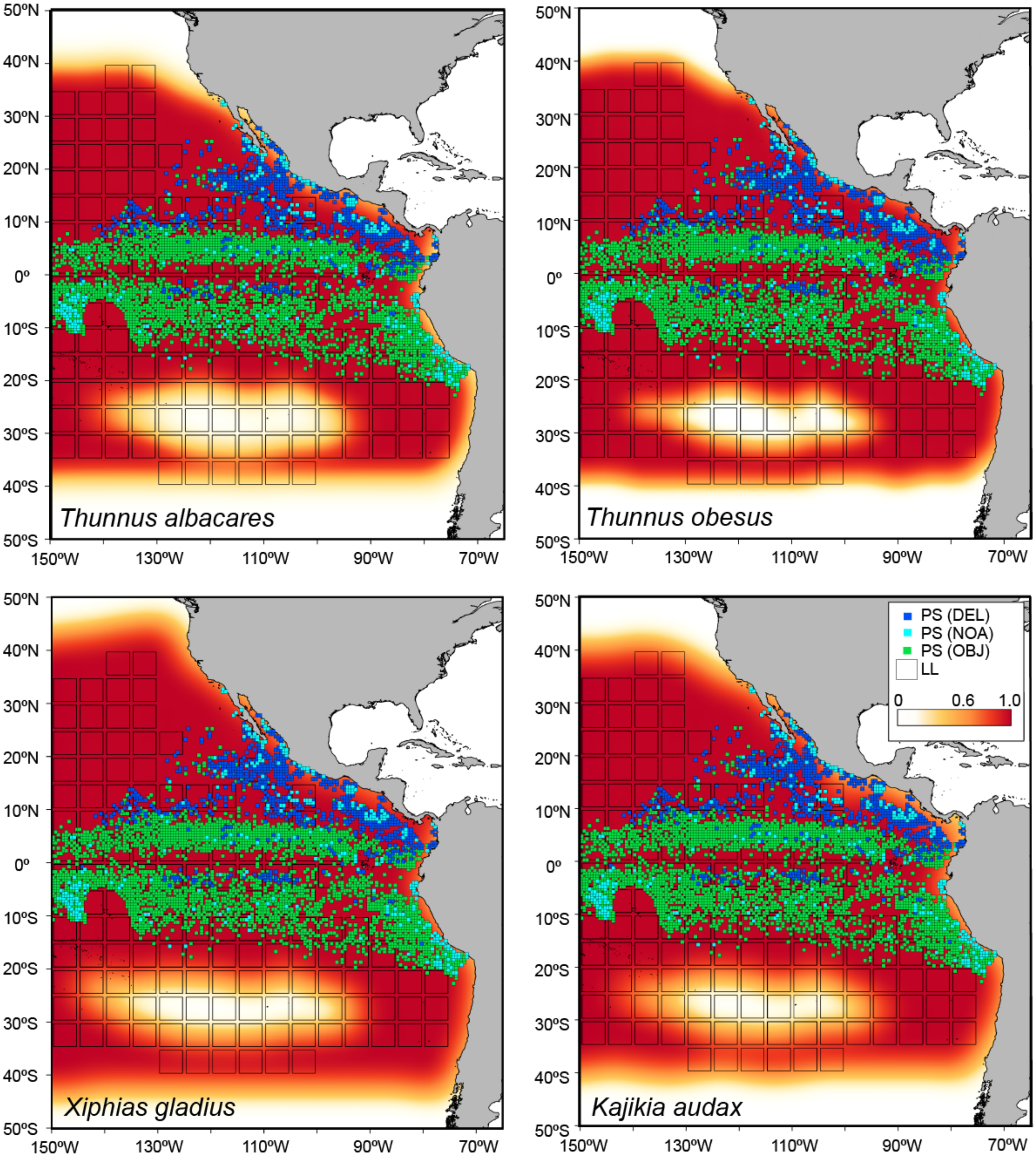


FIGURE 3. Maps showing the modelled distributions of four principal target species caught in the eastern Pacific Ocean overlaid with the distribution of the purse seine (PS) fishery (Class 6 vessels only) (0.5° x 0.5°) and the large-scale tuna longline (LL) fishery (5° x 5°) in 2016. Gradient bar in legend shows probability of occupancy of each species in 0.5° x 0.5° cells.

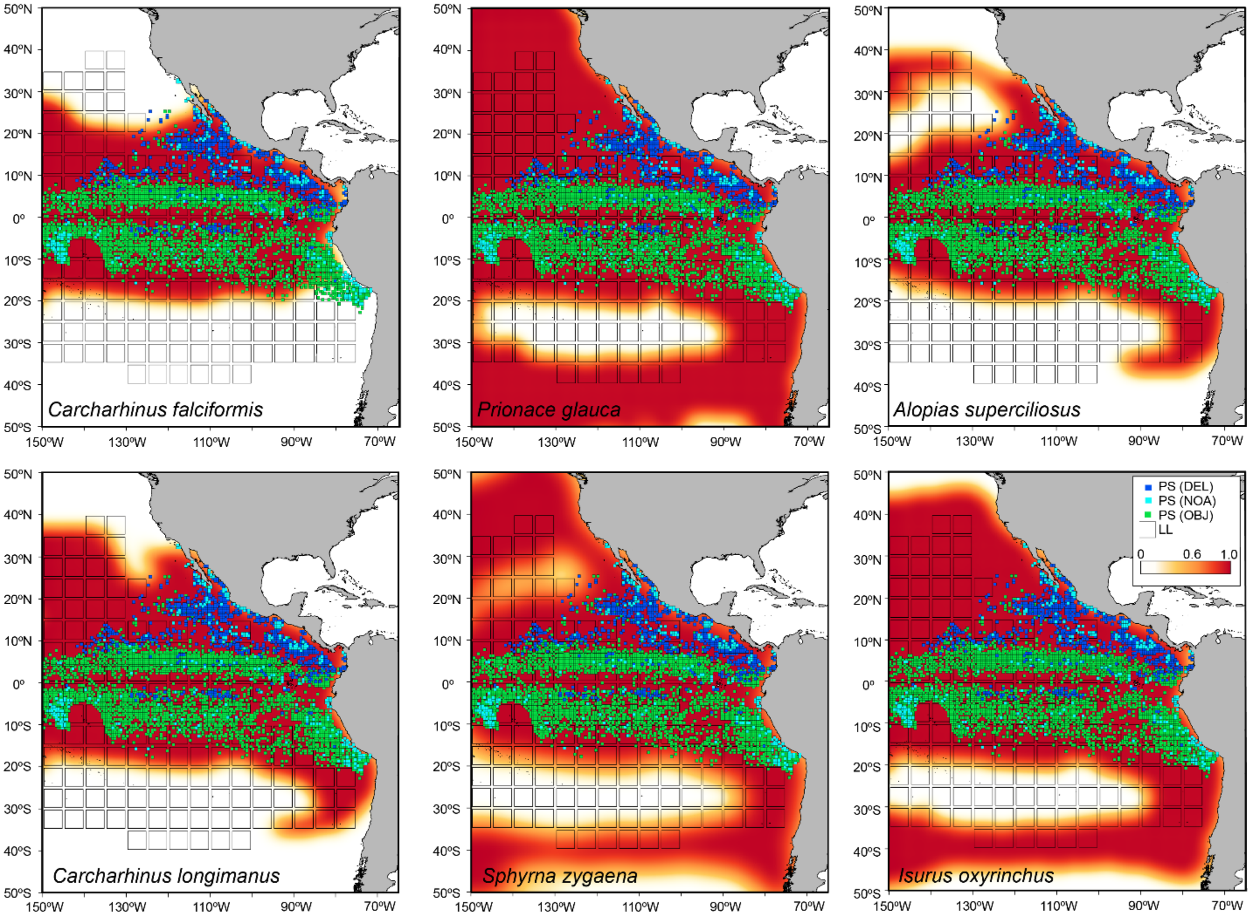


FIGURE 4. Maps showing the modelled distributions of six shark species caught in the eastern Pacific Ocean overlaid with the distribution of the purse seine (PS) fishery (Class 6 vessels only) (0.5° x 0.5°) and the large-scale tuna longline (LL) fishery (5° x 5°) in 2016. Gradient bar in legend shows probability of occupancy of each species in 0.5° x 0.5° cells.

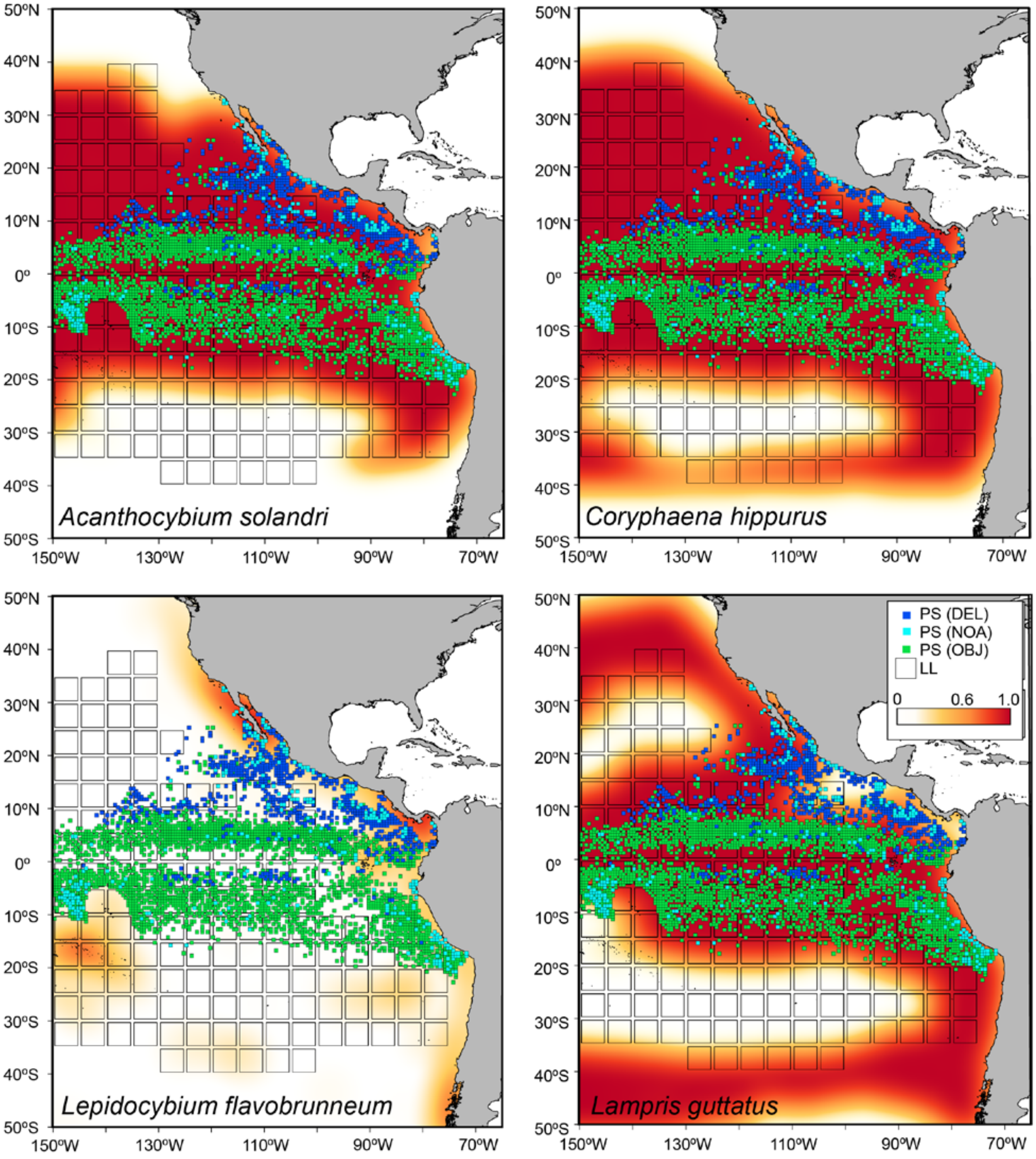


FIGURE 5. Maps showing the modelled distributions of four non-target teleost species caught in the eastern Pacific Ocean overlaid with the distribution of the purse seine (PS) fishery (Class 6 vessels only) (0.5° x 0.5°) and the large-scale tuna longline (LL) fishery (5° x 5°) in 2016. Gradient bar in legend shows probability of occupancy of each species in 0.5° x 0.5° cells.

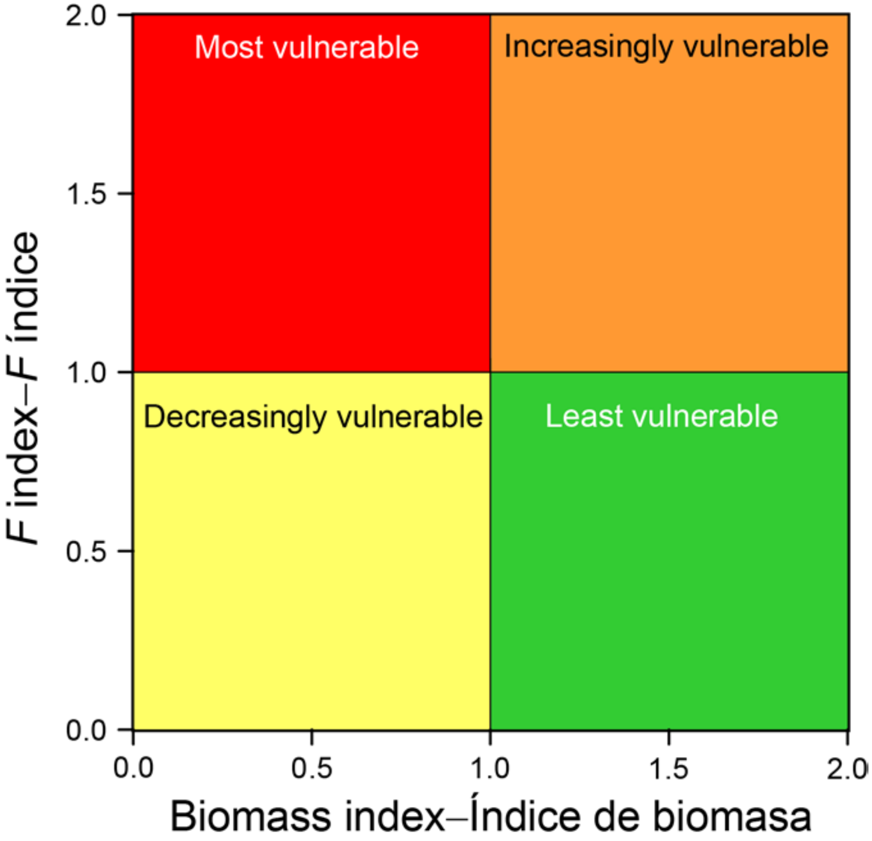


FIGURE 6. Phase plot illustrating how vulnerability status was defined for each species assessed using a desired fishing mortality and biomass index from the EASI-Fish model as a reference point on the x and y axis, respectively. If using an MSY-based index, vulnerability of a species would be defined by its position within one of four quadrants in the phase plot: "Least vulnerable" (green, $F_{2016}/F_{MSY} < 1$ and $SSB_{2016}/SSB_{MSY} > 1$), "Increasingly vulnerable" (orange, $F_{2016}/F_{MSY} > 1$ and $SSB_{2016}/SSB_{MSY} > 1$), "Most vulnerable" (red, $F_{2016}/F_{MSY} > 1$ and $SSB_{2016}/SSB_{MSY} < 1$), and "Decreasingly vulnerable" (yellow, $F_{2016}/F_{MSY} < 1$ and $SSB_{2016}/SSB_{MSY} < 1$). Maximum axis limits of 2.0 are for illustrative purposes only.

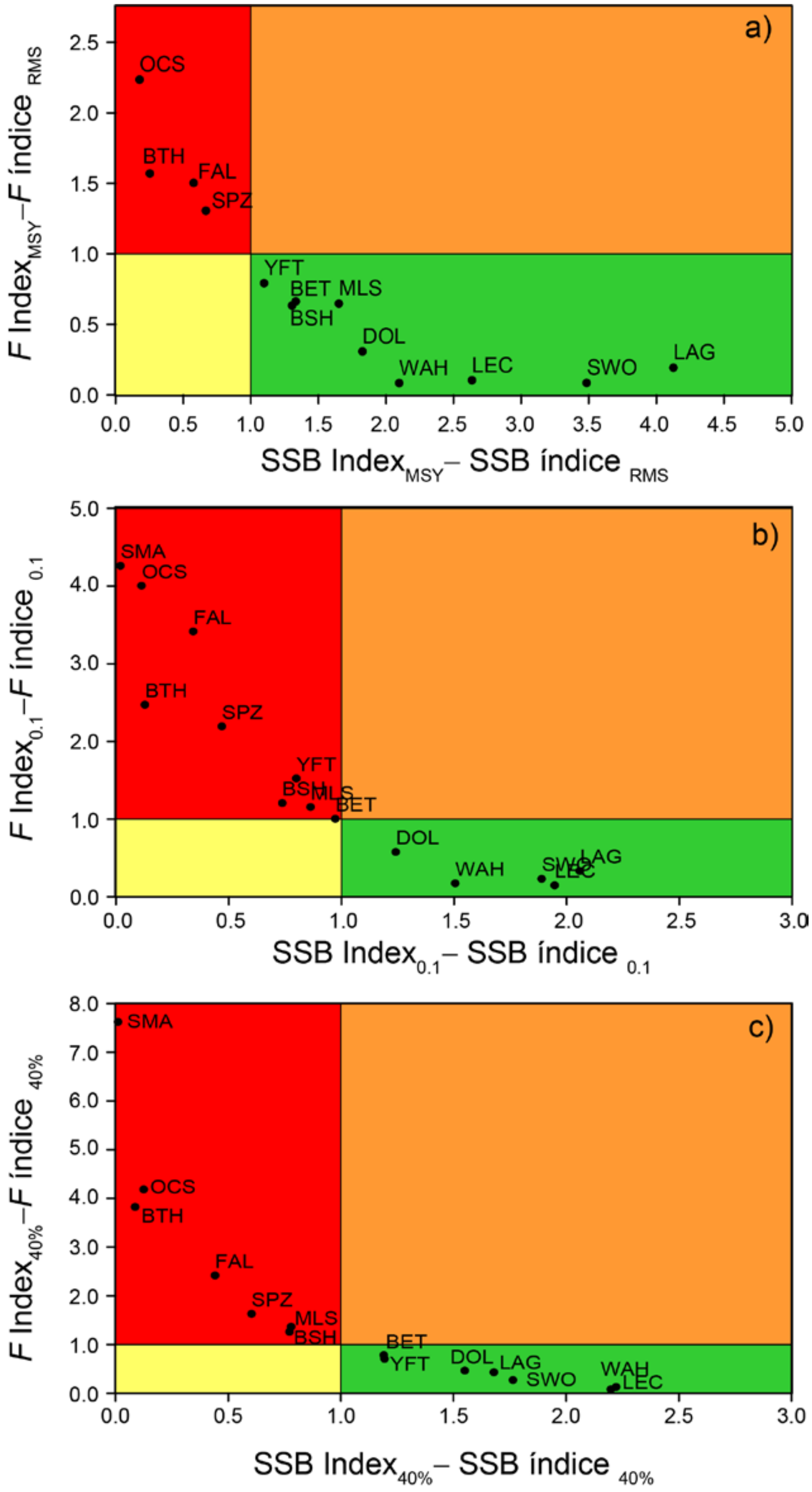
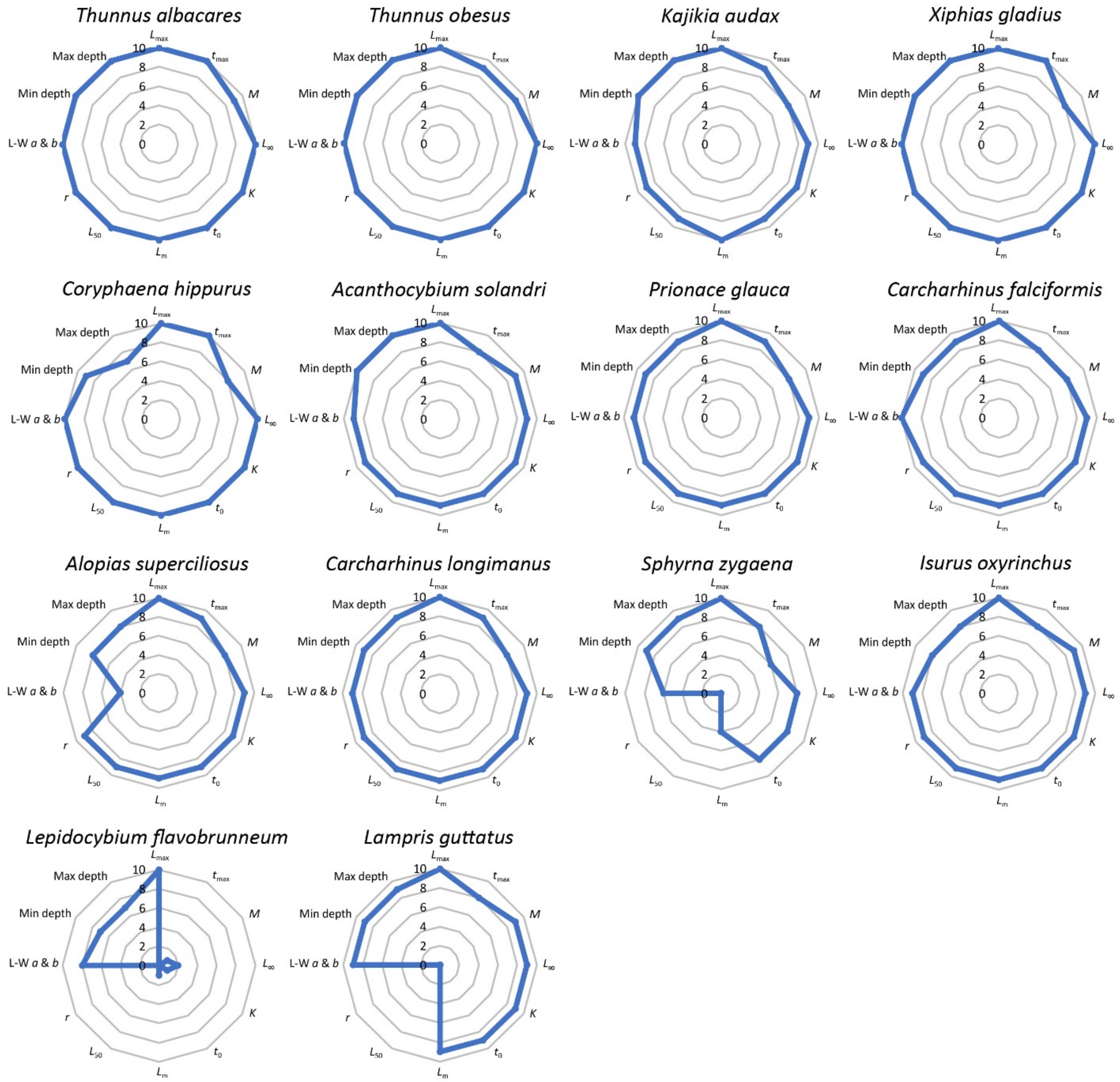


FIGURE 7. Kobe plots showing the status of 14 species caught in EPO tuna fisheries assessed by EASI-Fish represented by point estimates for three pairs of biological reference points: a) F_{2016}/F_{MSY} and SSB_{2016}/SSB_{MSY}, b) $F_{2016}/F_{0.1}$ and $SSB_{2016}/SSB_{0.1}$ and c) $F_{2016}/F_{40\%}$ and $SSB_{2016}/SSB_{40\%}$. Note the differences in axis scales between plots.


Thunnus albacares

Thunnus obesus

Kajikia audax

Xiphias gladius

Coryphaena hippurus

Acanthocybium solandri

Prionace glauca

Carcharhinus falciformis

Alopias superciliosus

Carcharhinus longimanus

Sphyrna zygaena

Isurus oxyrinchus

Lepidocybium flavobrunneum

Lampris guttatus

FIGURE 8. Radar plots showing the relative quality of biological and ecological parameters ($L_{\max}$, $t_{\max}$, M, L_∞, K, t_0, L_m, L_{50}, logistic curvature coefficient r, length-weight parameters a and b, minimum and maximum depth) used in EASI-Fish models of 14 species caught in EPO tuna fisheries. Scale ranges from 0 (data absent for the species and its closely related species) to 10 (high quality species-specific data derived from the EPO).

TABLE 1. Natural mortality (*M*) estimators used in the present study.

Estimator	Equation	Citation
Hoenig$_{tmax}$	$M = \frac{4.3}{t_{max}}$	Hoenig (1983)
Hoenig$_{\mathrm{nls}}$	$M = 4.899 t_{max}^{-0.916}$	Then *et al.* (2015)
Jensen	$M = 1.60\ K$	Jensen (1996)
Pauly$_{\mathrm{nls}}$	$M = 4.118 K^{0.73} L_{\infty}^{-0.33}$	Then *et al.* (2015)
Pauly$_{\mathrm{LKT}}$	$\log M = -0.0066 - 0.279 \ln L_{\infty} + 0.6543 \ln K + 0.4634 \ln T$	Pauly (1980)
Pauly$_{\mathrm{KT}}$	$M = K e^{-0.22 + 0.3 \ln T}$	Froese and Pauly (2017)
Pauly$_{\mathrm{LT}}$	$M = 10^{0.566 - 0.718 \ln L_{\infty}} + 0.02T$	Froese and Pauly (2017)

M = instantaneous natural mortality rate (yr^{-1})
$T_{\max}$ = maximum observed age of animals in the stock.
L_{∞} = the average length of a fish if it lived to an infinite age, and known as the asymptotic length of fish in the von Bertalanffy growth function (yr^{-1}).
K = the curvature parameter of the von Bertalanffy growth function (yr^{-1}).
T = mean water temperature (°C) at the location and depth range inhabited by the fish.

TABLE 2. Qualitative index used to rank the relative reliability of biological and ecological parameters used for each species in EASI-Fish assessments with respect to the reliability of the methodology to estimate the parameter and the precision of parameter estimate, relative to the data source's relevance to the species and region being assessed. EPO: Eastern Pacific Ocean WCPO: Western and Central Pacific Ocean

		High reliability		Medium reliability		Low reliability		No data
		High precision	Low precision	High precision	Low precision	High precision	Low precision	
Species-specific	EPO	10	9	8	7	6	5	0
	WCPO	9	8	7	6	5	4	0
	Other	8	7	6	5	4	3	0
Related species	EPO	7	6	5	4	3	2	0
	WCPO	6	5	4	3	2	1	0
	Other	5	4	3	2	1	1	0

TABLE 3. Parameter values used for variables describing the susceptibility of capture for each species in the four fisheries defined for the eastern Pacific Ocean tuna fishery. A description of susceptibility-at-length is given where parameter values differed by length for a particular variable.

Species	Fishery	Proportion of species-occupied grids (G_x/G) fished	Fishing season duration (D_x)	Seasonal availability (A_{xj})	Encounterability (E_{xj})	Contact selectivity (C_{xj})	Post-release mortality (P_{xj})
Thunnus albacares	**Longline**	**0.76**	**1.0** Year-round	**1.0** Year-round	**1.00** Deep sets assumed to fish 0-300m. Species inhabits 0-250m (Schaefer *et al.* 2007).	**0.41** Used logistic-shaped selectivity ogive for dominant EPO fleet (Minte-Vera *et al.* 2017).	**1.0** Assumed no release of target species.
	Purse-seine (DEL)	**0.19**	**0.83** 62-d closure	**1.0** Year-round	**0.80** DEL sets assumed to fish 0-200m. Species inhabits 0-250m (Schaefer *et al.* 2007).	**0.66** Used logistic-shaped selectivity ogive for dominant EPO DEL sets (Minte-Vera *et al.* 2017).	**1.0** Assumed no release of target species.
	Purse-seine (NOA)	**0.02**	**0.83** 62-d closure	**1.0** Year-round	**0.80** NOA sets assumed to fish 0-200m. Species inhabits 0-250m (Schaefer *et al.* 2007).	**0.65** Used dome-shaped selectivity ogive for dominant EPO NOA sets (Minte-Vera *et al.* 2017).	**1.0** Assumed no release of target species.
	Purse-seine (OBJ)	**0.05**	**0.83** 62-d closure	**1.0** Year-round	**0.80** OBJ sets assumed to fish 0-200m. Species inhabits 0-250m (Schaefer *et al.* 2007).	**0.61** Used dome-shaped selectivity ogive for dominant EPO OBJ sets (Minte-Vera *et al.* 2017).	**1.0** Assumed no release of target species.
Thunnus obesus	**Longline**	**0.76**	**1.0** Year-round	**1.0** Year-round	**0.73** Deep sets assumed to fish 0-300m. Species inhabits 30-400m (Schaefer and Fuller 2010).	**0.56** Used logistic-shaped selectivity ogive for dominant EPO fleet (Aires-da-Silva *et al.* 2016).	**1.0** Assumed no release of target species.
	Purse-seine (DEL)	**0.19**	**0.83** 62-d closure	**1.0** Year-round	**0.46** DEL sets assumed to fish 0-200m. Species inhabits 30-400m (Schaefer and Fuller 2010).	**0.64** Used logistic-shaped selectivity ogive for dominant EPO DEL sets (Aires-da-Silva *et al.* 2016).	**1.0** Assumed no release of target species.
	Purse-seine (NOA)	**0.02**	**0.83** 62-d closure	**1.0** Year-round	**0.46** NOA sets assumed to fish 0-200m. Species inhabits 30-400m (Schaefer and Fuller 2010).	**0.64** Used dome-shaped selectivity ogive for dominant EPO NOA sets (Aires-da-Silva *et al.* 2016).	**1.0** Assumed no release of target species.
	Purse-seine (OBJ)	**0.05**	**0.83** 62-d closure	**1.0** Year-round	**0.46** OBJ sets assumed to fish 0-200m. Species inhabits 30-400m (Schaefer and Fuller 2010).	**0.59** Used dome-shaped selectivity ogive for dominant EPO OBJ sets (Aires-da-Silva *et al.* 2016).	**1.0** Assumed no release of target species.
Kajikia audax	**Longline**	**0.79**	**1.0** Year-round	**1.0** Year-round	**1.00** Deep sets assumed to fish 0-300m. Species inhabits 0-100m (Brill *et al.* 1993).	**0.77** Used logistic-shaped selectivity ogive for dominant EPO fleet (Hinton 2009).	**1.0** Assumed no release of marketable species.
	Purse-seine (NOA)	**0.02**	**0.83** 62-d closure	**1.0** Year-round	**1.00** NOA sets assumed to fish 0-200m. Species inhabits 0-100m (Brill *et al.* 1993).	**0.61** Used logistic-shaped selectivity ogive for dominant EPO NOA sets (Hinton 2009).	**1.0** Assumed no release of marketable species.
	Purse-seine (OBJ)	**0.05**	**0.83** 62-d closure	**1.0** Year-round	**1.00** OBJ sets assumed to fish 0-200m. Species inhabits 0-100m (Brill *et al.* 1993).	**0.53** Used logistic-shaped selectivity ogive for dominant EPO OBJ sets (Hinton 2009).	**1.0** Assumed no release of marketable species.
Xiphias gladius	**Longline**	**0.73**	**1.0** Year-round	**1.0** Year-round	**0.75** Deep sets assumed to fish 0-300m. Species inhabits 0-400m (Brill *et al.* 1993).	**0.37** Used logistic-shaped selectivity ogive for dominant EPO fleet (Hinton and Maunder 2011).	**1.0** Assumed no release of marketable species.
Coryphaena hippurus	**Longline**	**0.76**	**1.0** Year-round	**1.0** Year-round	**1.00** Deep sets assumed to fish 0-300m. Species inhabits 0-60m (Furukawa *et al.* 2011).	**0.61** Used logistic-shaped selectivity ogive for dominant EPO fleet (Aires-da-Silva *et al.* 2017).	**1.0** Assumed no release of marketable species.
	Purse-seine (OBJ)	**0.06**	**0.83** 62-d closure	**1.0** Year-round	**1.00** OBJ sets assumed to fish 0-200m. Species inhabits 0-60m (Furukawa *et al.* 2011).	**0.19** Used logistic-shaped selectivity ogive for dominant EPO OBJ sets (Aires-da-Silva *et al.* 2017).	**1.0** Assumed no release of marketable species.

Acanthocybium solandri	**Longline**	**0.77**	**1.0** Year-round	**1.0** Year-round	**1.00** Deep sets assumed to fish 0-300m. Species inhabits 0-30m (Sepulveda *et al.* 2011).	**0.28** In absence of selectivity ogive for EPO longline fleet, mirrored logistic-shaped selectivity for yellowfin tuna.	**1.0** Assumed no release of marketable species.
	Purse-seine (NOA)	**0.03**	**0.83** 62-d closure	**1.0** Year-round	**1.00** NOA sets assumed to fish 0-200m. Species inhabits 0-30m (Sepulveda *et al.* 2011).	**0.70** In absence of selectivity ogive, used IATTC observer length-frequency data to assume knife-edge selectivity from 50cm FL.	**1.0** Assumed no release of marketable species.
	Purse-seine (OBJ)	**0.07**	**0.83** 62-d closure	**1.0** Year-round	**1.00** OBJ sets assumed to fish 0-200m. Species inhabits 0-30m (Sepulveda *et al.* 2011).	**0.72** In absence of selectivity ogive for EPO OBJ sets, mirrored logistic-shaped selectivity for yellowfin tuna.	**1.0** Assumed no release of marketable species.
Carcharhinus falciformis	**Longline**	**0.78**	**1.0** Year-round	**1.0** Year-round	**1.00** Deep sets assumed to fish 0-300m. Species inhabits 0-100m (Musyl *et al.* 2003).	**0.34** Used logistic-shaped selectivity ogive for dominant EPO fleet (Aires-da-Silva *et al.* 2014).	**1.0** Assumed 100% mortality in absence of release data.
	Purse-seine (DEL)	**0.30**	**0.83** 62-d closure	**1.0** Year-round	**1.00** DEL sets assumed to fish 0-200m. Species inhabits 0-100m (Musyl *et al.* 2003).	**0.36** Used logistic-shaped selectivity ogive for dominant EPO DEL sets (Aires-da-Silva *et al.* 2014).	**1.0** Mandatory release, but assumed 100% mortality in absence of data.
	Purse-seine (NOA)	**0.03**	**0.83** 62-d closure	**1.0** Year-round	**1.00** NOA sets assumed to fish 0-200m. Species inhabits 0-100m (Musyl *et al.* 2003).	**0.36** Used logistic-shaped selectivity ogive for dominant EPO NOA sets (Aires-da-Silva *et al.* 2014).	**1.0** Mandatory release, but assumed 100% mortality in absence of data.
	Purse-seine (OBJ)	**0.08**	**0.83** 62-d closure	**1.0** Year-round	**1.00** OBJ sets assumed to fish 0-200m. Species inhabits 0-100m (Musyl *et al.* 2003).	**0.47** Used logistic-shaped selectivity ogive for dominant EPO OBJ sets (Aires-da-Silva *et al.* 2014).	**1.0** Mandatory release, but assumed 100% mortality in absence of data.
Prionace glauca	**Longline**	**0.60**	**1.0** Year-round	**1.0** Year-round	**1.00** Deep sets assumed to fish 0-300m. Species inhabits 0-150m (Musyl *et al.* 2003).	**0.63** Used logistic-shaped selectivity ogive for dominant EPO fleet (ISC 2017).	**1.0** Assumed no release of marketable species.
Alopias superciliosus	**Longline**	**0.71**	**1.0** Year-round	**1.0** Year-round	**0.62** Deep sets assumed to fish 0-300m. Species inhabits 20-450m (Musyl *et al.* 2003).	**0.61** In absence of selectivity ogive for EPO longline fleet, used observer length-frequency data to assume knife-edge selectivity from 100cm FL.	**1.0** Assumed no release of marketable species.
	Purse-seine (DEL)	**0.28**	**0.83** 62-d closure	**1.0** Year-round	**0.40** DEL sets assumed to fish 0-200m. Species inhabits 20-450m (Musyl *et al.* 2003).	**1.00** In absence of selectivity ogive for EPO DEL sets, precautionary full selectivity of all length classes used.	**1.0** Assumed 100% mortality in absence of release data.
	Purse-seine (NOA)	**0.03**	**0.83** 62-d closure	**1.0** Year-round	**0.40** NOA sets assumed to fish 0-200m. Species inhabits 20-450m (Musyl *et al.* 2003).	**1.00** In absence of selectivity ogive for EPO NOA sets, precautionary full selectivity of all length classes used.	**1.0** Assumed 100% mortality in absence of release data.
	Purse-seine (OBJ)	**0.07**	**0.83** 62-d closure	**1.0** Year-round	**0.40** OBJ sets assumed to fish 0-200m. Species inhabits 20-450m (Musyl *et al.* 2003).	**1.00** In absence of selectivity ogive for EPO OBJ sets, precautionary full selectivity of all length classes used.	**1.0** Assumed 100% mortality in absence of release data.
Carcharhinus longimanus	**Longline**	**0.74**	**1.0** Year-round	**1.0** Year-round	**1.00** Deep sets assumed to fish 0-300m. Species	**0.72** In absence of selectivity ogive for EPO longline fleet, used	**1.0** Assumed 100% mortality

					inhabits 0-120m (Musyl *et al.* 2003).	observer length-frequency data to assume knife-edge selectivity from 70cm FL.	in absence of release data.
	Purse-seine (OBJ)	**0.07**	**0.83** 62-d closure	**1.0** Year-round	**1.00** OBJ sets assumed to fish 0-200m. Species inhabits 0-120m (Musyl *et al.* 2003).	**0.74** In absence of selectivity ogive, used IATTC observer length-frequency data to assume knife-edge selectivity from 70cm FL.	**1.0** Mandatory release, but assumed 100% mortality in absence of data.
Sphyrna zygaena	**Longline**	**0.63**	**1.0** Year-round	**1.0** Year-round	**1.00** Deep sets assumed to fish 0-300m. Species inhabits 0-60m (Francis 2016).	**0.67** In absence of selectivity ogive for EPO longline fleet, used observer length-frequency data to assume knife-edge selectivity from 80cm FL.	**1.0** Assumed 100% mortality in absence of release data.
	Purse-seine (OBJ)	**0.06**	**0.83** 62-d closure	**1.0** Year-round	**1.00** OBJ sets assumed to fish 0-200m. Species inhabits 0-60m (Francis 2016).	**0.78** In absence of selectivity ogive, used IATTC observer length-frequency data to assume knife-edge selectivity from 55cm FL.	**1.0** Assumed 100% mortality in absence of release data.
Isurus oxyrinchus	**Longline**	**0.65**	**1.0** Year-round	**1.0** Year-round	**1.00** Deep sets assumed to fish 0-300m. Species inhabits 0-150m (Musyl *et al.* 2003).	**0.79** In absence of selectivity ogive for EPO longline fleet, used observer length-frequency data to assume knife-edge selectivity from 65cm FL.	**1.0** Assumed no release of marketable species.
Lepidocybium flavobrunneum	**Longline**	**0.48**	**1.0** Year-round	**1.0** Year-round	**0.22** Deep sets assumed to fish 0-300m. Species inhabits 100-1000m (Kerstetter *et al.* 2008).	**1.00** In absence of selectivity ogive for EPO longline fleet, precautionary full selectivity of all length classes used.	**1.0** Assumed no release of marketable species.
Lampris guttatus	**Longline**	**0.59**	**1.0** Year-round	**1.0** Year-round	**0.62** Deep sets assumed to fish 0-300m. Species inhabits 50-400m (Polovina *et al.* 2008).	**0.72** In absence of selectivity ogive for EPO longline fleet, used Hawaiian market landings observer length-frequency data to assume knife-edge selectivity from 40cm FL (Sundberg and Underkoffler 2011).	**1.0** Assumed no release of marketable species.

TABLE 4. Biological parameters for 14 species assessed using EASI-Fish, with the first four species included in comparisons with integrated stock assessment results. Superscripts indicate the distribution type defined for priors used in 10,000 iterations of Monte Carlo simulations: N = normal, T = triangular, U = uniform. Values shown in parentheses are 95% confidence intervals for normal distribution priors, and upper and lower bounds for triangular and uniform distribution priors.

Species	L_{max} (TL)	t_{max} (yrs)	L_{inf} (yr^{-1})	K (yr^{-1})	t_0 (yr^{-1})	L-W a	L-W b	L_m (cm)	L_{50} (cm)	r	M (yr^{-1})
Thunnus albacares	220	5	198.9 (189.0-208.7)^{N}	0.341 (0.309-0.373)^{N}	0.002 (-0.141-0.145)^{N}	0.0139	3.086	L_{50} used	91.8 (88.6-95.0)^{N}	0.056 (0.055-0.057)^{N}	0.35 (0.20-0.50)^{T}
Thunnus obesus	250	16	200.8 (189.6-212.0)^{N}	0.440 (0.368-0.513)^{N}	1.260 (1.064-1.456)^{N}	0.0366	2.902	L_{50} used	138.2 (131.9-144.5)^{N}	0.168 (0.050-0.285)^{N}	0.20 (0.05-0.30)^{T}
Kajikia audax	420	11	256.5 (245.0-265.0)^{N}	0.600 (0.500-0.700)^{U}	-0.700 (-0.750--0.650)^{U}	0.0696	3.071	L_{50} used	210.0 (190.0-229.9)^{N}	0.090 (0.085-0.095)^{N}	0.50 (0.40-0.60)^{T}
Xiphias gladius	455	15	321 (304.7-337.3)^{N}	0.133 (0.118-0.148)^{N}	-2.460 (-2.653--2.267)^{N}	0.0045	3.210	L_{50} used	143.6 (140.8-146.4)^{N}	0.103 (0.090-0.110)^{N}	0.4 (0.2-0.6)^{T}
Coryphaena hippurus	210	4	140.5 (130.9-155.1)^{N}	0.67 (0.339-1.001)^{N}	-0.820 (-1.016--0.624)^{N}	0.0006	3.440	L_{50} used	77.0 (75.6-79.2)^{N}	0.110 (0.105-0.115)^{N}	1.0 (0.6-1.4)^{U}
Acanthocybium solandri	250	7	149.9 (135.0-165.0)^{N}	1.580 (1.380-1.780)^{N}	-0.170 (-0.250-0.090)^{N}	0.0009	3.280	L_{50} used	104.6 (95.0-110.0)^{N}	0.150 (0.140-0.160)^{N}	1.0 (0.8-1.1)^{U}
Carcharhinus falciformis	350	25	216.4 (210.5-222.3)^{N}	0.148 (0.089-0.207)^{N}	-1.760 (-1.799--1.721)^{N}	0.0273	2.860	L_{50} used	147.5 (144.6-150.4)^{N}	0.138 (0.118-0.158)^{N}	0.15 (0.10-0.20)^{U}
Prionace glauca	400	20	267.2 (260.1-274.3)^{N}	0.137 (0.133-0.141)^{N}	-1.130 (-1.230--1.030)^{N}	0.0041	3.160	L_{50} used	156.6 (154.6-158.6)^{N}	0.160 (0.121-0.199)^{N}	0.25 (0.15-0.35)^{T}
Alopias superciliosus	488	20	224.6 (222.7-226.5)^{N}	0.092 (0.091-0.093)^{N}	-4.210 (-4.268--4.152)^{N}	0.009	3.080	L_{50} used	180.2 (175.1-185.3)^{N}	0.600 (0.583-0.617)^{N}	0.15 (0.10-0.20)^{U}
Carcharhinus longimanus	400	11	244.6 (219.9-269.3)^{N}	0.103 (0.019-0.187)^{N}	-2.698 (-3.345--2.051)^{N}	0.0408	2.820	L_{50} used	140.5 (135.4-145.6)^{N}	0.230 (0.181-0.279)^{N}	0.20 (0.10-0.30)^{U}
Sphyrna zygaena	500	18	220.2 (214.7-225.7)^{N}	0.200 (0.180-0.219)^{N}	-0.710 (-0.712--0.708)^{N}	0.0117	2.770	114.9 (103.8-121.9)^{U}	L_m used	L_m used	0.20 (0.10-0.30)^{U}
Isurus oxyrinchus	445	29	269.5 (249.0-289.0)^{N}	0.115 (0.090-0.130)^{N}	-4.300 (-4.320--4.280)^{N}	0.0167	2.847	L_{50} used	256.0 (250.9-261.1)^{N}	0.160 (0.130-0.190)^{N}	0.15 (0.09-0.16)^{U}
Lepidocybium flavobrunneum	200	-	203.4 (199.9-206.9)^{U}	0.080 (0.070-0.090)^{U}	-1.290 (-1.320--1.270)^{U}	0.0048	3.152	104.4 (96.6-112.7)^{U}	L_m used	L_m used	0.15 (0.10-0.20)^{U}
Lampris guttatus	200	14	119 (114.3-123.7)^{N}	0.218 (0.177-0.259)^{N}	-0.780 (-1.368--0.192)^{N}	0.0281	3.000	80.0 (75.0-85)^{U}	L_m used	L_m used	0.35 (0.30-0.45)^{U}

TABLE 5. Sources of biological parameters used in EASI-Fish for assessing 14 species caught in the eastern Pacific Ocean tuna fishery. The first four species were included in comparisons with integrated stock assessment results.

Species	L_{max} (TL)	t_{max} (yrs)	L_{inf} (yr^{-1})	K (yr^{-1})	t_0 (yr^{-1})	L-W *a*	L-W *b*	L_m (cm)	L_{50} (cm)	*r*	*M* (yr^{-1})
Thunnus albacares	Froese and Pauly (2017)	Wild (1986)	Minte-Vera *et al.* (2015)	Minte-Vera *et al.* (2015)	Minte-Vera *et al.* (2015)	Wild (1986)	Wild (1986)		Schaefer (1998)	Schaefer (1998)	Minte-Vera *et al.* (2015)
Thunnus obesus	Froese and Pauly (2017)	Farley *et al.* (2006)	Aires-da-Silva *et al.* (2015)	Aires-da-Silva *et al.* (2015)	Aires-da-Silva *et al.* (2015)	Nakamura and Uchiyama (1966)	Nakamura and Uchiyama (1966)		Schaefer *et al.* (2005)	(Schaefer *et al.* 2005)	Aires-da-Silva and Maunder (2013)
Kajikia audax	Froese and Pauly (2017)	Kopf *et al.* (2005)	Kopf *et al.* (2011)	Kopf *et al.* (2011)	Kopf *et al.* (2011)	Wares and Sakagawa (1972)	Wares and Sakagawa (1972)		(Kopf *et al.* 2012)	Kopf *et al.* (2012)	Hinton (2009)
Xiphias gladius	Froese and Pauly (2017)	Hinton and Maunder (2011)	Cerna (2009)	Cerna (2009)	Cerna (2009)	Hinton and Maunder (2011)	Hinton and Maunder (2011)		DeMartini *et al.* (2000)	DeMartini *et al.* (2000)	Hinton and Maunder (2011)
Coryphaena hippurus	Froese and Pauly (2017)	Goicochea *et al.* (2012)	Aires-da-Silva *et al.* (2017)	Aires-da-Silva *et al.* (2011)	Aires-da-Silva *et al.* (2011)	(Guzman *et al.* 2015)	(Guzman *et al.* 2015)		Zúñiga-Flores *et al.* (2011)	Zúñiga-Flores *et al.* (2011)	Aires-da-Silva *et al.* (2011)
Acanthocybium solandri	Froese and Pauly (2017)	Zischke *et al.* (2013b)	Zischke *et al.* (2013b)	Zischke *et al.* (2013b)	Zischke *et al.* (2013b)	Zischke *et al.* (2013b)	Zischke *et al.* (2013b)		Zischke *et al.* (2013)	Zischke *et al.* (2013)	Estimated (Pauly$_{nls}$)
Carcharhinus falciformis	Froese and Pauly (2017)	Froese and Pauly (2017)	Oshitani *et al.* (2003)	Oshitani *et al.* (2003)	Oshitani *et al.* (2003)	Oshitani *et al.* (2003)	Oshitani *et al.* (2003)		Oshitani *et al.* (2003)	Oshitani *et al.* (2003)	Aires-da-Silva *et al.* (2014)
Prionace glauca	Froese and Pauly (2017)	Yokoi *et al.* (2017)	Yokoi *et al.* (2017)	Yokoi *et al.* (2017)	Yokoi *et al.* (2017)	Nakano (1994)	Nakano (1994)		Fujinami *et al.* (2017)	Fujinami *et al.* (2017)	ISC (2017)
Alopias superciliosus	Froese and Pauly (2017)	Liu *et al.* (1998)	Liu *et al.* (1998)	Liu *et al.* (1998)	Liu *et al.* (1998)	Froese and Pauly (2017)	Froese and Pauly (2017)		Liu *et al.* (1998)	Liu *et al.* (1998)	Fu *et al.* (2016)
Carcharhinus longimanus	Froese and Pauly (2017)	Seki *et al.* (1998)	Seki *et al.* (1998)	Seki *et al.* (1998)	Seki *et al.* (1998)	Seki *et al.* (1998)	Seki *et al.* (1998)		Seki *et al.* (1998)	Seki *et al.* (1998)	Estimated (Hoenig$_{nls}$)
Sphyrna zygaena	Froese and Pauly (2017)	Coelho *et al.* (2011)	Coelho *et al.* (2011)	Coelho *et al.* (2011)	Coelho *et al.* (2011)	Froese and Pauly (2017)	Froese and Pauly (2017)	Froese and Binohlan (2000)			Estimated (Hoenig$_{nls}$)
Isurus oxyrinchus	Froese and Pauly (2017)	Bishop *et al.* (2006)	Semba *et al.* (2011)	Semba *et al.* (2011)	Semba *et al.* (2011)	Bishop *et al.* (2006)	Bishop *et al.* (2006)		Semba *et al.* (2011)	Semba *et al.* (2011)	Bishop *et al.* (2006)
Lepidocybium flavobrunneum	Froese and Pauly (2017)	Froese and Pauly (2017)	Froese and Binohlan (2000)	Froese and Pauly (2017)	Froese and Pauly (2017)	Keller and Kerstetter (2014)	Keller and Kerstetter (2014)	Froese and Binohlan (2000)			Estimated (Pauly$_{LT}$)
Lampris guttatus	Froese and Pauly (2017)	Francis *et al.* (2004)	Francis *et al.* (2004)	Francis *et al.* (2004)	Francis *et al.* (2004)	Sundberg and Underkoffler (2011)	Sundberg and Underkoffler (2011)	Francis *et al.* (2004)			Francis *et al.* (2004)

TABLE 6. Values for fishing mortality (*F*) and spawning stock biomass (SSB) reference points derived from the EASI-Fish model for 14 species caught in purse seine and longline tuna fisheries in the eastern Pacific Ocean. Reference points used by SAFE (F_{msm} and F_{crash}) are also shown. Colors indicate if the current fishing mortality rate (F_{2016}) and spawning stock biomass (SSB_{2016}) exceed (red) or are less than (green) the relevant reference point.

Species	Code	F_{2016}	SSB_{2016}	F_{MSY}	SSB_{MSY}	$F_{0.1}$	$SSB_{0.1}$	$F_{40\%}$	$SSB_{40\%}$	F_{msm}	F_{crash}	F/F_{MSY}	SSB/SSB_{MSY}	$F/F_{0.1}$	$SSB/SSB_{0.1}$	$F/F_{40\%}$	$SSB/SSB_{40\%}$
Thunnus albacares	YFT	0.65	340.7	0.82	310.3	0.43	425.9	0.92	285.3	0.30	0.61	0.79	1.10	1.52	0.80	0.71	1.19
Thunnus obesus	BET	0.44	725.4	0.67	544.6	0.44	745.3	0.57	609.3	0.17	0.35	0.66	1.33	1.00	0.97	0.78	1.19
Kajikia audax	MLS	0.72	603.3	1.11	365.3	0.62	698.7	0.53	773.7	0.44	0.87	0.65	1.65	1.15	0.86	1.36	0.78
Xiphias gladius	SWO	0.21	166.4	2.48	47.76	0.92	88.11	0.78	94.31	0.35	0.70	0.09	3.48	0.23	1.89	0.27	1.76
Coryphaena hippurus	DOL	0.64	6.35	2.06	3.48	1.10	5.11	1.37	4.09	0.87	1.74	0.31	1.83	0.58	1.24	0.46	1.55
Acanthocybium solandri	WAH	0.36	46.07	4.24	21.97	2.06	30.61	4.37	20.96	0.87	1.74	0.08	2.10	0.17	1.51	0.08	2.20
Carcharhinus falciformis	FAL	0.70	33.68	0.47	58.39	0.21	98.43	0.29	76.21	0.06	0.12	1.50	0.58	3.42	0.34	2.42	0.44
Prionace glauca	BSH	0.43	24.73	0.68	18.94	0.36	33.51	0.34	32.01	0.10	0.21	0.63	1.31	1.21	0.74	1.26	0.77
Alopias superciliosus	BTH	0.47	0.87	0.30	3.45	0.19	6.81	0.12	9.99	0.06	0.12	1.57	0.25	2.47	0.13	3.82	0.09
Carcharhinus longimanus	OCS	0.83	14.44	0.37	81.71	0.21	127.6	0.20	115.1	0.08	0.16	2.24	0.18	4.00	0.11	4.19	0.13
Sphyrna zygaena	SPZ	0.59	20.45	0.45	30.65	0.27	43.58	0.36	33.88	0.08	0.16	1.31	0.67	2.19	0.47	1.63	0.60
Isurus oxyrinchus	SMA	0.62	0.13	0.22	3.55	0.15	6.89	0.08	11.64	0.06	0.12	2.80	0.04	4.26	0.02	7.62	0.01
Lepidocybium flavobrunneum	LEC	0.09	49.44	0.90	18.75	0.64	25.40	0.72	22.25	0.13	0.26	0.10	2.64	0.15	1.95	0.13	2.22
Lampris guttatus	LAG	0.26	8.68	1.35	2.10	0.79	4.22	0.61	5.17	0.30	0.61	0.19	4.13	0.33	2.06	0.43	1.68

TABLE 7. Comparison of productivity attributes used by the Sustainability Assessment of Impacts by Fisheries on Vulnerable Species (EASI-Fish) and a version of Productivity-Susceptibility Analysis (PSA) applied to six fisheries in the United States (Patrick *et al.* 2010).

	PSA	**EASI-Fish**
Productivity attribute		
Intrinsic rate of population increase (r)	X	
Maximum age (t_m)	X	X
Maximum size (L_{max})	X	X
Length-at-infinity (L_∞)		X
von Bertalanffy growth coefficient (K)	X	X
Natural mortality (M)	X	X
Fecundity	X	
Breeding strategy	X	
Recruitment pattern	X	
Age at maturity (t_m)	X	
Length-at-maturity (L_m or L_{50})		X
Mean trophic level	X	
Susceptibility attribute		
Areal overlap	X	X
Geographic concentration	X	
Fishing season duration		X
Vertical overlap (*i.e.* encounterability)	X	X
Seasonal availability	X	X
Schooling, aggregation, and behavioral responses	X	
Morphological characteristics affecting capture	X	
Gear selectivity		X
Desirability or value of the fishery	X	
Management strategy	X	
Fishing rate relative to M (equivalent to F-based BRPs)	X	X
Biomass of spawners (SSB) or other proxies (equivalent to spawning biomass-based BRPs)	X	X
Survival after capture and release	X	X
Impact of fisheries on essential fish habitat	X	