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Quantitative risk assessment methods with potential application to shark bycatch in the Indian Ocean

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1 Introduction

Risk assessment methods in fisheries are broadly concerned with estimating the rate of anthropogenic mortality to which a population is subjected. Risk typically refers to the probability of an undesirable outcome (Francis & Shotton, 1997), and fisheries management objectives usually include a reference to the relative population size (e.g., depletion) with a stipulation that the population should remain at or above this level over the long term. Risk then becomes the probability that fishing related mortality exceeds the maximum level allowed by this management objective.

Unfortunately, even if risk is intuitively expressed in terms of a probability, it cannot always be practically measured in this way. Often, risk assessment methods report risk qualitatively (i.e., the population is “at risk”) or on a comparative scale (i.e., the population is “the most at risk,” or even “the most likely to be at risk”). Only a subset of risk assessment methods are able to report risk in quantitative, probabilistic terms. Fisheries stock assessments provide an example. If mortality is too high, the stock is subject to over-fishing, which is by definition expected to lead to a decline in the population size away from the management target. The definition of over-fishing is therefore synonymous with the definition of risk, and fisheries stock assessment may be considered one form of risk assessment that is capable of producing a quantitative result.

Typically for bycatch species, however, population dynamics cannot be recreated because reliable time series of either catches or abundance are missing: even if current abundance is known, the depletion cannot be estimated. In these instances quantitative risk assessments focus on estimating an instantaneous mortality or rate of mortality (i.e., without reference to the temporal dynamics). This allows inference relative to the long term population status under current mortality levels, rather than current depletion, and is a feature that by convention differentiates risk assessment from more traditional fisheries stock assessment. The lower data requirements for risk assessment methods allow them to be applied more widely and they are therefore potentially useful for the assessment of fisheries non-target species and for Ecosystem-Based Fisheries Management (EBFM) in general (FAO, 2003).

2 Ecological Risk Assessement

Ecological Risk Assessment for the Effects of Fishing (ERAEF; Hobday et al., 2011) has become a useful and widely applied framework for conducting risk assessments, and is justified by it’s proponents as a means of supporting EBFM. The framework proposes a tiered approach, whereby a large number of species are progressively filtered using increasingly rigorous assessment techniques, culminating in a focused assessment of species or populations considered in the previous tiers to be most at risk. The first tier: Level 1; considers only the anthropogenic threats to the population, and uses a qualitative description of these threats to identify those species most likely to be at risk. At Level 2, susceptibility to these threats is more finely resolved, and a measure of population productivity is introduced. Species with high susceptibility and low productivity are considered at high risk relative to other candidates. Finally, at Level 3, risk is defined using quantitative methods within a statistical framework with the intention of estimating absolute rather than comparative risk, although this cannot always be achieved.

Following the terminology provided by Hobday et al. (2011), a Level 1 assessment is referred to as Scale Intensity Consequence Analysis (SICA, e.g.; Ford et al., 2018). This is a method of prioritisation that becomes unecessary when the threats and species of concern are already known or assumed and is not considered further here. A Level 2 assessment is referred to as a Productivity Susceptibility Analysis (PSA, e.g., Stobutzki et al., 2001, 2002). Using a PSA approach, species are assigned various productivity and susceptibility scores, often on a discrete scale (where sufficient life-history data exist it is possible to generate estimates of the intrinsic growth parameter as a quantitative measure of the productivity, and include this information in a PSA assessment). Summation of the scores on each axis allows a species to be plotted on two dimensions, and the Eucliden distance to the origin becomes a comparative measure of risk across species. Simplicity of the PSA approach has led to widespread application, including to sharks (e.g., Cortés et al., 2010, 2015, Arrizabalaga et al., 2011, Murua et al., 2009, 2012, 2018, Georgeson et al., 2020, Liu et al., 2021), even if it is not without cirticism (Zhou et al., 2016, Hordyk & Carruthers, 2018).

Level 3 quantitative risk assessment approaches, such as the Sustainability Assessment for Fishing Effects (SAFE; Zhou & Griffiths, 2008, Zhou et al., 2011, 2016, 2019) and Ecological Assessment of Sustainable Impacts of Fisheries (EASI-Fish; Griffiths et al., 2019) extend the PSA concept and estimate a fishing mortality that can be compared directly to a limit reference point. A defining feature of these approaches is that they include a more rigorous consideration of the spatial overlap. The Spatially Explicit Fisheries Risk Assessment (SEFRA; e.g., Waugh et al., 2008, 2015, Abraham et al., 2019, Richard et al., 2017, 2020, Large et al., 2019, Roberts et al., 2019, MacKenzie et al., 2022, Edwards et al., 2023) similarly includes an explicit representation of the species overlap with the fishery, and is now the primary risk assessment method applied to mega-fauna bycatch in New Zealand. In each of these Level 3 approaches, risk is expressed as a probability ratio that compares the estimated impact of fishing to a reference point thought to be consistent with management objectives.

Rate-based estimates of risk

The SAFE and SEFRA approaches both acknowledge the importance of spatial overlap in predicting fisheries risk (Queiroz et al., 2019), and further weight this overlap using an estimate of the fleet-specific catchability. However, whereas SAFE attempts to estimate the exploitation rate, SEFRA attempts to estimate the death explicitly. The estimated death from fishing is compared to a reference point that represents the total number of anthropogenic deaths that can be inflicted on the population whilst still meeting management objectives. The reference point used by SEFRA is analogous to the Potential Biological Removals (PBR; Wade, 1998, Taylor et al., 2000, Moore et al., 2013, Curtis et al., 2015) developed for the management of marine mammals and similarly requires knowledge of the population size to be calculated. This is a major shortcoming of SEFRA. It means that SEFRA cannot be applied to species where the population size is unknown. In constrast, SAFE does not require knowledge of the population size, because it estimates the exploitation rate directly and compares it to a rate-based reference point. In summary, whereas SEFRA and PBR approaches are notably dependent on the *number of* deaths, SAFE was developed for instances in which the population size is unknowable and designed to generate an estimate of risk based on the *rate of* fishery-dependent death.

To produce a reliable estimate of the exploitation rate, SAFE is dependent on well estimated catchability parameters. Initial iterations of the SAFE method either assumed a catchability or obtained it from auxiliary data. Subsequent improvements have allowed the catchability to be estimated within the model (Zhou et al., 2013, 2014, 2015), but a limitation of the SAFE approach is its high computational requirements. This has necessitated auxiliary information on species distributions (i.e., maps) to be available for risk to be estimated across the full spatial range of any species being assessed. This limitation has subsequently been addressed by developmental work in New Zealand, which has modified the SEFRA framework to allow estimation of the exploitation rate using statistical concepts introduced into fisheries by SAFE.

3 Recent methods development

Recent work in New Zealand (Neubauer et al., 2019, Edwards, 2023, Edwards et al., 2025) has proposed a risk assessment framework for pelagic shark species that may be applicable to shark bycatch in the Indian Ocean. The methodology uses the spatial overlap between fishing and the species distribution to define the exposure of a species to fishing effort. Exposure is converted to an exploitation rate via an estimated catchability term, and this exploitation rate is compared to a reference point referred to as the Impact Sustainability Threshold (IST).

Overview

The overlap for fishing event i is defined as the product of two probabilities: the probability of an individual being in spatial cell k (p_k), and the conditional probability of interaction with the fishing gear within that cell (s_i). Each fishing event belongs to a set of events defined by a unique combination of year j, grid cell k and fishing gear l.

The probability of interaction with the fishing gear within the cell is equal to the proportion of the cell area size affected by the fishing gear: i.e., $s_i = a_i/A_k$, where a_i is the gear affected area size and A_k is the grid cell area size, both of which are assumed to be known without error. The probability p_k defines the population's *probability distribution*. The notation n_{jk} is used for the *numbers distribution*, which is described by the parameters $\{p_{k=1}, p_{k=2}, \ldots, \mathrm{N}_j\}$, where N_j is the total population numbers in temporal stratum j. The probabilities p_k sum to one over the spatial domain of the population and the product of p_k and N_j equals the numbers in the grid cell at that time:

$$n_{jk} = p_k \cdot \mathrm{N}_j \tag{1}$$

This assumes that spatial and temporal effects act independently. From these prerequisites, the overlap metric is defined as:

$$\mathrm{O}_i = s_i \cdot p_k \tag{2}$$

which is the probability of an individual being in the cell multiplied by the probability of interacting with the fishing gear. The catchability is:

$$q_l = \pi_l \cdot \bar{a}_l \tag{3}$$

where π_l is defined by Paloheimo & Dickie (1964) as the probability of capture for individuals within the gear affected area. The notation $\bar{a}_l$ is used to represent the average gear affected area. This is required because the model estimates q_l as the product of π_l and $\bar{a}_l$. Pelagic species are often most at risk from surface longlines and by using this parameterisation it is acknowledged that the gear affected area is difficult to define for species that may or may not be attracted over large distances to baited fishing gear. Both π_l and $\bar{a}_l$ are dependent on the method of fishing and the gear type, which is referenced using the l subscript.

The population being fished is the *available* or *selected numbers*, and catchability is defined as conditional on the availability of individuals for capture by the fishing gear. The effect of areal availability on catches is represented by the spatial numbers distribution relative to fishing effort (i.e., as a determinant of n_{jk}). As is typical of swept-area approaches, where exploitation is a function of the two-dimensional overlap of fish and fishing (e.g., Walker et al., 2019), vertical availability is not explicitly represented (but see Griffiths et al., 2019, for a counter example). If necessary, it could be considered as an implicit determinant of π_l, which would require an assumption that fishing always takes place at the same depth relative to the depth distribution of whatever is being fished.

The catch equation can be written in terms of the numbers density:

$$\mathrm{C}_i = q_l \cdot d_{jk} \tag{4}$$

or as a function of the total population N_j, using the overlap notation:

$$\mathrm{C}_i = \pi_l \cdot \mathrm{O}_i \cdot \mathrm{N}_j \tag{5}$$

From this, it can be seen that for any given π_l and N_j, the catch per event increases with the overlap associated with that event, demonstrating the utility of this notation in providing a description of the exploitation. The exploitation rate per event, relative to the whole population, is the expected catch per individual, i.e., Equation 5 divided by N_j:

$$u_i = \pi_l \cdot \mathrm{O}_i \tag{6}$$

An important conclusion from this is that the exploitation rate can be written independently of the population size. Only the catchability and overlap are required. This is a characteristic of any system in which the catch is an increasing linear function of the available numbers: as the numbers change so will the catch, meaning that the exploitation rate will stay the same.

The population level exploitation rate per year is:

$$\mathrm{U}_j = \sum_{ikl} \pi_l \cdot \mathrm{O}_i \tag{7}$$

By taking the sum across events it is assumed that fishing events are independent (e.g., Zhou et al., 2011). This need not be the case (e.g., Mormede et al., 2021, Rowden et al., 2024), but is an adequate assumption for large and well mixed populations.

Parameterisation

With suitable data, it is possible to estimate the catchability term (Equation 4) from fits to the spatio-temporal catch and effort (Edwards, 2023, Edwards et al., 2025). This requires integration across the density surface, which is achieved using Bayesian MCMC. To date, applications have used the raw, unstandardised data, describing the catch in terms of the density d_{jk} and fleet or gear-specific catchabilities. Minimisation is helped through the prediction of d_{jk} within the model using environmental covariates, such as the sea-surface temperature.

Post-capture survival

Pelagic mega-fauna may be released alive following capture, and both the proportion of the catch that is discarded alive and the survival post-discard will influence the total number of fishing-related deaths. To include this in the model, the influence of post-capture release and survival is described by the variable Ψ_l, which is the product of: ψ_l, the probability of being discarded; and ω_l, the probability of post-discard survival (PDS). The discard fraction includes all discarded individuals whether dead or alive and PDS is a measure of the discarded fraction that survives. The product $\Psi_l = \psi_l \cdot \omega_l$ is therefore the overall probability of surviving capture. The adjusted exploitation rate is:

$$u_i = \pi_l \cdot O_i \cdot (1 - \Psi_l) \tag{8}$$

which will reduce the exploitation rate to zero as $\Psi_l \to 1$.

Unfished population size

The spatial domain of the risk assessment is defined by an area that contains the estimated population numbers distribution and the known fishing effort. However, movement is not restricted and individuals may leave this domain to varying degrees over time and according to the season. When outside of the assessment domain these individuals are not visible to the fishing effort included in the risk assessment but are potentially exposed to an unknown level of fishing pressure outside the risk assessment area. This is particularly important for the pelagic megafauna, for which the limits of the assessment domain represent the maximum extent that we are able to model, rather than the maximum extent of the population.

Typically, the spatial domain for the population, as distinct from the spatial domain for the assessment, cannot be defined using any available data. However, assuming that the population is well mixed, the full population domain should nevertheless be used for calculation of the exploitation rate. An adjustment to the exploitation rate is achieved using the variable Υ, which is the probability that an individual is outside the assessment domain. The adjusted exploitation rate is:

$$u_i = \pi_l \cdot O_i \cdot (1 - \Upsilon) \tag{9}$$

which, similar to Equation 8, will reduce the exploitation rate to zero as $\Upsilon \to 1$.

Risk

Given the catchability and a measure of overlap with the population, we are able to estimate the exploitation rate, which is the proportion of the exploitable numbers that experience mortality as a result of fisheries capture. The exploitable population is defined spatially, temporally, and also by the selectivity of the fishing gear. It can include parts of the population that are outside of the assessment domain (Equation 9), but does not include individuals that are not selected by any of the fishing gears included in the analysis.

Comparison of the exploitation rate with the IST reference point gives the risk ratio:

$$\text{Risk ratio}\,(R_j) = \frac{\text{Exploitation rate}\,(U_j)}{\text{IST}} \tag{10a}$$

The IST is defined as:

$$\text{IST} = \phi \cdot r_{\text{max}} \cdot \frac{1}{2} \tag{10b}$$

where r_{max} is the maximum intrinsic population growth rate (i.e., under optimal environmental conditions and in the absence of density dependent constraints). It is typically calculated from information on the fecundity, age at maturity and survivorship using theoretical approaches (Cortés & Travis, 2016).

The ϕ term is a tuning parameter set by management that will determine the status of the population at equilibrium when the risk ratio is equal to one. Under an assumption of deterministic, logistic growth, a value of $\phi = 1$ is equivalent to a management target for the population of half the carrying capacity, which is the depletion associated with an equilibrium harvest rate of $r_{\text{max}}/2$. Using Equation 10a, risk then becomes:

$$\text{Risk} = \mathbb{P}\,[\text{Risk ratio} > 1] \tag{11}$$

This is the probability that the exploitation rate is currently exceeding the IST. A low risk, assuming that the estimated exploitation rate remains constant, is synonymous with a high probability that this population status will be reached in the long term.

Assessment outputs

In summary, given the risk assessment structure described, the exploitation rate per year is:

$$\text{U}_j = \sum_{ikl} u_i \cdot (1 - \Psi_l) \cdot (1 - \Upsilon) \tag{12a}$$

where u_i is the exploitation rate per fishing event (defined per gear l, grid cell k and year j) and is calculated from the model estimated parameters using:

$$u_i = \frac{C_i}{N} = \frac{q_l \cdot p_k}{A_k} \tag{12b}$$

The exploitation rate U_j is the proportion of the available numbers that experience mortality as a result of being caught, with "available" defined as the population sum of all length classes that are able to be caught by at least one gear type, including individuals that are outside of the fished area. This rate is compared with the IST reference point to calculate the risk ratio (Equation 10a). Although the exploitation rate can be reported at any required resolution, the risk ratio is reported at the population level. This is because the IST reference point is a population level measure of the productivity. Given limitations in the data, the population size is assumed to be constant over time, but the exploitation rate, risk ratio and the risk (Equation 11) can be reported as time variant or as averages across a pre-specified time period.

Limitations

The model provides a coherent means of describing spatio-temporal catch and effort data in terms of catchability and population density parameters. However, explorations to date have not yet defined the conditions under which successful minimisation is achieved. Nor have simulations been conducted that would ascertain whether the catchability being estimated is reliable, noting that reliable estimation of this parameter is critical for the exploitation rate outputs. In the absence of further work, it is likely that estimated changes in the exploitation rate over time provide a more robust inference than estimates of the risk itself, although this is currently an area research.

4 Proposed research

It is proposed that data be collected on sharks and shark bycatch in the Indian Ocean for application of the methods described above, with a focus on silky shark (*Carcharhinus falciformis*), scalloped hammerhead (*Sphyrna lewini*) and oceanic whitetip (*Carcharhinus longimanus*). Application of the framework would most importantly require spatially resolved fishing effort and catch data, as well as an understanding of the spatial limits of the population being assessed.

Data requirements can be summarised as:

- Spatio-temporal catch and effort data for any fleets that catch the species in question within the IOTC area of competence;
- Knowledge of the spatial limits of the population, and environmental drivers of the spatial distribution, usually derived from expert opinion;
- Spatio-temporal covariates that may be useful for predicting the species distribution;
- Information or data on the probability of release (discard) and post-discard survival (PDS) rates;
- An estimate of the maximum intrinsic growth, or life-history data from which estimates can be calculated using theoretical approaches.

Application would require targeted methodological development to ensure parameter identifiability and robustness under the prevelant IOTC data conditions

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