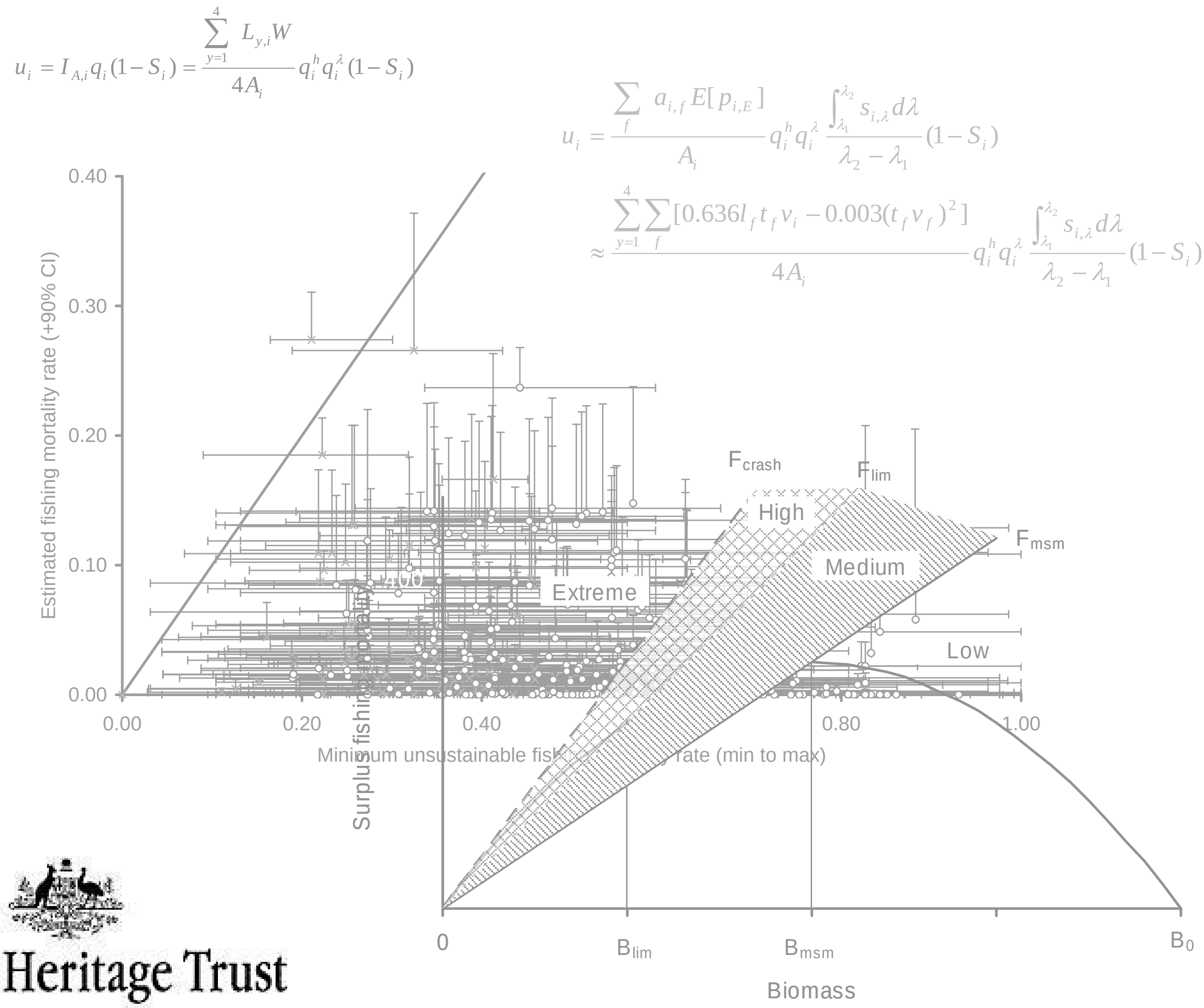
Rapid quantitative risk assessment for fish species in selected Commonwealth fisheries

Shijie Zhou, Tony Smith, Mike Fuller

December 2007
Australian Fisheries Management Authority



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Rapid quantitative risk assessment for fish species in major Commonwealth fisheries

Shijie Zhou, Tony Smith, Mike Fuller
December 2007
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NON-TECHNICAL SUMMARY

This project focuses on the rapid quantitative assessment of the risk from fishing to the sustainability of (mainly bycatch) species in several Commonwealth fisheries. The project has accomplished four major tasks: (1) developed and extended methods for quantitative sustainability assessment for fishing effects (SAFE) on data limited species; (2) completed sustainability assessment for the 5 sub-fisheries in the Southern and Eastern Scalefish and Shark Fishery (SESSF); (3) completed sustainability assessment for the Eastern Tuna and Billfish Fishery (ETBF); (4) reviewed the data availability for applications of SAFE to other Commonwealth managed fisheries and identified key information needs and analytical methods. This report also includes assessments on all bycatch fish species in the Northern Prawn Fishery (NPF), which were completed in a separate project.

Qualitative and semi-quantitative ecological risk assessments for the effects of fishing (ERAEF) have been conducted for most Commonwealth fisheries in recent years. These assessments used the Level 1 SICA (Scale Intensity Consequence Analysis) and Level 2 PSA (Productivity Susceptibility Analysis) methods in the ERAEF risk assessment framework. This project extends those analyses to provide quantitative estimates of risk for a large number of fish species in several fisheries. We used the SAFE method that was originally developed for the NPF and was largely independent of the SICA and PSA methods. Because of the fundamental differences between the two approaches and scientific rigor of the SAFE, to safeguard the Type I and Type II errors, we chose to assess all species rather than those that have been categorised as high risk in the Level 2 analysis. In this project we used the same data and included the same fish species as in the PSA analyses for these fisheries.

The SAFE framework includes two components: indicators and reference points. We focused on one single indicator—fishing mortality rate. We established reference points based on simple life history parameters to avoid the obstacle of formal stock assessment that requires more extensive fishery and fishery-independent data.

We developed new methods for estimating fishing mortality rate, based on limited data, for four gear types: bottom trawl, Danish seine, gillnet, and longline (demersal and pelagic longlines). The general approach involves estimating spatial overlap between species distribution and fishing effort distribution, catchability resulting from probability of encountering the gear and size-dependent selectivity, and post-capture mortality. The methods for gillnet and longline fisheries represent an extension of the basic SAFE methods. For the gillnet fishery, we first derived gear affected area from net dimension, soak time, and fish swimming speed. We then estimated the gear effective area by incorporating the probability of encountering the net for each individual within the gear affected area. Finally, we computed the fishing mortality rate from effective area, catchability, and post-capture mortality. For the longline fisheries, in addition to spatial overlap and catchability, we made use of known fishing mortality rates for target species to improve the accuracy of the estimates for the non-target species.

We defined three reference points: (1) u_{msm}--fishing mortality rates corresponding to the maximum sustainable fishing mortality (MSM) at B_{msm} (biomass that supports MSM, which is equivalent to MSY for target species); (2) u_{lim}--fishing mortality rate corresponding to limit biomass B_{lim}, where B_{lim} is defined as half of the biomass that supports a maximum sustainable mortality; and (3) u_{crash}--minimum unsustainable fishing mortality rate that, in theory, may lead to population extinction in the long term. For convenience, we labelled the risk categories as follows:

Low risk: fishing mortality rate u is less than u_{msm};
Medium risk: fishing mortality rate is greater than u_{msm} but less than u_{lim};
High risk: fishing mortality rate is greater than u_{lim} but less than u_{crash};
Extreme high risk: fishing mortality rate is greater than u_{crash}.

Each of these categories has a corresponding precautionary criterion which takes into account uncertainty in both estimated fishing mortality rate and reference point. We used six alternative methods to estimate these reference points and their feasible range (minimum to maximum values). These methods only require simple life history parameters, which may include intrinsic population growth rate, natural mortality rate, von Bertalanffy growth parameters, maximum reproductive age, and average age at maturity.

We carried out a sustainability assessment for all fish species (teleosts and chondrichthyans) in the SESSF. Based on the assumptions and methods used in SAFE, we found no species in the Great Australia Bight Trawl sub-fishery and Danish Seine sub-fishery where the estimated fishing mortality exceeded the MSM reference level. However, 24 species in the otter trawl sub-fishery, 11 species in the shark gillnet sub-fishery, and 13 species in the auto longline sub-fishery are in or above the precautionary medium risk category (i.e., either the mean fishing mortality rate u is greater or equal to the minimum u_{msm}, or the upper 90% of confidence interval of u is greater or equal to the mean u_{msm}) after experts' opinions are considered. Among these species, two in the otter trawl sub-fishery and three in the shark gillnet sub-fishery are in the high risk category (their mean fishing mortality rates are greater than their mean u_{lim}). Further, one species in the otter trawl

sub-fishery and two species in the shark gillnet sub-fishery are in the extreme high risk category (their mean fishing mortality rates are greater than their mean u_{crash}). Fishing has greater impact on sustainability of chondrichthyan than on teleost species, mainly because they generally have lower biological productivity.

Overall we assessed a total of 499 fish species in five sub-fisheries in SESSF, among which 99 are chondrichthyans and 400 are teleosts. These species lists were taken from the ERAEF lists for target, byproduct and bycatch species previously assessed for the fishery). We estimated that 72 species (39 chondrichthyans and 33 teleosts) were at precautionary medium risk by cumulative impacts where we assumed sub-fisheries within the jurisdiction impose impact on the same stock for each species. Among these species, experts have overridden at least 4 species based on their knowledge. For the remaining 68 species, 31 species are in medium risk category, 15 species in the high risk category, and 10 species in the extreme high risk category. We also examined the species that are at confidence risk, i.e., species whose **lower** 90% confidence limit of the estimated cumulative fishing mortality rate is greater than the mean value of a reference point, or species whose mean cumulative fishing mortality rate is greater than the **maximum** value of a reference point. We found that seven chondrichthyans and one teleost are at confident medium risk category, two chondrichthyans are at confident high risk category, one chondrichthyan is at confident extreme high risk category (i.e., the mean fishing mortality rate u is greater or equal to the maximum u_{crash}, or the lower 90% confidence interval of u covers the mean u_{crash}). However, these results have not taken experts' judgement separately on each sub-fishery into account.

We examined 207 fish species in the ETBF fishery and found seven species (5 chondrichthyans and 2 teleosts) are at precautionary medium risk category (i.e., either the mean fishing mortality rate u is greater or equal to the minimum u_{msm}, or the upper 90% of confidence interval of u is greater or equal to the mean u_{msm}) after experts' overriding on another four species. Only one species has an estimated mean fishing mortality rate greater than its mean u_{msm}. Among the seven species at precautionary medium risk, six are at precautionary high risk and three species are in the precautionary extreme high risk category.

We reviewed the availability of suitable data for other Commonwealth-managed fisheries, including 12 major fisheries and their 26 sub-fisheries. The results indicate that there are sufficient data to carry out quantitative risk assessment for the majority of these remaining fisheries, assuming similar methods developed in this report. For a few fisheries that do not have species spatial distribution information, or hook and line fisheries that do not have known fishing mortality rates for target species, alternative methods may need to be developed.

We discuss the pros and cons of the methods and emphasize specific issues encountered in this project. We also propose provisional management rules for non-target species and provide recommendation for future research in this area.

Sustainability for fishing effects has been assessed for fish bycatch in the Northern Prawn Fishery in a separate project. We include the results from the previous assessment as two appendices, one for the elasmobranchs and the other for teleosts.

The work needed to complete these analyses goes well beyond that originally anticipated. We developed new methods for estimating fishing mortality rate for various gear types and extended the biological reference point concept from a previous assessment in the NPF. We have also undertaken additional analyses, including estimating cumulative impacts from several sub-fisheries.

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CHAPTER 1. OVERVIEW

Qualitative and semi-quantitative ecological risk assessments for the effects of fishing (ERAEF) have been conducted for most Commonwealth fisheries in recent years (Smith et al. 2007). These assessments used the Level 1 and Level 2 method in the ERAEF risk assessment framework (Hobday et al., 2007). While these methods provide a useful screening tool to prioritise species and habitats, they do not provide absolute estimates of risk from fishing. This project extends those analyses to provide absolute (albeit uncertain) estimates of risk for a large number of mainly bycatch species in several fisheries.

The Australian Fisheries Management Authority (AFMA) is in the process of developing an Ecological Risk Management (ERM) response to the ERAs available so far, and will benefit from a better understanding of the quantitative risks, particularly for bycatch species and threatened, endangered and protected (TEP) species. The need is to mobilise and analyse existing quantitative data, particularly for high priority species arising from existing ERA analyses. Quantitative estimates of risk for such species will allow AFMA to better target their ERM response, providing precautionary reference points and other direct inputs for management. AFMA requires that this work be undertaken quickly, which implies that the project will need to use existing data and methods to complete the quantitative risk assessments for those species where suitable data are available. As top priorities, AFMA asked that the focus in the short term be on the bycatch fish (throughout this report, “fish” refers to teleost and chondrichthyan species) species for the Northern Prawn Fishery (NPF), the Southern and Eastern Scale Fish and Shark Fisheries (SESSF) and the Eastern Tuna and Billfish Fishery (ETBF).

The objectives of this project include: providing quantitative estimates of fishing impacts on all fish bycatch species and their sustainability risk for NPF and SESSF where suitable data are available; review quantitative data available for bycatch species for ETBF and conduct a quantitative risk assessment, and review data availability for other Commonwealth fisheries and identify key information needs and analytical methods.

In this project we carried out a quantitative sustainability assessment for fishing effects (SAFE) for all fish species (including target species, by-product species, by-catch species, and a few TEP species) in the SESSF and ETBF, and revised recent assessment for fish bycatch species in the NPF (Brewer et al. 2006; Zhou and Griffith in press). We estimated fishing mortality rates by comparing spatial distribution of species and fisheries as well as utilizing existing data from the previous ERA project. We established three sustainability reference points for bycatch species: maximum sustainable fishing mortality (similar to the maximum sustainable yield for target species), limit fishing mortality, and the minimum unsustainable fishing mortality. In practice, we estimated fishing mortality rates corresponding to these mortalities as actual management reference points. We derived these mortality rates by using simple life history parameters that have been obtained in the previous ERA project. The work needed to complete these analyses goes well beyond that originally anticipated. We developed new methods for estimating fishing mortality rate for various gear types and extended biological reference point concept from a previous assessment in the NPF. We also undertook additional analyses, including estimating cumulative impacts from several sub-fisheries in the SESSF.

This report is structured to present quantitative risk assessments for selected Commonwealth managed fisheries. Chapter 2 describes detailed methods for estimating fishing mortality rates for different gear types and methods for establishing biological reference points. It also illustrates the theoretical concept of suggested reference points and provides provisional fishery control rules for managing bycatch species that have low economic values. Chapter 3 is an assessment of the Southern and Eastern Scalefish and Shark Fishery. It includes separate assessment outcomes for each sub-fishery and the cumulative impacts from all sub-fisheries within the SESSF jurisdiction. Chapter 4 presents the assessment results for the Eastern Tuna and Billfish Fishery. It briefly describes the assessment method for this fishery. Chapter 5 reviews the data availability for potential application of the SAFE approach in all other Commonwealth managed fisheries. We discuss the pros and cons of the methods and provide recommendation for future research and management in Chapter 6. Assessment of fish bycatch in the NPF is attached as two appendices at the end of the report.

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CHAPTER 2. SAFE METHODOLOGY

The methodology of SAFE (sustainability assessment for fishing effects) consists of two major components: indicators and reference points. To that extent, the framework is consistent with the approach used for the Commonwealth Harvest Strategy Policy that applies to target and significant byproduct species. It also reflects the general approach advocated for ecosystem-based fishery management (Garcia and Staples 2000; Sainsbury et al. 2000; Garcia and Cochrane 2005). In this case we focus on one single indictor — fishing mortality rate — and develop methods to estimate this indicator for hundreds of species by using limited available data. As it is literally infeasible to do full stock assessments for hundreds of non-target species that have little information, we look for alternative approaches to establish reference points based on simple life history traits. The achievement of such a framework enables high-level policy goals in the Ecosystem Approach to Fisheries to be implemented at an operational level (FAO 2003). The SAFE method extends to Level 3 the framework developed in the ERAEF approach, and continues the process of building practical scientific tools to support ecosystem based fisheries management (Smith et al 2007).

2.1.1 Estimating fishing impacts

2.1.1.1 *Otter Trawl Fishery*

Fishing impact is expressed as annual mortality rate within the specific fishery management jurisdiction. The mean fishing mortality rate *u* is derived from fishing activity overlapping with species core distribution area within the fishery jurisdictional boundary (Figure 2-1), adjusted by the probability of being caught by the trawl. Note that *u* is the fraction of population killed by a fishery, not the instantaneous fishing mortality rate. We use actual logbook data from 2003 to 2006 to map effort distribution, while Bioregional mapping and Core range species mapping provide species distribution (Commonwealth of Australia 2005; Heap et al. 2005). Fishing mortality rate for species *i* is:

$$u_i = I_{A,i} q_i (1 - S_i) = \frac{\sum_{y=1}^{4} L_{y,i} W}{4A_i} q_i^h q_i^\lambda (1 - S_i), \qquad (1)$$

where $I_{A,i}$ = fraction of distribution area for species i impacted (trawled) by fishing activity, q_i is the conditional probability of an individual being caught in the trawl when it is on the fishing ground, and S_i is the post-capture survival rate (so 1-S_i is the post-capture mortality rate), $L_{y,i}$ = total trawl length in year y that occurs within the species distribution range, W = width of trawl wing spread, A_i = area of species distribution within the fishery boundary, $q^h{}_i$ = habitat-dependent encounterability, and $q^\lambda{}_i$ = size- and behaviour-dependent selectivity. This formulation is similar to estimating fishing mortality for elasmobranchs (Walker 2005). In this equation, L_y, q_i, and S_i are treated as random variables, while A_i and W are considered constants since species distribution is mapped by habitat and trawl wing spread is more or less fixed. Equation (1) assumes that there would be no local depletion effects from repeat trawls at the same location, i.e., populations rapidly mix between trawled and untrawled area. The fishing mortality will likely be overestimated under this assumption. For the SE otter trawl fishery, we use W = 23 m (Larcombe et al. 2001).

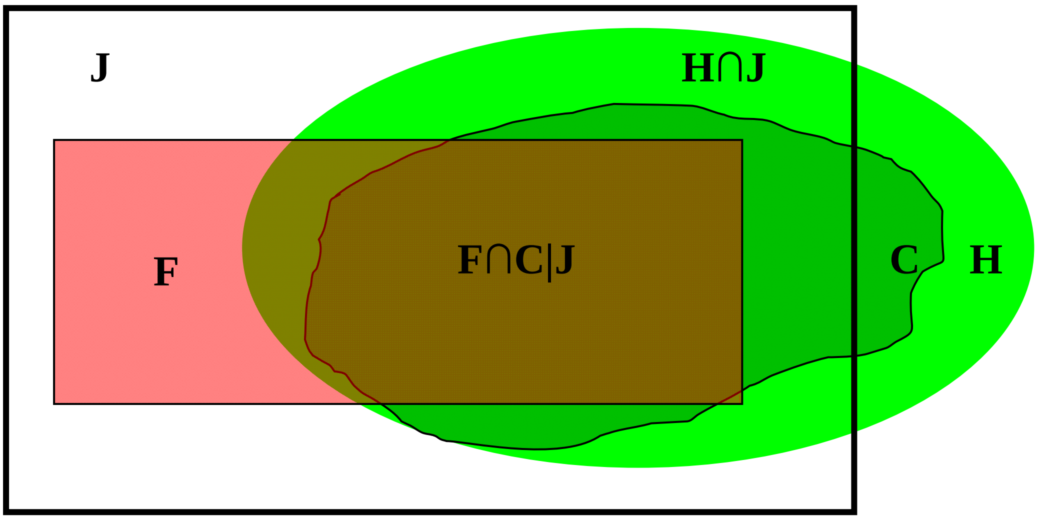

Figure 2-1. Diagram of species, fishery, and fishing effort distribution. J = SESSF jurisdiction; H = species distribution (habitat for one species based on Bioregionalisation mapping area); C = core distribution area of the species; F = fished (trawled) area. The overlap between fished area and species core distribution area, F∩C|J, is particularly important for SAFE.

Catchability q_i results from two factors: size and behaviour dependent catchability or selectivity $q^\lambda{}_i$ and habitat related encounterability $q^h{}_i$. We adopted $q^\lambda{}_i$ = 0.3, 0.47, and 1.0 for small (< 22.5 cm), larger (≥ 22.5 but < 100 cm), and very large and slow moving species (≥ 100 cm) (Blaber et al. 1990), but we also modified the very large species category and used q^{λ}_i = 0.47 for species between 400 and 500 cm, and q^{λ}_i = 0.3 for species > 500 cm. In the PSA analysis used in ERAEF, habitat types of demersal, soft bottom, sand, mud, benthopelagic bottom, and midwater are considered as high risk; habitat type of hard bottom (less likely to be trawled) is considered medium risk, and habitat types of benthopelagic bottom and midwater, and mesopelagic midwater are considered as low risk. We used q^h_i = 0.33, 0.67, and 1.0 for species that live in habitats with low, medium, and high risk of being caught in the trawl based on the PSA analyses (Wayte et al. 2006). Post-capture survival rate results from two separate processes: surviving handling on the deck and surviving after being returned to the water. We assumed S_i = 0.0, 0.34, and 0.67 for species that have low, medium, and high probability of surviving after being caught and returned to the water. To estimate uncertainty, we assume catchability and survival rate follow binomial distributions. For simplicity, we use a delta method to calculate the variance of fishing mortality rate u_i (Zhou 2002):

$$V[u_i] = \left(\frac{W}{A_i}\right)^2 \left(V[\bar{L}_y][q_i(1-S_i)]^2 + \frac{q_i(1-q_i)}{n-1}[\bar{L}_y(1-S_i)]^2 + \frac{S_i(1-S_i)}{n-1}(\bar{L}_y q_i)^2 \right) \tag{2}$$

We used species core distribution as primary distribution range. For a few species that do not have core distribution information we used Bioregional mapping data (Commonwealth of Australia 2005; Heap et al. 2005). A critical assumption is that mean fish density for each species does not vary between trawled area and non-trawled area within their distribution range. Level of risk will be over-estimated for species found primarily in non-trawl habitat, while risk will be under-estimated for species that prefer trawl habitat.

2.1.1.2 *Danish seine*

The Danish seine fishery uses the vessel to tow the net to encircle the fish. Similar to otter trawl, the fishing mortality rate is estimated by

$$u_i = I_{A,i} q_i (1-S_i) = \frac{\sum_{y=1}^{4} \sum_{f} a_{i,f} f_{y,i}}{4A_i} q_i^h q_l^{\lambda} (1-S_i) \tag{3}$$

In the SESSF, the length of the rope ranged from 1,000 to 2,800 m. We used 2,000 m as circumference to estimate fished area in one operation: $a_{i,f} = \pi R^2 = \pi (2000/2\pi)^2$, which results in a 0.32 km^2/shot. The annual total affected area is the sum of fishing efforts $f_{y,i}$ (number of shots) multiplied by $a_{i,f}$. Similar to the trawl fishery, we used size-dependent selectivity q^{λ}_i and habitat-dependent encounterability q^h_i and set them to 0.33, 0.66, and 1.0 for species with low, medium, and high selectivity scores and encounterability scores in the PSA analysis (Wayte et al. 2007). We also assumed S_i = 0.00, 0.34, and 0.67 for species that have low, medium, and high probability of surviving after being caught and returned to the water.

2.1.1.3 *Shark gillnet*

One of the problems in analysing the shark gillnet fishery is the low resolution grids (30 min by 30 min grid) on which the catch and effort data are reported, resulting in high overlap between fishing effort and species distribution within the jurisdictional boundary. Below we derive a method to obtain a more realistic estimate of the fished area.

The affected fishing area (i.e., the maximum area within which a fish could encounter the net), is a function of gillnet length, soak time, and swimming speed of fish (Figure 2-2). The gear affected area a_i during one fishing operation (shot) is species-specific and can be estimated as (Griffiths et al., in press):

$$a_i = 2\, l\, D_i + \pi D_i^2. \qquad (4)$$

Where
a_i = affected fishing area by gillnet for species i;
l = gillnet length;
D_i = maximum distance from the net. $D_i = t\, v_i$, where t = net soak time and v_i = sustained swimming speed for species i.

The probability of a fish at any (x, y) position encountering the net (which ranges between 0 and 0.5) can be obtained by:

$$p_{i,E}(encounter \mid x, y) = \frac{\alpha}{2\pi}.$$

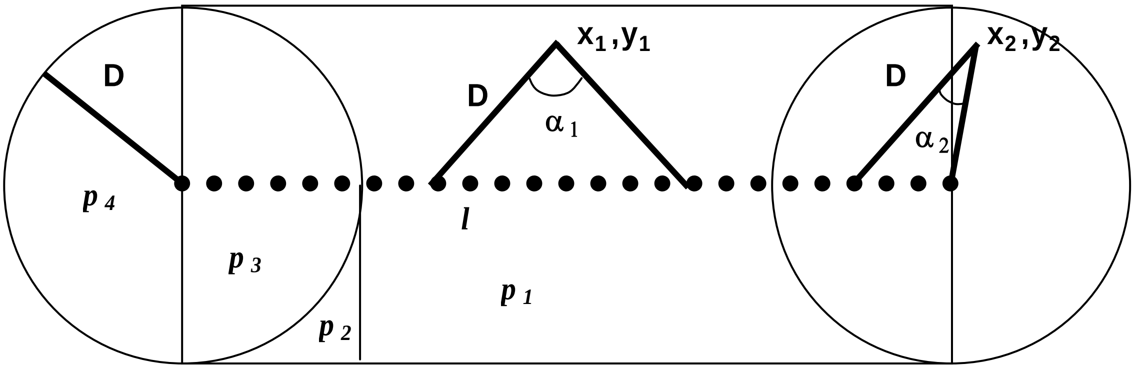

Figure 2-2. Diagram of gillnet affected fishing area. D = maximum distance from the net during the soak time that a fish can encounter the net.

We divide the affected area into four sections to estimate the expected probability of encountering the net. For these four sections, the expected probability of encountering the net is approximately 0.318 (1/π), 0.197, 0.30, and 0.05 for E[p_1], E[p_2], E[p_3], E[p_4], respectively. The overall encounter probability within the affected area is:

$$E[p_{i,E}(encounter \mid a_i, x, y)] = \frac{1}{a_i}\iint \frac{\alpha_{x,y}}{2\pi} dxdy \approx \frac{0.636lD_i - 0.003D_i^{\ 2}}{2lD_i + \pi D_i^{\ 2}} \qquad (5)$$

Using this approach, the annual fishing mortality rate in the gillnet fishery is estimated as:

$$\begin{aligned} u_i &= \frac{\sum_f a_{i,f} E[p_{i,E}]}{A_i} q_i^h q_i^\lambda \frac{\int_{\lambda_1}^{\lambda_2} s_{i,\lambda} d\lambda}{\lambda_2 - \lambda_1}(1 - S_i) \\ &\approx \frac{\sum_{y=1}^{4}\sum_f [0.636 l_f t_f v_i - 0.003(t_f v_f)^2]}{4A_i} q_i^h q_i^\lambda \frac{\int_{\lambda_1}^{\lambda_2} s_{i,\lambda} d\lambda}{\lambda_2 - \lambda_1}(1 - S_i) \end{aligned},$$

(6)

where $q^h{}_i$ is habitat related encounterability of species *i*, $q^\lambda{}_i$ is the size-dependent maximum overall fishing power (gear efficiency), $s_{i,\lambda}$ is the relative gear selectivity on size λ, λ_1 and λ_2 are size range of species *i* caught by the gillnet, and *f* is fishing activity (shot). In eq. (5), $a_{i,f}$

$E[p_{i,E}]$ is the "effective fishing area", which is a theoretical area where all individuals have 100% probability of encountering the net (Hovgard and Lassen, 2000).

Based on 2006 logbook data, the mean gillnet length used in SESSF shark gillnet fishery is 3.7 km (range 1.0 to 4.2, SD = 0.87, n = 10850), the mean mesh size is 15.8 cm (range 10.0 to 16.8 cm, SD =0.7, n = 10850), and the mean net soak time is 2.39 h (range 0.25 to 24.0 h, SD = 1.50, n = 10850). We obtained sustainable swimming speed for a few species. For the majority of species, we use fish body length (average length at maturity) to estimate sustainable swimming speed: $v_i = \lambda_{mat,\, i}^{0.8}$ (Blake 1983). This method may over- or under-estimate the swimming speed. In addition, in estimating the affected area, we assume a fish continuously swims in a straight line and in a fixed but random direction (i.e. in any direction around 2π radius).

We used q^h_i = 0.33, 0.66, and 1.0 for species that live in habitat with low, medium, and high risk of encountering gillnet based on the PSA analyses (Walker et al. 2007). For the size-dependent overall catchability we use the size-dependent selectivity score from the PSA analysis and assume q^λ_i = 0.33, 0.66, and 1.0 for low, medium, and high scores (Walker et al. 2007). As a gillnet is a reasonably selective gear, only a fraction of the population will be retained by the gear even when the q^λ_i = 1.0 for that species. The last term in equation (5) estimates this fraction of the population retained. Kirkwood and Walker (1986) and Walker (2005) derived the selectivity pattern for gummy shark (*Mustelus antarcticus*), school shark (*Galeorhinus galeus*), elephant fish (*Callorhinchus milii*), common sawshark (*Prisiophorus cirratus*), and southern sawshark (*P. nudipinnis*) in this fishery. They assumed the selectivity curve to follow a gamma function:

$$s_{i,\lambda} = \left(\frac{\lambda}{\alpha\beta}\right)^{\alpha} \exp\left(\alpha - \frac{\lambda}{\beta}\right), \qquad (7)$$

where α and β depend on mesh size. For commercial gillnets used in the SESSF (mean mesh size = 15.8 cm, SD = 0.7, n = 10850), we derived α and β values for these five species. In this paper, we assumed the average shape of these five selectivity curves is the same for all fish species, but the mode can vary from species to species. For such a selectivity curve, we estimated about 40% of the population for a species having a q^λ_i = 1.0 with sizes corresponding to $s_{i,\lambda} \geq 1\%$ could be selectively retained. Again, we assumed S_i = 0.00, 0.34, and 0.67 for species that have low, medium, and high probability of surviving after being caught and returned to the water.

2.1.1.4 *Automatic longline*

Auto longlines employed in the SESSF typically measure over 6 km and fish over 10 hours before being retrieved. Fishing effort per year averages more than one thousand sets, with a total of over 7 million hook lifts per year during 2003 to 2006. Until recently, the Longline fishery also reported catch and effort at low spatial resolution (30 min by 30 min grid). We refined the fishing effort distribution from the original ½ degree blocks to the areas within each block with a depth range of 200-700 meters, corresponding to the main depths fished by this gear. Area overlaps between fishing effort and species distribution were then calculated based on each ½ degree block within this depth range. Fishing effort (number of hooks) was also allocated using the refined range.

We use the following method to estimate fishing mortality rate for the longline fishery:

$$u_i = \frac{A_{i,f}}{A_{i,J}} q_i^h q_i^\lambda \rho(1 - S_i), \qquad (8)$$

where $A_{i,f}$ is the area (30 min by 30 min grid) within species i's core distribution area and where longline fishing activity has been recorded during 2003 – 2006 period, and $A_{i,J}$ is the total core distribution area for species i within the fishery jurisdiction. For a few species that do not have core distribution information, we used their distribution based on Bioregionalisation mapping. This ratio between the two areas is essentially the fraction of species spatial distribution overlapping with the longline fishery. Again, the habitat-dependent encounterability $q^h{}_i$ is set to 0.33, 0.66, and 1.0 for species with low, medium, and high scores of encountering the fishing gear in the PSA analysis. We assigned the size-dependent catchability $q^\lambda{}_i$ based on average length at maturity as in PSA: 0.33 for fish < 10 cm or > 500 cm, 0.66 for fish between 10 and 20 cm and between 400 and 500 cm, and 1.0 for fish between 20 and 400 cm (Daley et al. 2007). The additional parameter ρ can be considered as a correction factor for the combination of affected area (due to bait odour dispersion and fish movement), probability of responding to bait, probability of encountering the hooks, gear efficiency after encountering, etc. We derived this parameter from target species in the auto longline fishery as:

$$\rho = \frac{1}{n}\sum_{i=1}^{n} \frac{u_i^T A_{i,J}^T}{q_i^h q_i^\lambda A_{i,f}^T}, \qquad (9)$$

where u_i^T is the exploitation rate for target species *i*. Auto longline targets two main species: blue eye trevalla (*Hyperoglyphe Antarctica*) and pink ling (*Genypterus blacodes*). Using various methods (catch curve, modified catch curve, and regression) Fay (2006) estimated that the total mortality *Z* of blue eye trevalla varied between 0.158 and 0.201. It is assumed the natural mortality *M* = 0.08 for this species. Based on these data, we estimated that the exploitation rate *u* for the non-trawl fisheries is approximately 8% (ranging from 4% to 14%, SD = 0.02). Taylor (Marine and Freshwater Fisheries Research institute, personal communication) estimated the non-trawl exploitation rate for pink ling as *u* = 13.6% (ranging from 0.05 to 0.23, SD = 0.06) for different regions in SESSF from 2003 to 2005. Based on these target species, we calculated ρ = 0.16 (SE = 0.11). We used S_i = 0.00, 0.34, and 0.67 for species that have low, medium, and high probability of surviving after being caught and returned to the water.

2.1.1.5 *Cumulative impacts*

Theoretically, the methods for estimating fishing mortality rate developed in this report are quantitative. Fishing impacts by multiple fisheries can be added together to derive cumulative impacts. That is, the total annual fishing mortality rate for species *i* is:

$$U_i = \sum_f u_i \ . \qquad (10)$$

The assumption behind this equation is that sub-fisheries within the jurisdiction impose impact on the same stock for each species. To estimate uncertainty we simply assumed that the operation of sub-fisheries is independent of each other so the variance associated with u_i in each sub-fishery can be summed to obtain the total uncertainty.

2.1.2 Deriving sustainability reference points

Because population sizes (abundance or biomass) are extremely difficult to estimate for hundreds of bycatch species, we focused on the relative quantity--the fishing mortality rate--as the most easily obtainable management reference point. We defined the following three biological reference points based on a simple surplus production model (Figure 2-3):

u_{msm} = fishing mortality rates corresponding to the maximum sustainable fishing mortality (MSM) at B_{msm} (biomass that supports MSM);

u_{lim} = fishing mortality rate corresponding to limit biomass B_{lim}, where B_{lim} is defined as half of the biomass that supports a maximum sustainable fishing mortality ($0.5B_{msm}$); and

u_{crash} = minimum unsustainable fishing mortality rate that, in theory, will lead to population extinction in the longer term.

We assumed these reference points to be a function of basic life history parameters of each species. Specifically, we linked them to the intrinsic population growth rate r and instantaneous natural mortality M. Many species have published estimates for r and/or M. We also estimated M based on growth parameters, maximum length, environmental temperature, longevity, and age at maturity. We applied a total of six methods to derive these reference points:

$F_{msm} = r/2$, $F_{lim} = 0.75\ r$, and $F_{crash} = r$;

$F_{msm} = M$, $F_{lim} = 1.5\ M$, and $F_{crash} = 2M$;

$F_{msm} = M$, $F_{lim} = 1.5\ M$, and $F_{crash} = 2M$, where $\ln(M) = -0.0152 - 0.279\ln(L_\infty) + 0.6543\ln(k) + 0.4634\ln(T)$ (Pauly 1980; Quinn and Deriso 1999);

$F_{msm} = M$, $F_{lim} = 1.5\ M$, and $F_{crash} = 2M$, where $\ln(M) = 1.44 - 0.982\ \ln(t_m)$ (Hoenig 1983).

$F_{msm} = M$, $F_{lim} = 1.5\ M$, and $F_{crash} = 2M$, where $M = 10^{0.566-0.718\ln(L_\infty)} + 0.02T$ (www.Fishbase.org);

$F_{msm} = M$, $F_{lim} = 1.5\ M$, and $F_{crash} = 2M$, where $M = 1.65/t_{mat}$ (Jensen 1996);

In these equations, k and L_∞ are von Bertalanffy growth parameters, T = average annual water temperature, t_m = maximum reproductive age, and t_{mat} = average age at maturity. If L_∞ is unknown but the maximum length L_{max} is known, we estimated length infinity as: $\log(L_\infty) = 0.044 + 0.9841\log(L_{\max})$ (Froese and Binohlan 2000). Considering the uncertainty in the parameters themselves that come from the literature and from applying the methods, we gave equal weight to these six methods to derive the mean and ranges of u_{msm}, u_{lim}, and u_{crash}.

To compare fishing mortality rate and the reference points, we converted the instantaneous rate to the fraction of population by $u = 1 - \exp(-F)$. Note here that we did not include natural mortality. The full formula is: $u = F/(F+M)[1-\exp(-F-M)]$ (Quinn and Deriso 1999). However, since we estimated the fishing mortality rate based on area overlapping rather than initial abundance at the beginning of the year, natural mortality occurs simultaneously in fished and unfished area, and is not included in the estimation (actually it can be considered that *M* has happened and continues to happen in both fished and unfished area but cancels out each other. Our estimated *u* could equal 1 if the overlap between fishing effort and species distribution is 100% and the overall catchability is 1.0. However, *u* will never be 1 if we include natural mortality in the estimation.)

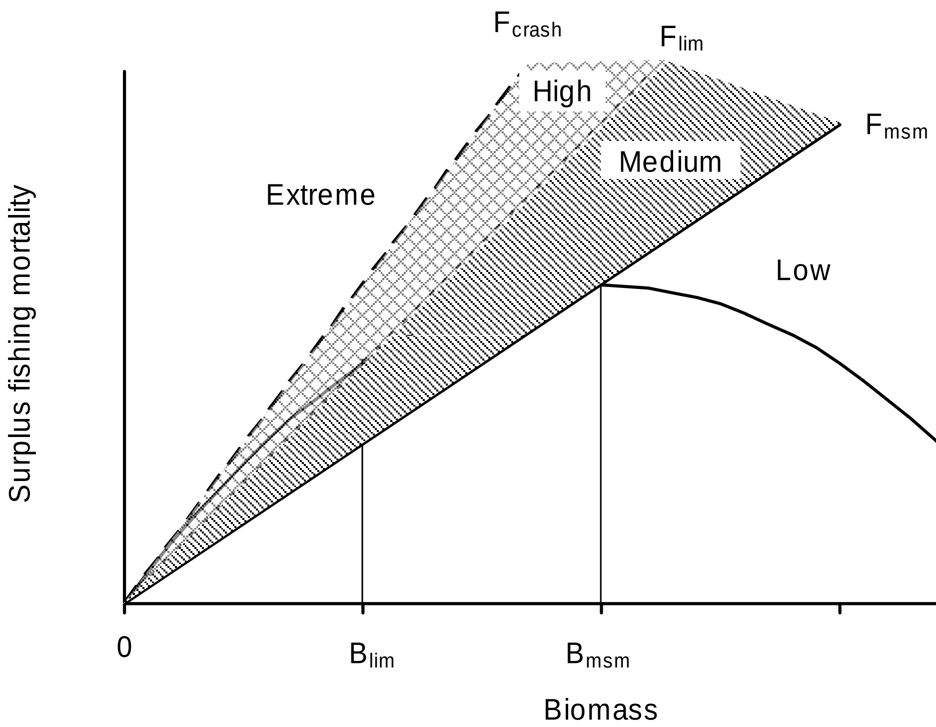

Figure 2-3. Stock productivity, biological reference points and ecological risk assessment categories for managing bycatch species

Because input parameters for estimating fishing mortality rates and reference points typically involve large uncertainty, as well as the simplicity of the method, the results also have high uncertainty for many species. To link with harvest strategy policy for target species, we may consider overfishing – fishing impacts that drive the population below the level that can support its maximum sustainable fishing mortality- as the primary concern. We used point estimates and their uncertainty in categorising risk level as follows:

Low risk (L): $E[u] < E[u_{msm}]$;

Medium risk (M): $E[u_{msm}] \leq E[u] < E[u_{lim}]$;

Precautionary medium risk (m): $E[u] \geq \min[u_{msm}]$ or $E[u]$ + 90%CI $\geq E[u_{msm}]$;

High risk (H): $E[u_{lim}] \leq E[u] < E[u_{crash}]$;

Precautionary high risk (h): $E[u] \geq \min[u_{lim}]$ or $E[u]$ + 90%CI $\geq E[u_{lim}]$;

Extreme high risk (E): $E[u] \geq E[u_{crash}]$;

Precautionary extreme high risk (e): $E[u] \geq \min[u_{crash}]$ or $E[u]$ + 90%CI $\geq E[u_{crash}]$.

Using instantaneous fishing mortality F, we present these risk categories and the corresponding ecological consequence in Table 2-1. When taking uncertainty into account, we may prefer the precautionary risk categories (i.e., codes m, h, and e).

Table 2-1. Biological reference points, proposed ecological risk assessment category, ecological consequence, and provisional management rules for bycatch species

	$\mathbf{F < F_{msm}}$	$\mathbf{F_{lim} > F > F_{msm}}$	$\mathbf{F_{crash} > F > F_{lim}}$	$\mathbf{F > F_{crash}}$
ERA risk	Low (L)	Medium (M)	High (H)	Extreme high (E)
Ecological consequence	Overfishing not occurring. May keep population above 50% of virgin level	Overfishing is occurring but population can be sustainable	May drive population to very low levels in longer term	Population is unsustainable in long term – possibility of extinction
Management rule	Reduction of F not needed	Reduction in F may be required if this level of F occurs over seven consecutive years	Reduce fishing mortality below F_{msm} if this F occurs in five consecutive years	Reduce fishing mortality below F_{msm} if this F occurs in three consecutive years

Because the estimated fishing mortality rate and reference points may contain high uncertainty, we are interested in species for which we have higher confidence that they are in a certain risk category. This is opposite to the precautionary risk consideration and we refer to it as “confidence risk’ assessment. We defined confident risk as follows and used the following method for categorising the cumulative impact only.

Confident medium risk (1): $E[u] \geq \max[u_{msm}]$ or $E[u]$ - 90%CI $\geq E[u_{msm}]$;

Confident high risk (2): $E[u] \geq \max[u_{lim}]$ or $E[u]$ - 90%CI $\geq E[u_{lim}]$;

Confident extreme high risk (3): $E[u] \geq \max[u_{crash}]$ or $E[u]$ - 90%CI $\geq E[u_{crash}]$.

To date, we have not seen clear harvest control rules being developed for bycatch species that have low economic values. Considering the possible negative impacts on the fishing industry, the simplicity of our assessment approach, and the uncertainty associated with the estimates, we tentatively propose a provisional management strategy for bycatch species in Table 2-1. These management rules clearly need further discussion and consideration, as the time frames suggested are more or less arbitrary.

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CHAPTER 3. ASSESSMENT ON SOUTHERN AND EASTERN SCALE FISH AND SHARK FISHERY (SESSF)

The Southern and Eastern Scalefish and Shark Fishery (SESSF) extends from waters off southern Queensland, south and west to Cape Leeuwin in Western Australia (Figure 3-1). It is a complex multi-sector, multi-gear and multi-species fishery targeting scalefish and shark stocks of various size, distribution and composition. Almost half the waters of the Australian Fishing Zone off southern mainland Australia and Tasmania are in the fishery management area. The SESSF is one of the most important Commonwealth-managed fisheries, with landings of over 35,000 t annually at a value of around $95 million. We assessed five major sub-fisheries in the SESSF using the method developed in Chapter 2: the South East Otter trawl fishery, the Great Australian Bight Trawl Fishery, the Danish Seine Fishery, the Shark Gillnet Fishery, and the Auto Longline Fishery.

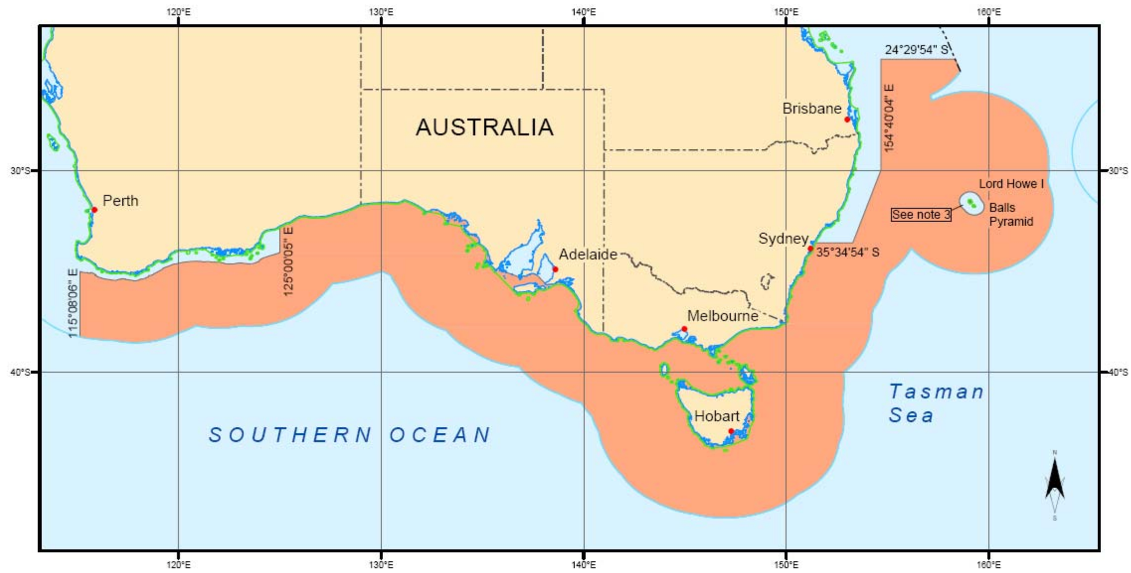


Figure 3-1. Area of the Southern and Eastern Scalefish and Shark Fishery.

3.1 South East Trawl Sub-fishery

We assessed 440 species of fish (88 chondrichthyans, 352 teleosts) caught in the otter trawl fishery. Among these species 411 (including all chondrichthyans) have spatial distribution information (i.e. overlapping with fishery or outside the boundary).

Four species have estimated mean fishing mortality rate greater than mean u_{msm} (*Dipturus gudgeri*, *Centrophorus squamosus, Eptatretus longipinnis,* and *Odontaspis ferox*) (Figure 3-2). Three species are chondrichthyans and one is teleost (hagfish). The first two species are also found to be at high risk ($E[u] \geq E[u_{lim}]$, Figure 3-3). If we include uncertainty in both estimated fishing mortality rate and the reference points, 38 species are at risk of potential overfishing (precautionary medium risk, $E[u] \geq \min[u_{msm}]$, or $E[u]$ + 90% CI $\geq E[u_{msm}]$) (Table 3-1). One species, *Dipturus gudgeri*, is found at risk level of unsustainable ($E[u] \geq E[u_{crash}]$). However, if we consider uncertainties, there are 11 species have $E[u] \geq \min[u_{crash}]$ or $E[u]$ +90% CI $\geq E[u_{crash}]$ (Table 3-1, Figure 3-4).

Note that Table 3-1 includes all precautionary risk species when uncertainty is taken into account. For most of these species, they are in the list of potential risk not because of their point estimates but uncertainty associated with the estimates. The upper 90% CI of the fishing mortality rates may have been overestimated or the minimum value of a reference points underestimated for some species. Biologists with first-hand knowledge of the SESSF scrutinized these listed species and believed the results are credible for most species. However, they overrode some species based on their experience and felt uncertain for an additional few species, which are listed on the lower part of Table 3-1. Specific comments for these species are as follows.

Isistius brasiliensis (cookie cutter shark): a benthopelgic species rarely caught by demersal trawl.

Caelorinchus mirus (gargoyle fish) and *Caelorinchus fasciatus* (banded whiptail): they are part of a species complex that has not been consistently identified. If they are at risk, the other related species (*C. maurofasciatus* and *C. parvifasciatus*) may also be at risk.

Hoplostethus latus (giant sawbelly): a rare species and may be a data outlier.

Caelorinchus fasciatus (banded whiptail): Uncertain about the vertical overlap with the trawl fishery.

Lepidorhynchus denticulatus (toothed whiptail): abundant species caught in large numbers, a benthopelagic species spending a lot of time in mid-water. Catchability *q* may have been overestimated.

Odontaspis ferox (sand tiger shark) and *Galeocerdo cuvier* (tiger shark): appear to have lower encounterability with this sub-fishery.

Centrolophus niger (rudderfish): pelagic species and should have a lower encounterability.

Malacocephalus laevis (smooth whiptail): Uncertain about the vertical overlap with the trawl fishery.

Hoplostethus intermedius (common sawbelly): a rare species and may have been misidentified.

Gephyroberyx darwinii (Darwin's roughy): a relatively uncommon tropical species occasionally caught in quantity.

Lepidopus caudatus (Southern frostfish): Uncertain about the vertical overlap with the trawl fishery.

Zenopsis nebulosus (mirror dory): a benthopelagic species with apparently high spatial variability (and perhaps recruitment success). If this species is at risk then its close relative, the king dory, may also be at risk.

Note that *Centrophorus squamosus* is probably *Centrophorus moluccensis*. *Centrophorus uyato* is recognized as *Centrophorus zeehani*, and *Squalus mitsukurii* is recognized as *Squalus chloroculus*.

Regulation for South East Trawl sub-fishery changed in 2007, which prohibits trawling in water deeper than 700 m. AFMA is interested to know the impact of this change on bycatch species. We repeated the sustainability assessment for this sub-fishery by eliminating fishing effort that occurred in water deeper than 700 m in 2003-2006. We did not consider potential effort re-distribution into shallower water. This assessment may underestimate the actual impact if the total fishing effort has not been reduced.

This scenario removes three species out of Table 3-1 (Table 3-2): *Oreosoma atlanticum* (Oxeye Oreo), *Gephyroberyx darwinii* (darwin's roughy), and *Neocyttus rhomboidalis* (Spiky Oreo). Also, *Centrophorus squamosus* (nilson's deepsea dogfish) changes from high risk category (H) to medium risk (M).

Table 3-1. Species at potential risk of overfishing in the SESSF **otter trawl sub-fishery**. Species are sorted by [u - u_{msm}]. I_A = fraction of distribution area impacted; q = overall catchability, S = post-capture survival rate, Method = methods used for estimating the reference points. See text for risk category codes. Species without a risk code are in the precautionary medium risk (m) only.

					u		u_{msm}		u_{lim}		u_{crash}			
Scientific name	Common name	I_A	q	1-S	Mean	se	Mean	Min	Mean	Min	Mean	Min	Method	Risk
Dipturus gudgeri	bight skate	0.27	1.00	1.00	0.27	0.02	0.11	0.09	0.16	0.13	0.21	0.16	23456	EeHhM
Centrophorus squamosus	nilson's deepsea dogfish	0.18	1.00	1.00	0.18	0.02	0.12	0.05	0.17	0.07	0.22	0.09	23456	eHhM
Eptatretus longipinnis	hagfish	0.45	0.47	1.00	0.21	0.08	0.17	0.17	0.24	0.24	0.31	0.31	356	ehM
Deania quadrispinosa	Platypus Shark	0.23	0.47	1.00	0.11	0.04	0.12	0.04	0.17	0.05	0.22	0.07	23456	eh
Centrophorus harrissoni	Harrison's dogfish	0.23	0.47	1.00	0.11	0.04	0.13	0.06	0.18	0.09	0.23	0.12	23456	h
Squalus mitsukurii	Green-Eyed Dogfish	0.28	0.47	1.00	0.13	0.05	0.15	0.09	0.21	0.13	0.27	0.16	123456	h
Centroscymnus plunketi	plunket's shark	0.10	1.00	1.00	0.10	0.01	0.12	0.07	0.17	0.11	0.22	0.14	456	
Centrophorus uyato (east)	southern dogfish	0.22	0.47	1.00	0.10	0.04	0.13	0.06	0.19	0.09	0.25	0.12	23456	h
Dipturus australis	common skate	0.09	0.47	1.00	0.04	0.02	0.08	0.03	0.12	0.05	0.16	0.06	23456	
Polyprion oxygeneios	Hapuku	0.27	0.31	1.00	0.08	0.04	0.13	0.07	0.18	0.11	0.24	0.14	23456	
Dipturus sp. B	grey skate	0.24	0.47	1.00	0.12	0.04	0.18	0.08	0.25	0.12	0.32	0.16	23456	
Hydrolagus lemures	bight ghost shark	0.35	0.47	1.00	0.17	0.06	0.23	0.20	0.33	0.28	0.41	0.36	356	
Hyperoglyphe antarctica	Blue Eye Trevalla	0.30	0.47	1.00	0.14	0.05	0.21	0.10	0.30	0.15	0.38	0.19	123456	
Etmopterus lucifer	Blackbelly Lantern Shark)	0.18	0.47	1.00	0.09	0.03	0.17	0.06	0.23	0.09	0.29	0.11	23456	
Cephaloscyllium sp. A [in Last	Whitefin Swell Shark	0.29	0.47	1.00	0.14	0.05	0.22	0.10	0.31	0.15	0.39	0.20	23456	
Bassanago bulbiceps	swollen-headed conger eel	0.26	0.47	1.00	0.12	0.04	0.22	0.12	0.31	0.17	0.38	0.22	2356	
Caelorinchus kaiyomaru	whiptail	0.21	0.47	1.00	0.10	0.04	0.20	0.07	0.28	0.10	0.35	0.13	23456	
Azygopus pinnifasciatus	righteye flounder	0.30	0.47	1.00	0.14	0.05	0.25	0.10	0.34	0.15	0.43	0.20	123456	
Epigonus lenimen	big-eyed cardinalfish	0.20	0.30	1.00	0.06	0.03	0.17	0.02	0.23	0.02	0.27	0.03	456	eh
Caelorinchus innotabilis	notable whiptail	0.18	0.47	1.00	0.08	0.03	0.20	0.07	0.28	0.10	0.35	0.13	23456	
Helicolenus barathri	Ocean Perch - Offshore	0.29	0.30	1.00	0.09	0.05	0.20	0.07	0.28	0.10	0.35	0.14	23456	
Epigonus robustus	robust cardinalfish	0.17	0.30	1.00	0.05	0.03	0.17	0.02	0.22	0.02	0.27	0.03	456	eh
Epigonus denticulatus	white cardinalfish	0.19	0.30	1.00	0.06	0.03	0.17	0.02	0.23	0.02	0.28	0.03	456	eh
Neocyttus rhomboidalis	Spiky Oreo	0.10	0.47	1.00	0.04	0.02	0.17	0.04	0.24	0.07	0.30	0.09	123456	
Oreosoma atlanticum	Oxeye Oreo	0.12	0.47	1.00	0.05	0.02	0.23	0.04	0.31	0.06	0.38	0.08	123456	
Species overriden by experts														
Odontaspis ferox	sand tiger shark	0.32	1.00	0.66	0.21	0.05	0.18	0.10	0.26	0.15	0.32	0.19	23456	ehM
Isistius brasiliensis	cookie-cutter shark (cigar shark)	0.27	0.47	1.00	0.13	0.05	0.14	0.06	0.20	0.09	0.26	0.11	23456	eh
Hoplostethus latus	giant sawbelly	0.43	0.30	1.00	0.13	0.07	0.16	0.05	0.23	0.07	0.29	0.10	123456	eh
Caelorinchus mirus	gargoyle fish	0.29	0.47	1.00	0.14	0.05	0.20	0.07	0.28	0.10	0.35	0.13	23456	eh
Caelorinchus fasciatus	banded whiptail	0.28	0.47	1.00	0.13	0.05	0.20	0.07	0.28	0.10	0.35	0.13	23456	h
Lepidorhynchus denticulatus	Toothed Whiptail	0.25	0.47	1.00	0.12	0.04	0.20	0.07	0.28	0.10	0.35	0.13	23456	h
Centrolophus niger	Rudderfish	0.24	1.00	1.00	0.24	0.02	0.28	0.21	0.39	0.29	0.48	0.37	123456	
Galeocerdo cuvier	Tiger Shark	0.10	1.00	1.00	0.10	0.01	0.17	0.08	0.25	0.11	0.31	0.15	123456	
Malacocephalus laevis	smooth whiptail	0.24	0.47	1.00	0.11	0.04	0.20	0.07	0.28	0.10	0.35	0.13	23456	h
Hoplostethus intermedius	common sawbelly	0.27	0.30	1.00	0.08	0.04	0.16	0.05	0.23	0.07	0.29	0.10	123456	h
Gephyroberyx darwinii	darwin's roughy	0.17	0.30	1.00	0.05	0.03	0.16	0.05	0.23	0.07	0.29	0.10	123456	
Lepidopus caudatus	Southern Frostfish	0.31	0.47	1.00	0.14	0.05	0.29	0.14	0.39	0.20	0.48	0.26	23456	
Zenopsis nebulosus	Mirror Dory	0.28	0.47	1.00	0.13	0.05	0.25	0.10	0.35	0.14	0.43	0.18	123456	

Table 3-2. Species at potential risk of overfishing in the SESSF **otter trawl sub-fishery after effort in water deeper than 700 m was removed**. Species are sorted by [u - u_{msm}]. I_A = fraction of distribution area impacted; q = overall catchability, S = post-capture survival rate, Method = methods used for estimating the reference points. See text for risk category codes. Species without a risk code are in the precautionary medium risk (m) only.

					u		u_{msm}		u_{lim}		u_{crash}			
Scientific name	Common name	I_A	q	1-S	Mean	se	Mean	Min	Mean	Min	Mean	Min	Method	Risk
Dipturus gudgeri	bight skate	0.27	1.00	1.00	0.27	0.02	0.11	0.09	0.16	0.13	0.21	0.16	23456	EeHhM
Eptatretus longipinnis	hagfish	0.45	0.47	1.00	0.21	0.08	0.17	0.17	0.24	0.24	0.31	0.31	356	ehM
Centrophorus squamosus	nilson's deepsea dogfish	0.16	1.00	1.00	0.16	0.01	0.12	0.05	0.17	0.07	0.22	0.09	23456	ehM
Deania quadrispinosa	Platypus Shark	0.22	0.47	1.00	0.10	0.04	0.12	0.04	0.17	0.05	0.22	0.07	23456	eh
Centrophorus harrissoni	Harrison's dogfish	0.23	0.47	1.00	0.11	0.04	0.13	0.06	0.18	0.09	0.23	0.12	23456	h
Squalus mitsukurii	Green-Eyed Dogfish	0.28	0.47	1.00	0.13	0.05	0.15	0.09	0.21	0.13	0.27	0.16	123456	h
Centrophorus uyato (east)	southern dogfish	0.22	0.47	1.00	0.10	0.04	0.13	0.06	0.19	0.09	0.25	0.12	23456	h
Centroscymnus plunketi	plunket's shark	0.08	1.00	1.00	0.08	0.01	0.12	0.07	0.17	0.11	0.22	0.14	456	
Dipturus australis	common skate	0.09	0.47	1.00	0.04	0.02	0.08	0.03	0.12	0.05	0.16	0.06	23456	
Polyprion oxygeneios	Hapuku	0.27	0.31	1.00	0.08	0.04	0.13	0.07	0.18	0.11	0.24	0.14	23456	
Dipturus sp. B	grey skate	0.24	0.47	1.00	0.11	0.04	0.18	0.08	0.25	0.12	0.32	0.16	23456	
Hydrolagus lemures	bight ghost shark	0.35	0.47	1.00	0.17	0.06	0.23	0.20	0.33	0.28	0.41	0.36	356	
Hyperoglyphe antarctica	Blue Eye Trevalla	0.30	0.47	1.00	0.14	0.05	0.21	0.10	0.30	0.15	0.38	0.19	123456	
Cephaloscyllium sp. A [in Last	Whitefin Swell Shark	0.29	0.47	1.00	0.14	0.05	0.22	0.10	0.31	0.15	0.39	0.20	23456	
Etmopterus lucifer	Blackbelly Lantern Shark)	0.16	0.47	1.00	0.07	0.03	0.17	0.06	0.23	0.09	0.29	0.11	23456	
Bassanago bulbiceps	swollen-headed conger eel	0.26	0.47	1.00	0.12	0.04	0.22	0.12	0.31	0.17	0.38	0.22	2356	
Caelorinchus kaiyomaru	whiptail	0.20	0.47	1.00	0.09	0.03	0.20	0.07	0.28	0.10	0.35	0.13	23456	
Azygopus pinnifasciatus	righteye flounder	0.30	0.47	1.00	0.14	0.05	0.25	0.10	0.34	0.15	0.43	0.20	123456	
Helicolenus barathri	Ocean Perch - Offshore	0.29	0.30	1.00	0.09	0.05	0.20	0.07	0.28	0.10	0.35	0.14	23456	
Epigonus denticulatus	white cardinalfish	0.19	0.30	1.00	0.06	0.03	0.17	0.02	0.23	0.02	0.28	0.03	456	eh
Epigonus lenimen	big-eyed cardinalfish	0.17	0.30	1.00	0.05	0.03	0.17	0.02	0.23	0.02	0.27	0.03	456	eh
Caelorinchus innotabilis	notable whiptail	0.15	0.47	1.00	0.07	0.03	0.20	0.07	0.28	0.10	0.35	0.13	23456	
Epigonus robustus	robust cardinalfish	0.14	0.30	1.00	0.04	0.02	0.17	0.02	0.22	0.02	0.27	0.03	456	eh
Species overriden by experts														
Odontaspis ferox	sand tiger shark	0.32	1.00	0.66	0.21	0.05	0.18	0.10	0.26	0.15	0.32	0.19	23456	ehM
Isistius brasiliensis	cookie-cutter shark (cigar shark)	0.27	0.47	1.00	0.13	0.05	0.14	0.06	0.20	0.09	0.26	0.11	23456	eh
Hoplostethus latus	giant sawbelly	0.43	0.30	1.00	0.13	0.07	0.16	0.05	0.23	0.07	0.29	0.10	123456	eh
Centrolophus niger	Rudderfish	0.23	1.00	1.00	0.23	0.02	0.28	0.21	0.39	0.29	0.48	0.37	123456	
Caelorinchus mirus	gargoyle fish	0.29	0.47	1.00	0.14	0.05	0.20	0.07	0.28	0.10	0.35	0.13	23456	eh
Caelorinchus fasciatus	banded whiptail	0.28	0.47	1.00	0.13	0.05	0.20	0.07	0.28	0.10	0.35	0.13	23456	h
Galeocerdo cuvier	Tiger Shark	0.10	1.00	1.00	0.10	0.01	0.17	0.08	0.25	0.11	0.31	0.15	123456	
Lepidorhynchus denticulatus	Toothed Whiptail	0.25	0.47	1.00	0.12	0.04	0.20	0.07	0.28	0.10	0.35	0.13	23456	h
Hoplostethus intermedius	common sawbelly	0.27	0.30	1.00	0.08	0.04	0.16	0.05	0.23	0.07	0.29	0.10	123456	h
Malacocephalus laevis	smooth whiptail	0.24	0.47	1.00	0.11	0.04	0.20	0.07	0.28	0.10	0.35	0.13	23456	h
Zenopsis nebulosus	Mirror Dory	0.28	0.47	1.00	0.13	0.05	0.25	0.10	0.35	0.14	0.43	0.18	123456	
Lepidopus caudatus	Southern Frostfish	0.31	0.47	1.00	0.14	0.05	0.29	0.14	0.39	0.20	0.48	0.26	23456	

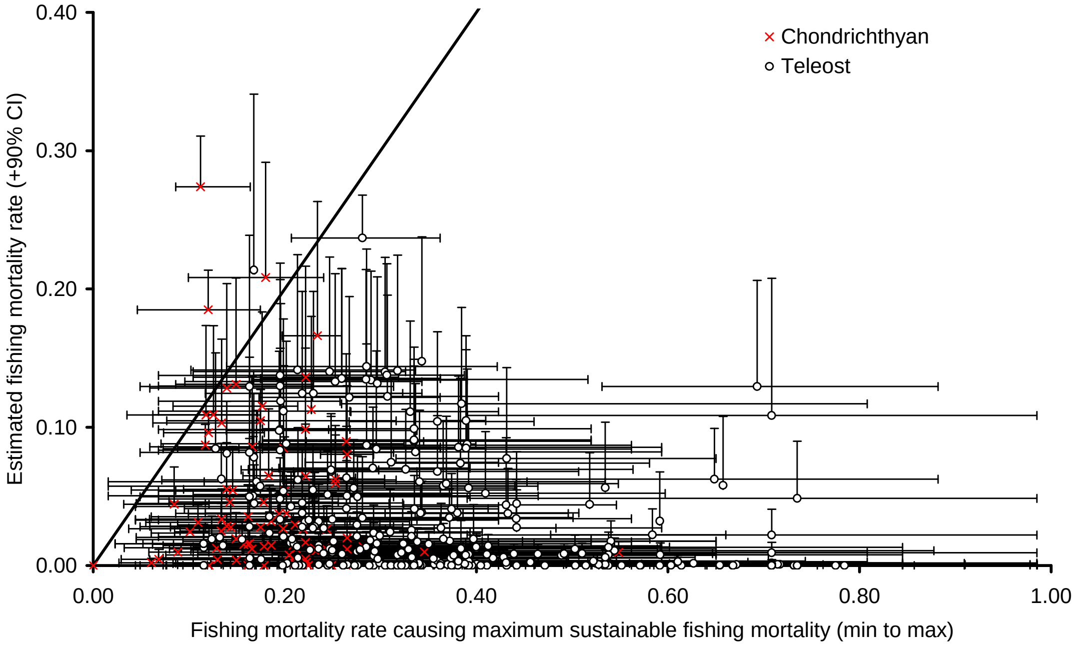


Figure 3-2. Comparison of estimated fishing mortality rate within the fishery jurisdiction and the fishing mortality rate corresponding to the maximum sustainable mortality for fish species caught in the SESSF otter trawl sub-fishery. The diagonal line is where $u = u_{msm}$.

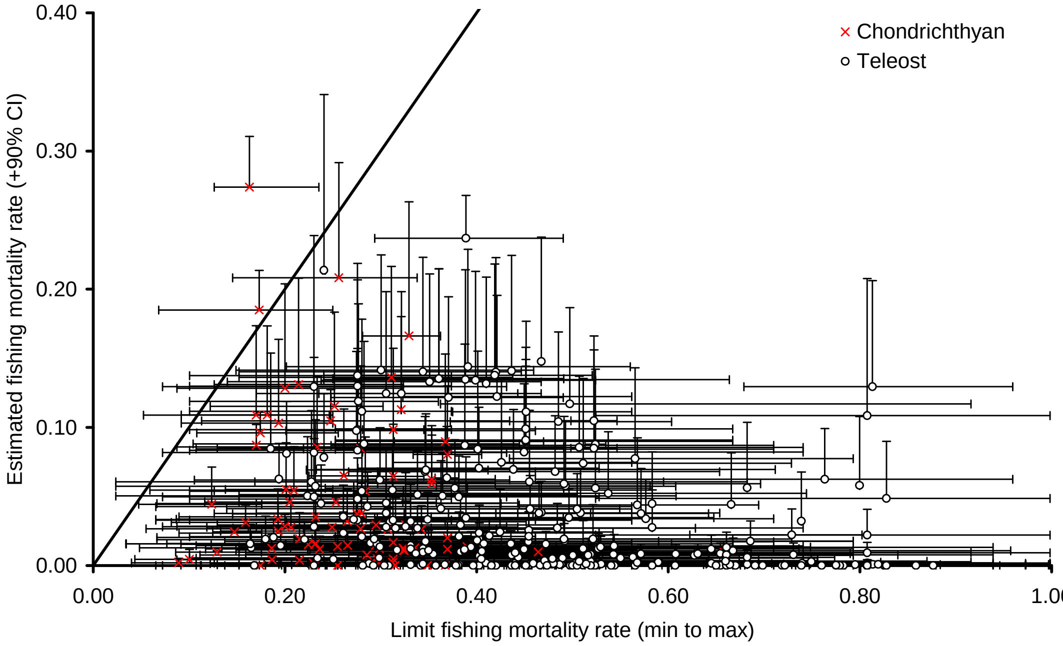


Figure 3-3. Comparison of estimated fishing mortality rate within the fishery jurisdiction and the limit fishing mortality rate for fish species caught in the SESSF otter trawl sub-fishery. The diagonal line is where $u = u_{lim}$.

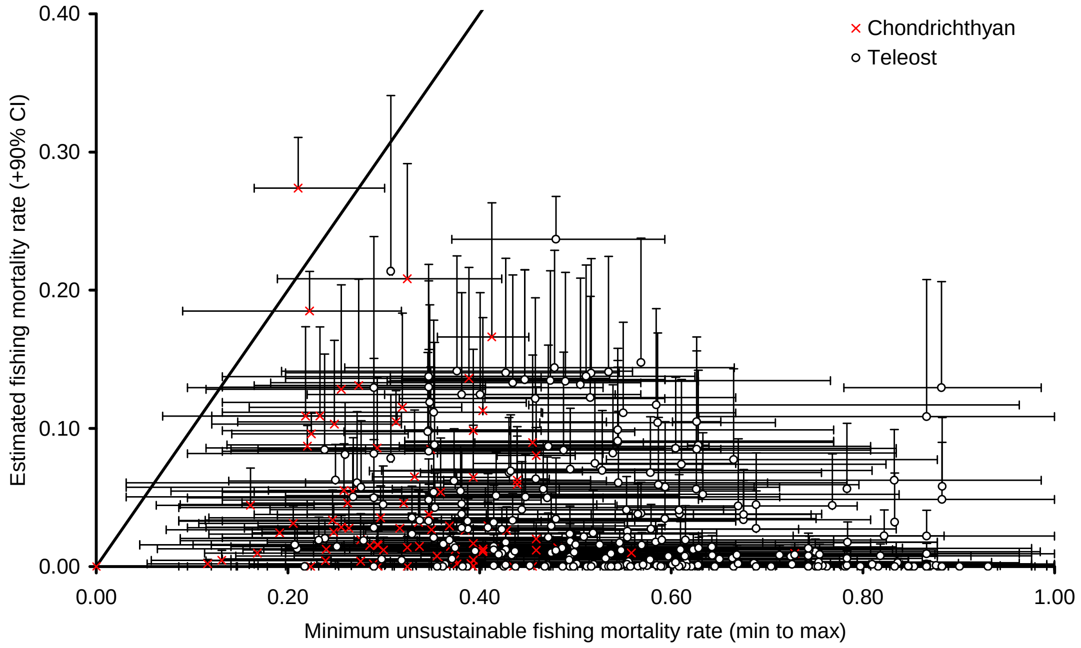


Figure 3-4. Comparison of estimated fishing mortality rate within the fishery jurisdiction and the minimum unsustainable fishing mortality rate for fish species caught in the SESSF otter trawl sub-fishery. The diagonal line is where $u = u_{crash}$.

3.2 Great Australian Bight Trawl Sub-fishery

We assessed 204 species of fish (52 chondrichthyans and 152 teleosts) that may be impacted by the GAB trawl fishery. Among these species 195 species (including all chondrichthyan) have spatial distribution information.

Estimated fishing mortality rate is low for this fishery, mainly due to low overlap between fishing effort and species distribution. No species is found to have fishing mortality (including uncertainty) greater than any reference point (either u_{msm}, u_{lim}, or u_{crash}, including minimum reference points) (Figures 3-5 and 3-6). However the spatial distribution of the fishery is changing, with more fishing on the upper slope in recent years, so this assessment should be updated if the fishery continues this trend.

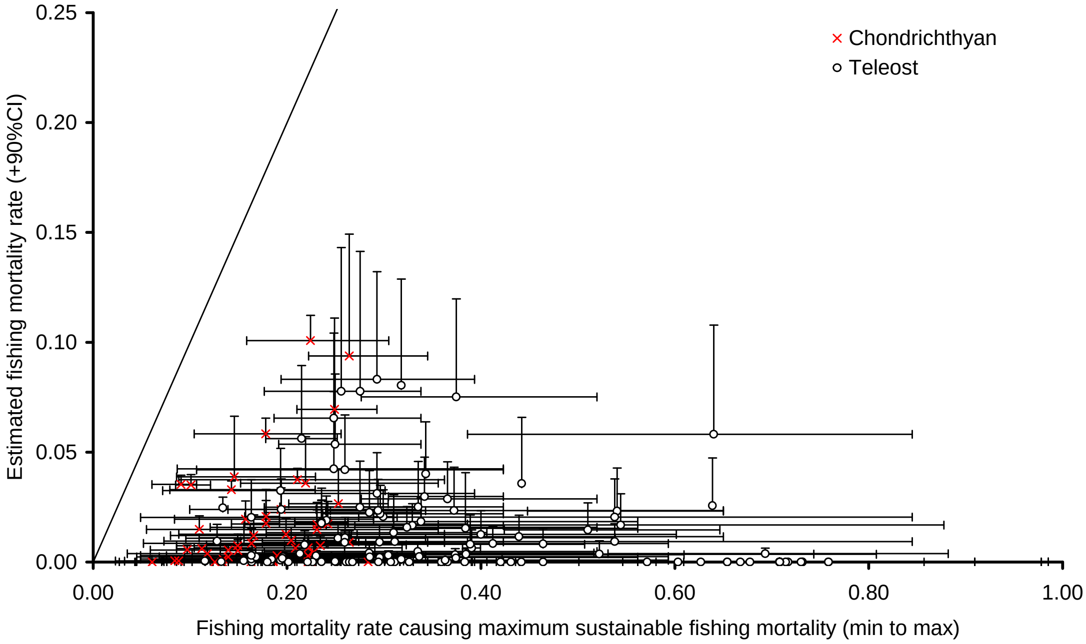


Figure 3-5. Comparison of estimated fishing mortality rate within the fishery jurisdiction and the fishing mortality rate corresponding to the maximum sustainable mortality for fish species caught in the SESSF GAB trawl sub-fishery. The diagonal line is where $u = u_{msm}$.

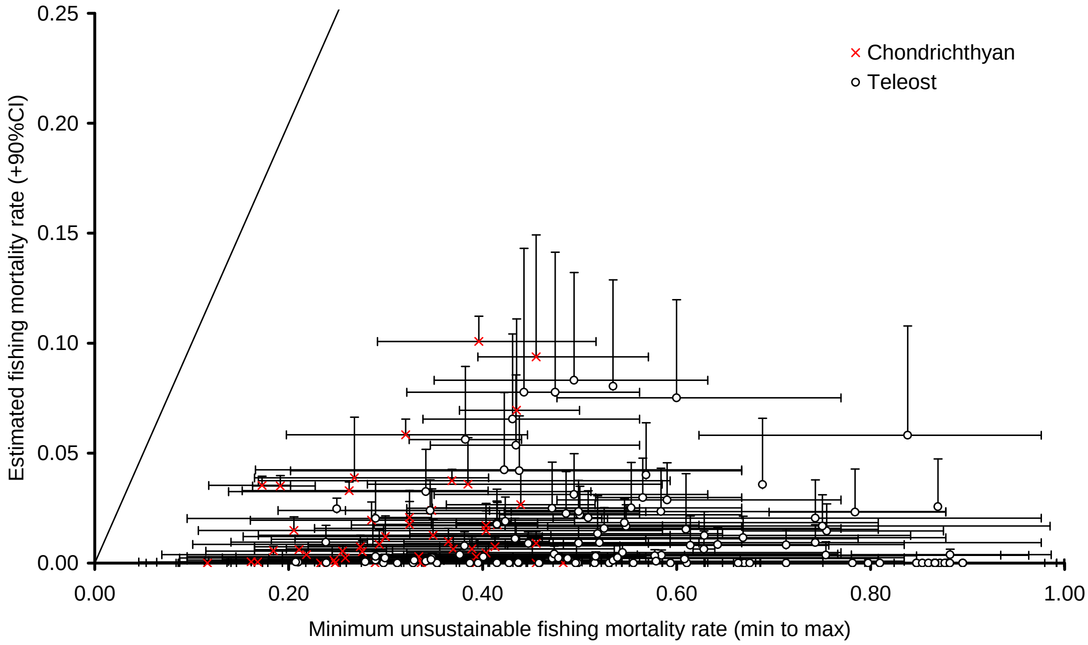


Figure 3-6. Comparison of estimated fishing mortality rate within the fishery jurisdiction and the minimum unsustainable fishing mortality rate for fish species incidentally caught in the GAB trawl fishery in SESSF. The diagonal line is where $u = u_{crash}$.

3.3 Shark Gillnet Sub-fishery

We assessed 195 species of fish (40 chondrichthyans and 155 teleosts) that may encounter shark gillnet. Among these species 177 species (including all chondrichthyans) have spatial distribution information.

The spatial resolution in the gillnet fishery is much lower than the trawl fisheries. Our assessment indicates that 6 species have estimated mean fishing mortality rate greater than mean u_{msm}: *Carcharhinus brachyurus*, *Carcharhinus obscurus*, *Carcharodon carcharias*, *Notorynchus cepedianus*, *Sphyrna zygaena*, and *Rhincodon typus* (Table 3-3, Figure 3-7). These are all chondrichthyan species. Among these species, experts believe *Sphyrna zygaena* (hammerhead) may be at risk only because the juveniles are demersal (adults are pelagic) so the early life history stage may be vulnerable. *Rhincodon typus* (whale shark) could be caught during migrating but this is a rare event. Three have mean fishing mortality rate greater than mean u_{lim} (Figure 3-8) and two species have mean fishing mortality rate greater than mean u_{crash} (Figure 3-9). If we include uncertainty in both estimated fishing mortality rate and the reference points, 18 species are at precautionary medium risk category (either $E[u] \geq \min[u_{msm}]$, or $E[u]$ + 90%CI $\geq E[u_{msm}]$), 13 species are at precautionary high risk category (either $E[u] \geq \min[u_{lim}]$, or $E[u]$ + 90%CI $\geq E[u_{lim}]$), and 9 species are at precautionary extreme high risk category (either $E[u] \geq \min[u_{crash}]$, or $E[u]$ + 90%CI $\geq E[u_{crash}]$) (Table 3-3).

As for the trawl fishery, Table 3-2 includes all precautionary risk species when uncertainty is taken into account. The estimated impact (especially the upper 90% CI) may have been overestimated or the reference points underestimated (especially the minimum value of a reference points) for some species. Species that are overridden by experienced biologists or are felt uncertain are listed on the lower part of Table 3-3. Specific comments for these species and a few others are as follows.

Rhincodon typus (whale shark): the overlap of fishing effort with species distribution may have been overestimated. Result should be verified by actual data.

Odontaspis ferox (sand tiger shark): Fishing impact, especially the overlap of effort with species distribution, may have been overestimated. This species is only found off New South Wales and the shark gillnet fishery is excluded from this area.

Dactylophora nigricans (dusky morwong): available to shallow set gear in temperate waters, *but a resident of heavy reef, so may not be vulnerable.*

Alopias vulpinus (Thintail Thresher Shark): mainly pelagic and encounterability with gillnet is low because they tend to be high in the water column.

Lamna nasus (Porbeagle shark): mainly pelagic and encounterability with gillnet is low because they tend to be high in the water column.

Isurus oxyrinchus (shortfinned mako): mainly pelagic and encounterability with gillnet is low because they tend to be high in the water column.

Prionace glauca (Blue Shark): mainly pelagic and encounterability with gillnet is low because they tend to be high in the water column.

Centrophorus harrissoni (Harrison's dogfish): found deeper than gillnets set in the shark fishery.

Orectolobus maculates (spotted wobbegong): tends to occur mostly on reef bottom whereas gillnets are set on sandy substrates. Also the highest densities of this species occur mostly off NSW where gillnetting is prohibited.

Carcharhinus obscurus (dusky shark): occurs mostly off Western Australia and much more likely to be affected by the Western Australian shark fishery rather than by the SESSF.

Table 3-3. Species at potential risk of overfishing in the SESSF **shark gillnet sub-fishery**. Species are sorted by [u - u_{msm}]. I_A = fraction of distribution area impacted; q = overall catchability, S = post-capture survival rate, Method = methods used for estimating the reference points. See text for risk category codes. Species without a risk code are in the precautionary medium risk (m) only.

					u		u_{msm}		u_{lim}		u_{crash}			
Scientific name	Common name	I_A	q	1-S	Mean	se	Mean	Min	Mean	Min	Mean	Min	Method	Risk
Carcharhinus brachyurus	Bronze Whaler	0.68	0.40	1.0	0.27	0.13	0.09	0.06	0.13	0.09	0.17	0.11	123456	EeHhM
Carcharhinus obscurus	Dusky Shark	0.50	0.40	1.0	0.20	0.09	0.10	0.06	0.14	0.09	0.18	0.12	123456	EeHhM
Carcharodon carcharias	white shark	0.63	0.26	1.0	0.17	0.10	0.11	0.05	0.16	0.08	0.21	0.11	123456	eHhM
Notorynchus cepedianus	Broadnose sevengill shark	1.00	0.26	1.0	0.26	0.15	0.21	0.10	0.30	0.14	0.37	0.19	123456	ehM
Sphyrna zygaena	smooth hammerhead	0.54	0.40	1.0	0.22	0.10	0.18	0.11	0.26	0.16	0.33	0.21	23456	ehM
Isurus oxyrinchus	Shortfinned Mako or Blue P	0.50	0.26	1.0	0.13	0.08	0.16	0.08	0.23	0.12	0.30	0.16	123456	h
Orectolobus maculatus	Spotted wobbegong	0.18	0.26	1.0	0.05	0.03	0.08	0.08	0.12	0.12	0.16	0.16	5	
Galeorhinus galeus	School Shark, Tope shark	0.27	0.40	1.0	0.11	0.06	0.15	0.09	0.21	0.14	0.27	0.18	123456	
Squatina australis	Australian Angel Shark	0.25	0.40	1.0	0.10	0.05	0.16	0.12	0.24	0.18	0.30	0.23	3456	
Pristiophorus cirratus	common saw shark	0.27	0.40	1.0	0.11	0.06	0.18	0.10	0.25	0.15	0.32	0.20	2456	
Furgaleus macki	Whiskery Shark	0.25	0.40	1.0	0.10	0.05	0.21	0.04	0.29	0.06	0.36	0.07	123456	eh
Species overriden by experts														
Rhincodon typus	whale shark	1.00	0.09	1.0	0.09	0.09	0.07	0.03	0.10	0.04	0.13	0.05	13456	ehM
Odontaspis ferox	sand tiger shark	1.00	0.13	1.0	0.13	0.11	0.19	0.10	0.27	0.15	0.34	0.19	23456	h
Dactylophora nigricans	Dusky Morwong	0.97	0.13	1.0	0.13	0.13	0.19	0.04	0.27	0.07	0.33	0.09	123456	eh
Alopias vulpinus	Thintail Thresher Shark, thr	0.57	0.26	1.0	0.15	0.09	0.17	0.05	0.24	0.08	0.30	0.10	23456	eh
Lamna nasus	Porbeagle shark	0.41	0.26	1.0	0.11	0.07	0.13	0.09	0.19	0.14	0.24	0.18	123456	h
Prionace glauca	Blue Shark	0.43	0.26	1.0	0.11	0.07	0.17	0.08	0.24	0.12	0.31	0.15	123456	
Centrophorus harrissoni	Harrison's dogfish	0.66	0.13	1.0	0.09	0.08	0.14	0.09	0.20	0.13	0.26	0.16	23456	h

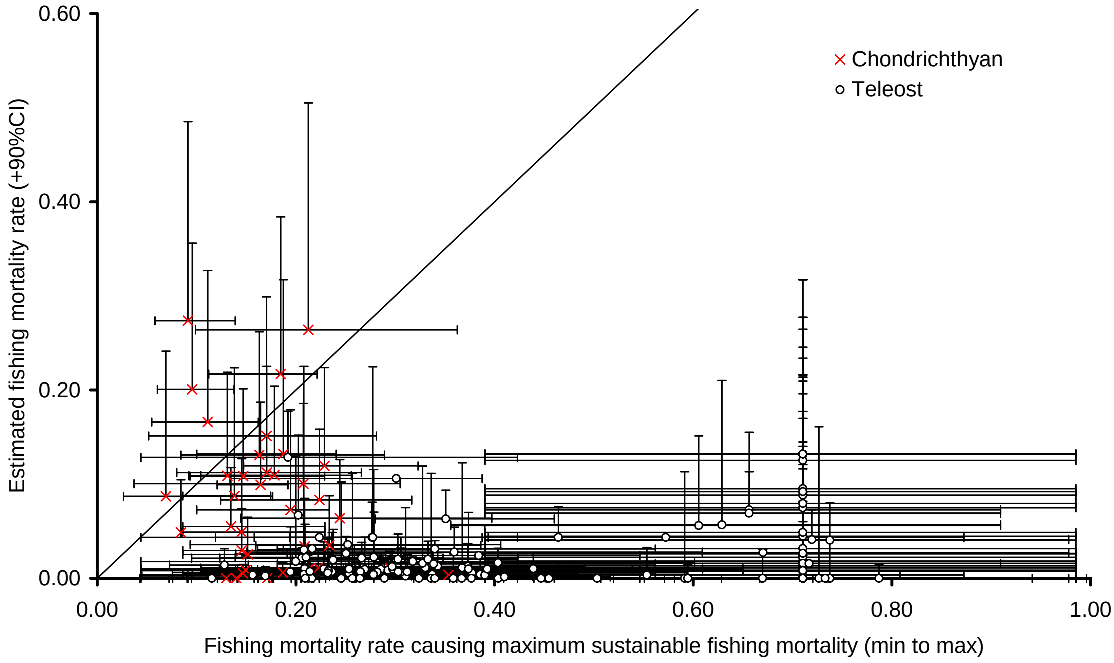


Figure 3-7. Comparison of estimated fishing mortality rate within the fishery jurisdiction and fishing mortality rate corresponding to the maximum sustainable fishing mortality for fish species caught in the SESSF shark gillnet sub-fishery. The diagonal line is where $u = u_{msm}$.

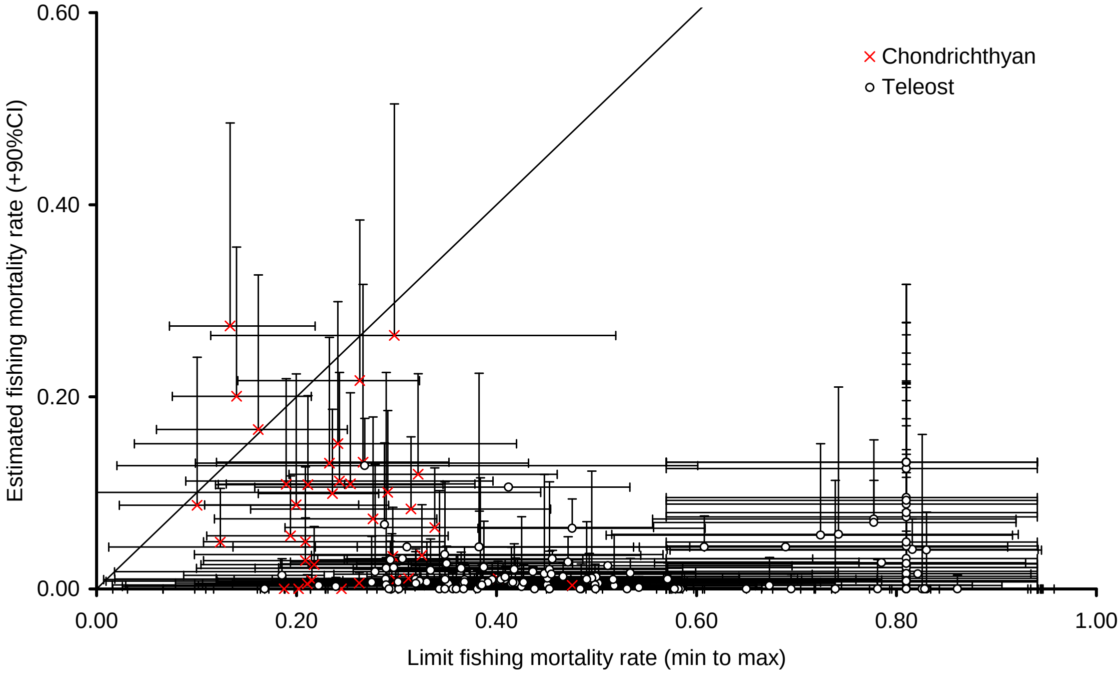


Figure 3-8. Comparison of estimated fishing mortality rate within the fishery jurisdiction and the limit fishing mortality rate for fish species caught in the SESSF shark gillnet sub-fishery. The diagonal line is where $u = u_{lim}$.

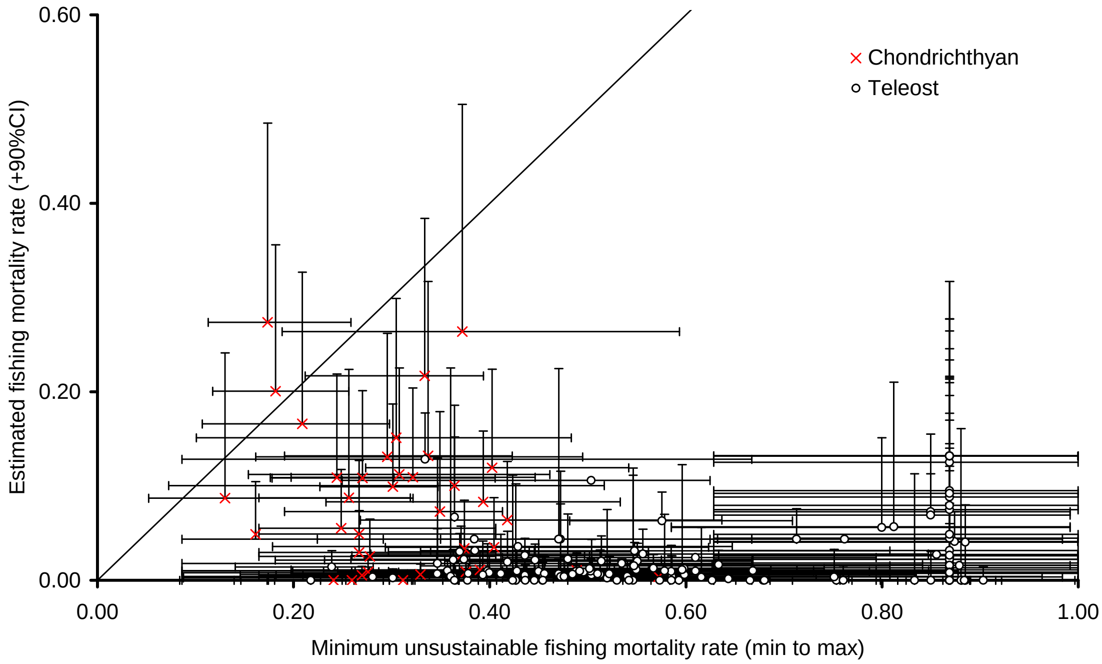


Figure 3-9. Comparison of estimated fishing mortality rate within the fishery jurisdiction and the minimum unsustainable fishing mortality rate for fish species incidentally caught in the SESSF shark gillnet sub-fishery. The diagonal line is where $u = u_{crash}$.

3.4 Danish Seine Sub-fishery

We assessed 71 species of fish (3 chondrichthyans and 68 teleosts) caught in the Danish seine fishery. Among these species 64 species have spatial distribution information.

Fishing efforts and affected area in the seine fishery are relatively small compared with other sub-fisheries. As a result, the estimated fishing mortality rate is low for this fishery. No species is found to have fishing mortality rate (including uncertainty) greater than the minimum u_{msm}, u_{lim} or u_{crash} (Figure 3-10, Figure 3-11).

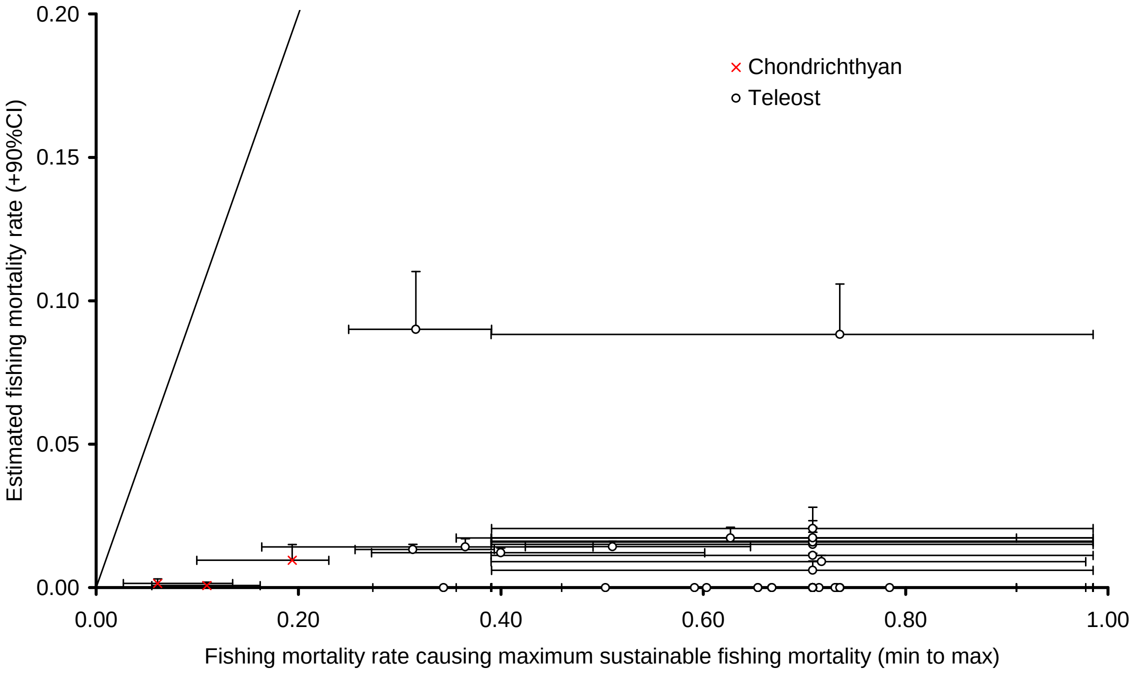


Figure 3-10. Comparison of estimated fishing mortality rate within the fishery jurisdiction and the fishing mortality rate corresponding to the maximum sustainable fishing mortality for fish species caught in the Danish seine fishery in SESSF. The diagonal line is where $u = u_{msm}$.

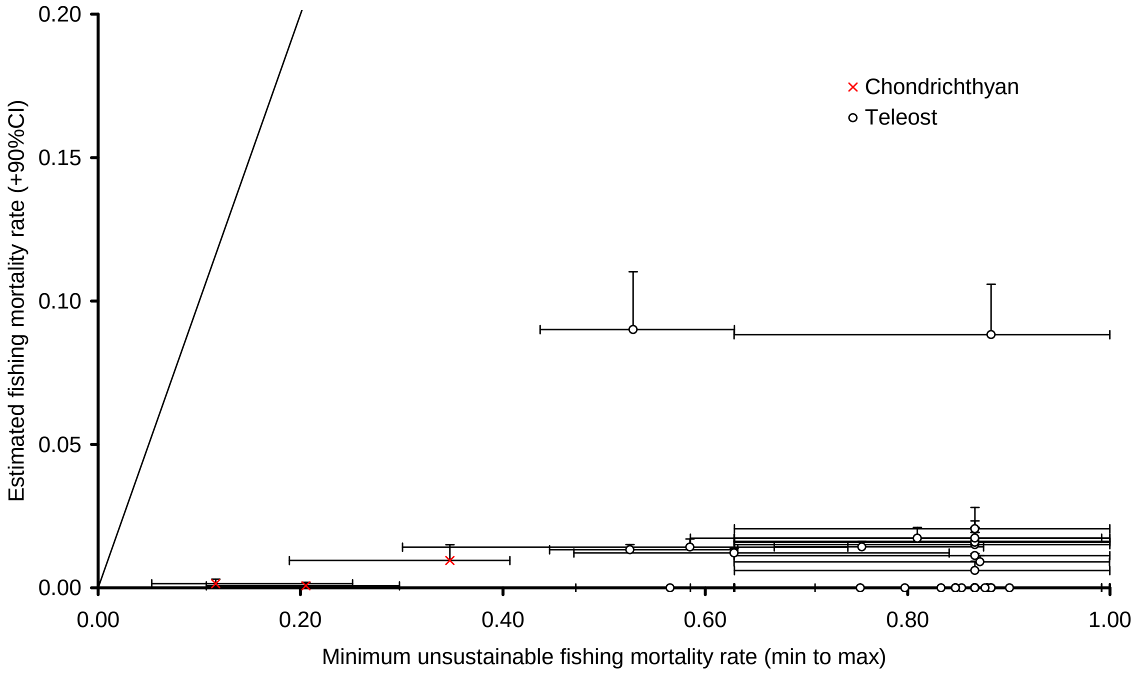


Figure 3-11. Comparison of estimated fishing mortality rate within the fishery jurisdiction and the minimum unsustainable fishing mortality rate for fish species caught in the Danish seine fishery in SESSF. The diagonal line is where $u = u_{crash}$.

3.5 Automatic Longline Sub-fishery

In this sub-fishery gear is restricted to waters deeper than 183 m and the target species (ling and blue eye trevalla) are typically targeted in 300-600 m (Daley et al. 2007). We limited effort distribution within the 200 to 700 m depth range. We assessed 160 species of fish (39 chondrichthyans and 121 teleosts). Among these species 146 have spatial distribution information (i.e. overlapping with fishery or outside the boundary).

The assessment result indicates that 5 species have estimated mean fishing mortality rate greater than mean u_{msm}: (*Dipturus gudgeri*, *Centrophorus harrissoni*, *Centrophorus uyato*, *Deania quadrispinosa*, and *Polyprion oxygeneios*) (Table 3-4., Figure 3-12). Four of which are chondrichthyans and one is a teleost. No species have mean fishing mortality rate greater than mean u_{lim} (Figure 3-13) or mean u_{crash} (Figure 3-14). If we include uncertainty in both estimated fishing mortality rate and the reference points, 16 species are at precautionary medium risk category (either $E[u] \geq \min[u_{msm}]$, or $E[u]$ + 90%CI $\geq E[u_{msm}]$), 10 species are at precautionary high risk category (either $E[u] \geq \min[u_{lim}]$, or $E[u]$ + 90%CI $\geq E[u_{lim}]$), and 3 species are at precautionary extreme high risk category (either $E[u] \geq \min[u_{crash}]$, or $E[u]$ + 90%CI $\geq E[u_{crash}]$) (Table 3-4.).

Biologists have commented and overridden three species that are estimated to be above the precautionary medium risk in this sub-fishery (lower part of Table 3-4):

Cirrhigaleus barbifer (mandarin shark): not a common species.

Caelorinchus fasciatus (banded whiptail): a benthic feeder with a small ventral mouth so catchability may have been overestimated.

Lepidorhynchus denticulatus (toothed whiptail): a very abundant true benthopelagic species that spends a lot of time in mid-water. Catchability may be too high or the species requires a closer look at actual catch as most individuals are rather small in relation to the hooks and bait used by at least the temperate fishery.

Mora moro (ribaldo): not in the table but is expected to be at risk. Our result shows this species has u = 0.1 (se = 0.01) while u_{msm} = 0.34 (min = 0.2). Clearly it is not at the precautionary medium risk.

Table 3-4. Species at potential risk of overfishing in the SESSF **auto longline sub-fishery**. Species are sorted by [u - u_{msm}]. I_A = fraction of distribution area impacted; q = overall catchability, S = post-capture survival rate, Method = methods used for estimating the reference points. See text for risk category codes. Species without a risk code are in the precautionary medium risk (m) only.

					u		u_{msm}		u_{lim}		u_{crash}			
Scientific name	Common name	I_A	q	1-S	Mean	se	Mean	Min	Mean	Min	Mean	Min	Method	Risk
Dipturus gudgeri	Bight skate	0.96	1.0	1.0	0.15	0.02	0.11	0.09	0.16	0.13	0.21	0.16	23456	hM
Centrophorus harrissoni	Harrison's dogfish	0.98	1.0	1.0	0.15	0.02	0.13	0.06	0.18	0.09	0.23	0.12	23456	ehM
Centrophorus uyato (east)	southern dogfish	0.98	1.0	1.0	0.15	0.02	0.13	0.06	0.19	0.09	0.25	0.12	23456	ehM
Deania quadrispinosa	platypus Shark	0.84	1.0	1.0	0.13	0.01	0.12	0.04	0.17	0.05	0.22	0.07	23456	ehM
Polyprion oxygeneios	hapuku	0.85	1.0	1.0	0.13	0.01	0.13	0.07	0.18	0.11	0.24	0.14	23456	hM
Squalus mitsukurii	greeneye dogfish	0.87	1.0	1.0	0.14	0.02	0.15	0.09	0.21	0.13	0.27	0.16	123456	h
Dipturus sp. B	grey skate	0.92	1.0	1.0	0.14	0.02	0.18	0.08	0.25	0.12	0.32	0.16	23456	h
Genypterus blacodes	ling	0.59	1.0	1.0	0.14	0.02	0.19	0.14	0.27	0.20	0.35	0.26	123456	
Hyperoglyphe antarctica	blueeye trevalla	0.97	1.0	1.0	0.15	0.02	0.21	0.10	0.30	0.15	0.38	0.19	123456	h
Dalatias licha	black Shark	0.45	1.0	1.0	0.07	0.01	0.14	0.06	0.20	0.09	0.26	0.11	23456	
Cephaloscyllium sp. A [in Las	whitefin swellshark	0.93	1.0	1.0	0.15	0.02	0.22	0.10	0.31	0.15	0.39	0.20	23456	
Etmopterus lucifer	blackbelly lantern shark	0.57	1.0	1.0	0.09	0.01	0.17	0.06	0.23	0.09	0.29	0.11	23456	h
Helicolenus barathri	ocean Perch - Offshore	0.94	0.7	1.0	0.10	0.03	0.20	0.07	0.28	0.10	0.35	0.14	23456	
Species overriden by experts														
Cirrhigaleus barbifer	Mandarin Shark	0.62	1.0	1.0	0.10	0.01	0.14	0.06	0.20	0.09	0.26	0.11	23456	h
Caelorinchus fasciatus	banded whiptail	0.95	0.7	1.0	0.10	0.03	0.20	0.07	0.28	0.10	0.35	0.13	23456	
Lepidorhynchus denticulatus	toothed Whiptail	0.93	0.7	1.0	0.10	0.03	0.20	0.07	0.28	0.10	0.35	0.13	23456	

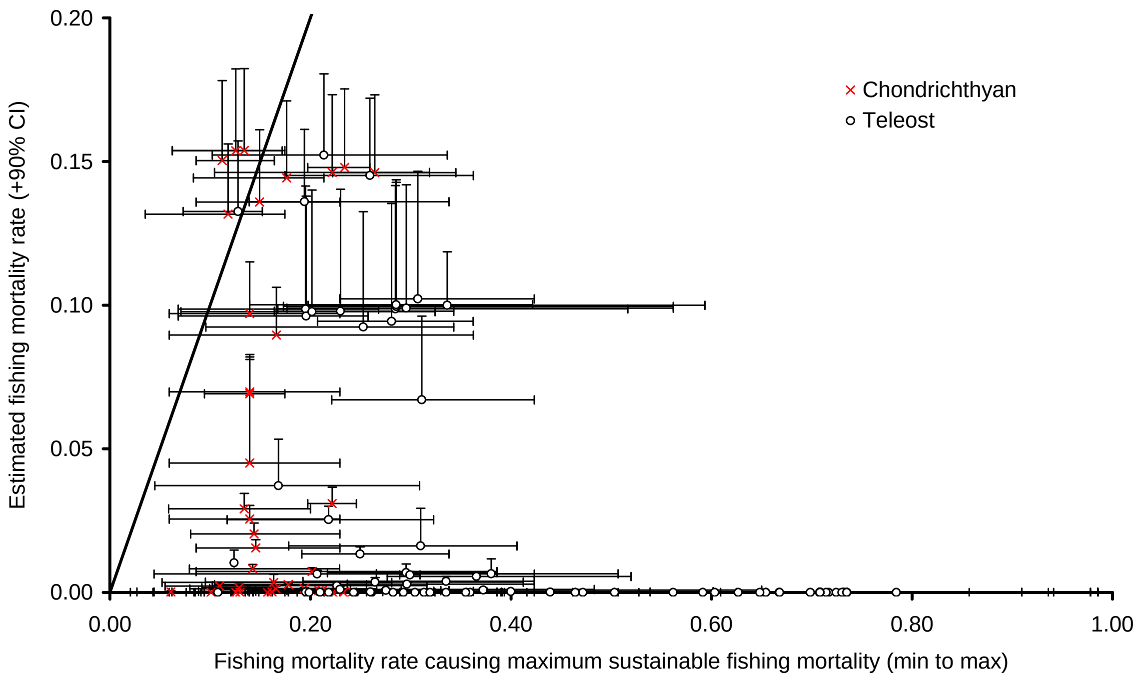


Figure 3-12. Comparison of estimated fishing mortality rate within the fishery jurisdiction and the fishing mortality rate at the maximum sustainable fishing mortality level for fish species caught in the Auto Longline fishery in SESSF. The diagonal line is where $u = u_{msm}$.

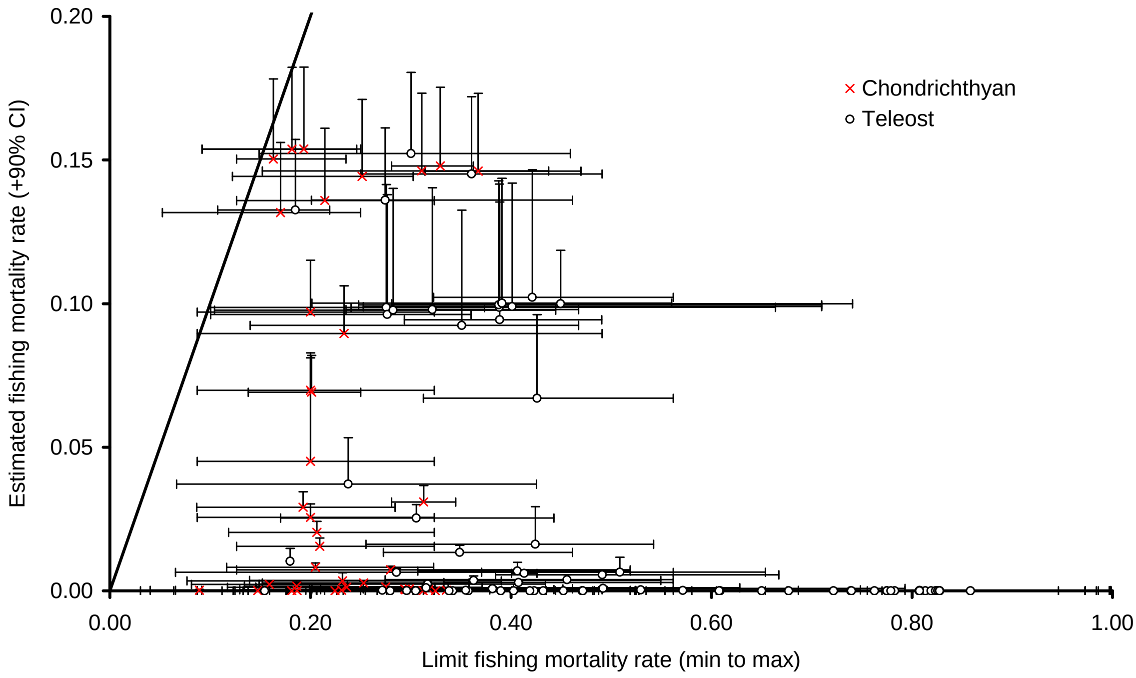


Figure 3-13. Comparison of estimated fishing mortality rate within the fishery jurisdiction and the limit fishing mortality rate for fish species caught in the Auto Longline fishery in SESSF. The diagonal line is where $u = u_{lim}$.

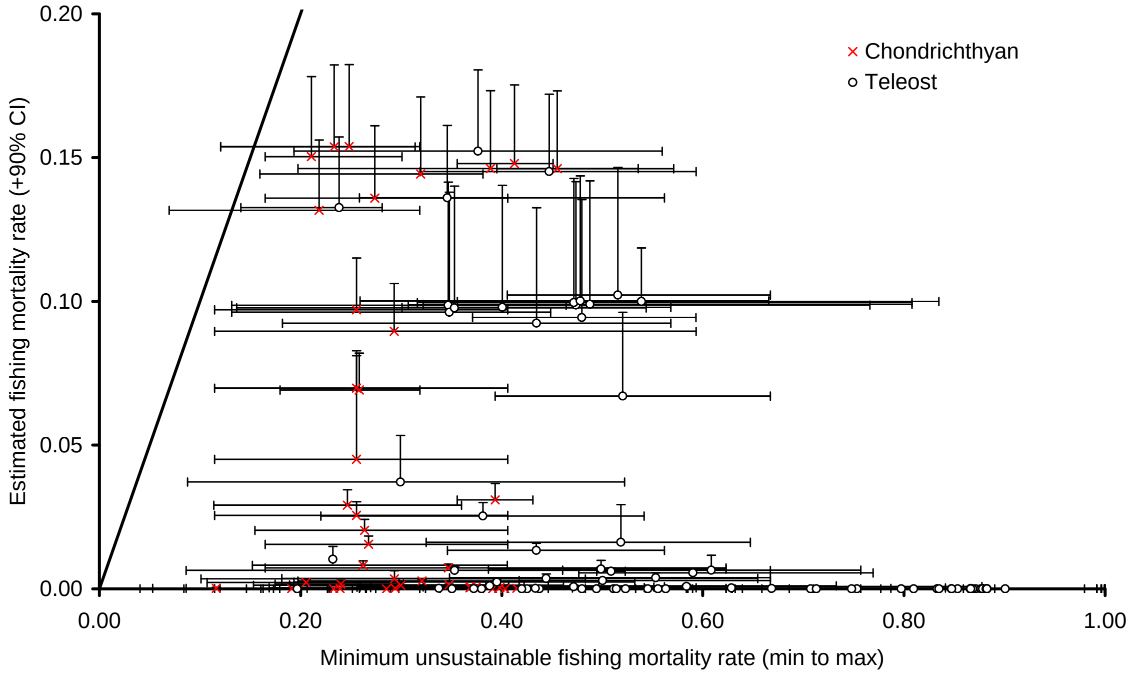


Figure 3-14. Comparison of estimated fishing mortality rate within the fishery jurisdiction and the minimum unsustainable fishing mortality rate for fish species caught in the Auto Longline fishery in SESSF. The diagonal line is where $u = u_{crash}$.

3.6 Cumulative impacts from sub-fisheries

We assessed a total of 499 fish species in five sub-fisheries in SESSF, among which 99 are chondrichthyans and 400 are teleosts. All chondrichthyans have spatial distribution information while 45 teleosts do not have distribution information.

The assessment result shows that 24 chondrichthyans and 9 teleosts have estimated mean cumulative fishing mortality rate greater than mean u_{msm} (**Error! Reference source not found.**, Table 3-6, Figure 3-15), 13 chondrichthyans and 2 teleosts have estimated mean cumulative fishing mortality rates greater than mean u_{lim} (Figure 3-16), and 9 chondrichthyans and 1 teleosts have estimated mean cumulative fishing mortality rates greater than mean u_{crash} (Figure 3-17). If we include uncertainty in both estimated fishing mortality rate and the reference points, 39 chondrichthyans and 33 teleosts are at precautionary medium risk category (either $E[u] \geq \min[u_{msm}]$, or $E[u] + 90\%CI \geq E[u_{msm}]$), 27 chondrichthyans and 23 teleosts species are at precautionary high risk category (either E[u] ≥ min[u_{lim}], or E[u] + 90%CI ≥ E[u_{lim}]), and 23 chondrichthyans and 15 teleosts are at precautionary extreme high risk category (either E[u] ≥ min[u_{crash}], or E[u] + 90%CI ≥ E[u_{crash}]) (**Error! Reference source not found.**, Table 3-6).

Biologists have commented and overridden four species in Table 3-5 and Table 3-6:

Etmopterus lucifer (blackbelly lantern shark): an uncommon benthopelagic species in very deep water. More data are needed to verify the result.

Centrolophus niger (rudderfish): a deepwater pelagic species so may be not at potential risk.

Lepidopus caudatus (southern frostfish): an abundant benthopelgic species spending much time in the water column so may be not at potential risk.

Beryx decadactylus (imperador) and *B. splendens* (oxeye oreo): rarely seen fish in temperate waters so may be not at potential risk.

The results of cumulative analysis (Table 3-5 and Table 3-6) have not taken into account the over-rides listed for the individual sub-fisheries, as a species may be overridden by experts in one sub-fishery but that species may be at risk under cumulative impacts.

Several other species (thintail thresher shark, Porbeagle shark, and blue shark) are also impacted by the Eastern Tuna and Billfish Fishery. These species may have greater ecological risk from fishing.

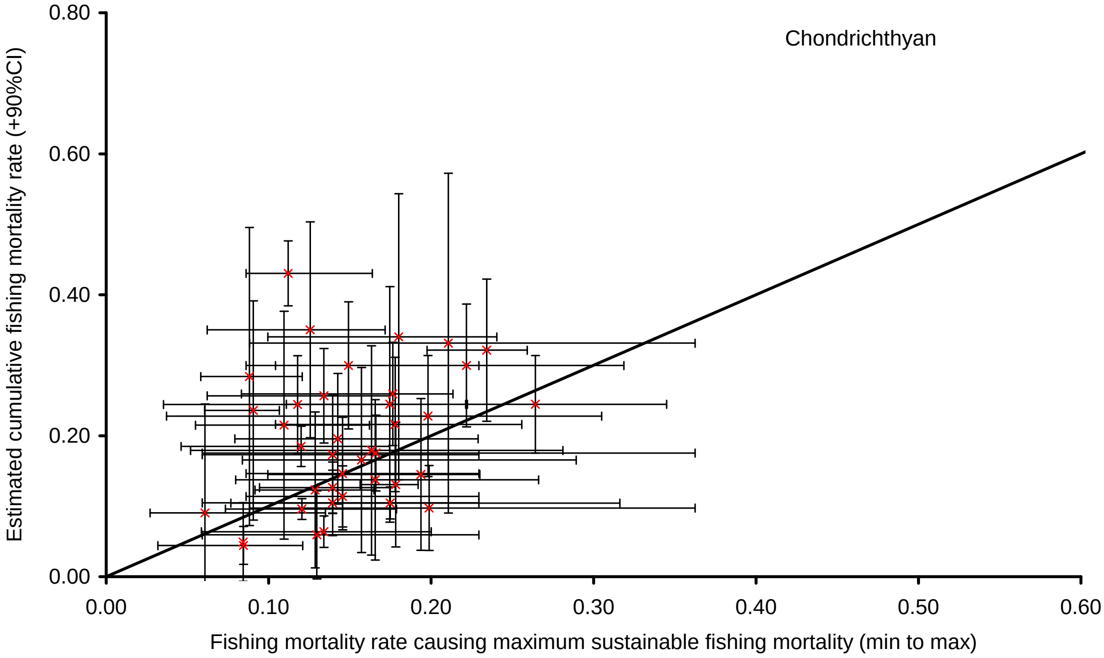
On the opposite side of precautionary consideration, we examined the species that are at confident risk, i.e., species whose lower 90% confidence limit of the estimated cumulative fishing mortality rate is greater than the mean value of a reference point, or species whose mean cumulative fishing mortality rate is greater than the maximum value of a reference point. We found that 7 chondrichthyans and 1 teleosts are at confident medium risk category (either E[u] ≥ max[u_{msm}], or E[u] - 90%CI ≥ E[u_{msm}], Risk category 1) (**Error! Reference source not found.**, Table 3-6, Figure 3-15), 2 chondrichthyans are at confident high risk category (either E[u] ≥ max[u_{lim}], or E[u] - 90%CI ≥ E[u_{lim}], Risk category 2) (Figure 3-16), and 1 chondrichthyan is at confident extreme high risk category (either E[u] ≥ max[u_{crash}], or E[u] - 90%CI ≥ E[u_{crash}], Risk category 3) (Figure 3-17).

Table 3-5. Chondrichthyan species at potential risk of overfishing by cumulative impacts from five sub-fisheries in the SESSF. Species are sorted by [u - u_{msm}]. See text for risk category codes. Species without a risk code are in the precautionary medium risk (m) only.

		Estimated fishing mortality rate u							u_{msm}		u_{lim}		u_{crash}		
Scientific name	Common name	Trawl	GAB	Gillnet	Seine	Longline	**Cum**	**SE**	Mean	Min	Mean	Min	Mean	Min	Risk
Dipturus gudgeri	bight skate	0.27	0.01			0.15	0.43	0.03	0.11	0.09	0.16	0.13	0.21	0.16	EeHhM123
Centrophorus harrissoni	Harrison's dogfish	0.11		0.09		0.15	0.35	0.09	0.13	0.06	0.18	0.09	0.23	0.12	EeHhM12
Carcharhinus brachyurus	Bronze Whaler	0.01	0.00	0.27			0.28	0.13	0.09	0.06	0.13	0.09	0.17	0.11	EeHhM
Odontaspis ferox	sand tiger shark	0.21		0.13			0.34	0.12	0.18	0.10	0.26	0.15	0.32	0.19	EeHhM
Squalus mitsukurii	Green-Eyed Dogfish	0.13	0.01	0.03		0.14	0.30	0.05	0.15	0.09	0.21	0.13	0.27	0.16	EeHhM1
Carcharhinus obscurus	Dusky Shark		0.04	0.20			0.24	0.09	0.09	0.06	0.13	0.09	0.17	0.12	EeHhM
Deania quadrispinosa	Platypus Shark	0.11	0.00			0.13	0.24	0.04	0.12	0.04	0.17	0.05	0.22	0.07	EeHhM1
Centrophorus uyato (east)	southern dogfish	0.10				0.15	0.26	0.04	0.13	0.06	0.19	0.09	0.25	0.12	EeHhM1
Notorynchus cepedianus	Broadnose sevengill shark	0.03	0.04	0.26		0.00	0.33	0.15	0.21	0.09	0.29	0.13	0.37	0.17	eHhM
Carcharodon carcharias	white shark	0.03	0.01	0.17	0.00	0.00	0.22	0.10	0.11	0.05	0.16	0.08	0.21	0.11	EeHhM
Hydrolagus lemures	bight ghost shark	0.17	0.01			0.15	0.32	0.06	0.23	0.20	0.33	0.28	0.41	0.36	ehM
Dipturus sp. B	grey skate	0.12				0.14	0.26	0.04	0.18	0.08	0.25	0.12	0.32	0.16	eHhM1
Cephaloscyllium sp. A [in L	Whitefin Swell Shark	0.14	0.01	0.01		0.15	0.30	0.05	0.22	0.10	0.31	0.15	0.39	0.20	ehM
Sphyrna zygaena	smooth hammerhead	0.03		0.22			0.24	0.10	0.17	0.11	0.25	0.16	0.32	0.21	ehM
Centrophorus squamosus	nilson's deepsea dogfish	0.18					0.18	0.02	0.12	0.05	0.17	0.07	0.22	0.09	eHhM1
Galeorhinus galeus	School Shark, Tope shark	0.05	0.03	0.11		0.01	0.20	0.06	0.14	0.08	0.20	0.12	0.26	0.15	ehM
Pristiophorus cirratus	common saw shark	0.05	0.06	0.11		0.00	0.22	0.06	0.18	0.10	0.25	0.15	0.32	0.20	ehM
Isistius brasiliensis	cookie-cutter shark (cigar sh	0.13				0.05	0.17	0.05	0.14	0.06	0.20	0.09	0.26	0.11	ehM
Rhincodon typus	whale shark	0.00	0.00	0.09	0.00	0.00	0.09	0.09	0.06	0.03	0.09	0.04	0.12	0.05	eHhM
Furgaleus macki	Whiskery Shark	0.03	0.10	0.10			0.23	0.05	0.20	0.04	0.28	0.06	0.35	0.07	ehM
Alopias vulpinus	Thintail Thresher Shark, thre	0.02	0.01	0.15		0.00	0.18	0.09	0.16	0.05	0.23	0.08	0.29	0.10	ehM
Isurus oxyrinchus	Shortfinned Mako or Blue Po	0.02	0.02	0.13		0.00	0.17	0.08	0.16	0.08	0.22	0.12	0.29	0.16	ehM
Cirrhigaleus barbifer	Mandarin Shark			0.05		0.10	0.15	0.05	0.15	0.09	0.21	0.13	0.27	0.16	hM
Lamna nasus	Porbeagle shark	0.01		0.11		0.00	0.12	0.07	0.13	0.09	0.19	0.13	0.24	0.17	h
Deania calcea	Brier Shark	0.06	0.00			0.07	0.13	0.02	0.14	0.09	0.20	0.14	0.26	0.18	
Galeus boardmani	sawtail shark	0.09	0.01			0.15	0.24	0.04	0.26	0.22	0.37	0.31	0.46	0.40	
Centroscymnus plunketi	plunket's shark	0.10					0.10	0.01	0.12	0.07	0.17	0.11	0.22	0.14	
Prionace glauca	Blue Shark	0.01	0.01	0.11		0.00	0.14	0.07	0.17	0.08	0.24	0.12	0.30	0.15	h
Squalus megalops	Piked Dogfish	0.05	0.04	0.01		0.02	0.11	0.03	0.15	0.09	0.21	0.13	0.27	0.16	
Dalatias licha	Black Shark		0.01	0.03		0.07	0.10	0.03	0.14	0.06	0.20	0.09	0.26	0.11	h
Orectolobus maculatus	Spotted wobbegong		0.00	0.05			0.05	0.03	0.08	0.08	0.12	0.12	0.16	0.16	
Dipturus australis	common skate	0.04					0.04	0.02	0.08	0.03	0.12	0.05	0.16	0.06	
Squatina australis	Australian Angel Shark	0.01	0.02	0.10			0.13	0.05	0.18	0.16	0.25	0.23	0.32	0.29	
Carcharias taurus	grey nurse shark	0.04	0.02	0.07	0.01	0.00	0.15	0.07	0.19	0.10	0.28	0.15	0.35	0.19	
Centroscymnus owstoni	owston's dogfish	0.03	0.00			0.03	0.06	0.01	0.13	0.06	0.19	0.09	0.25	0.11	
Galeocerdo cuvier	Tiger Shark	0.10					0.10	0.01	0.17	0.08	0.25	0.11	0.31	0.15	
Squalus acanthias	white-spotted dogfish	0.00		0.06		0.00	0.06	0.04	0.13	0.06	0.19	0.09	0.24	0.11	
Heptranchias perlo	sharpnose seven-gill shark	0.08	0.01				0.10	0.04	0.20	0.09	0.28	0.13	0.35	0.17	
Species overriden by experts															
Etmopterus lucifer	Blackbelly Lantern Shark)	0.09				0.09	0.18	0.03	0.17	0.06	0.23	0.09	0.29	0.11	ehM

Table 3-6. Teleost species at potential risk of overfishing by cumulative impacts from five sub-fisheries in the SESSF. Species are sorted by [u - u_{msm}]. See text for risk category codes. Species without a risk code are in the precautionary medium risk (m) only.

		Estimated fishing mortality rate u							u_{msm}		u_{lim}		u_{crash}		
Scientific name	Common name	Trawl	GAB	Gillnet	Seine	Longline	**Cum**	**SE**	Mean	Min	Mean	Min	Mean	Min	Risk
Polyprion oxygeneios	Hapuku	0.08	0.01	0.01		0.13	0.24	0.05	0.13	0.07	0.18	0.11	0.24	0.14	EeHhM1
Hyperoglyphe antarctica	Blue Eye Trevalla	0.14	0.00	0.01		0.15	0.30	0.06	0.21	0.10	0.30	0.15	0.38	0.19	eHhM
Genypterus blacodes	Ling	0.10	0.02	0.01		0.14	0.26	0.04	0.19	0.14	0.27	0.20	0.35	0.26	ehM
Eptatretus longipinnis	hagfish	0.21					0.21	0.08	0.17	0.17	0.24	0.24	0.31	0.31	ehM
Rexea solandri	Gemfish -east	0.14	0.01	0.01		0.15	0.30	0.05	0.26	0.17	0.36	0.25	0.45	0.32	hM
Caelorinchus fasciatus	banded whiptail	0.13				0.10	0.23	0.05	0.20	0.07	0.28	0.10	0.35	0.13	ehM
Lepidorhynchus denticulatu	Toothed Whiptail	0.12	0.00			0.10	0.22	0.05	0.20	0.07	0.28	0.10	0.35	0.13	ehM
Macruronus novaezelandia	Blue Grenadier	0.12	0.00	0.01		0.10	0.23	0.05	0.23	0.16	0.32	0.24	0.40	0.30	hM
Zenopsis nebulosus	Mirror Dory	0.13	0.01	0.01		0.09	0.24	0.05	0.25	0.10	0.35	0.14	0.43	0.18	eh
Helicolenus barathri	Ocean Perch - Offshore	0.09	0.00	0.00		0.10	0.19	0.05	0.20	0.07	0.28	0.10	0.35	0.14	eh
Hoplostethus latus	giant sawbelly	0.13					0.13	0.07	0.16	0.05	0.23	0.07	0.29	0.10	eh
Caelorinchus australis	southern whiptail	0.13	0.00			0.10	0.24	0.06	0.28	0.17	0.39	0.24	0.47	0.31	
Dactylophora nigricans	Dusky Morwong		0.00	0.13			0.13	0.13	0.18	0.04	0.25	0.07	0.31	0.09	eh
Caelorinchus mirus	gargoyle fish	0.14					0.14	0.05	0.20	0.07	0.28	0.10	0.35	0.13	eh
Seriolella caerulea	White Trevalla	0.12	0.00	0.01		0.10	0.23	0.05	0.31	0.23	0.42	0.32	0.52	0.41	
Hoplostethus intermedius	common sawbelly	0.08	0.00				0.08	0.04	0.16	0.05	0.23	0.07	0.29	0.10	h
Neocyttus rhomboidalis	Spiky Oreo	0.04	0.00	0.00		0.04	0.09	0.02	0.17	0.04	0.24	0.07	0.30	0.09	h
Malacocephalus laevis	smooth whiptail	0.11					0.11	0.04	0.20	0.07	0.28	0.10	0.35	0.13	h
Bassanago bulbiceps	swollen-headed conger eel	0.12					0.12	0.04	0.22	0.12	0.31	0.17	0.38	0.22	
Caelorinchus kaiyomaru	whiptail	0.10					0.10	0.04	0.20	0.07	0.28	0.10	0.35	0.13	
Beryx splendens	Alfonsino	0.08	0.00	0.01		0.10	0.19	0.05	0.30	0.18	0.40	0.25	0.49	0.32	
Nemadactylus macropteru	Jackass Morwong	0.04	0.04	0.01		0.01	0.10	0.02	0.21	0.04	0.29	0.07	0.35	0.09	eh
Azygopus pinnifasciatus	righteye flounder	0.14					0.14	0.05	0.25	0.10	0.34	0.15	0.43	0.20	
Caelorinchus innotabilis	notable whiptail	0.08	0.00				0.09	0.03	0.20	0.07	0.28	0.10	0.35	0.13	
Epigonus lenimen	big-eyed cardinalfish	0.06					0.06	0.03	0.17	0.02	0.23	0.02	0.27	0.03	eh
Gephyroberyx darwinii	darwin's roughy	0.05	0.00				0.05	0.03	0.16	0.05	0.23	0.07	0.29	0.10	
Nemadactylus valencienne	queen snapper	0.03	0.04	0.00		0.00	0.08	0.03	0.20	0.04	0.27	0.07	0.34	0.09	h
Epigonus robustus	robust cardinalfish	0.05					0.05	0.03	0.17	0.02	0.22	0.02	0.27	0.03	eh
Epigonus denticulatus	white cardinalfish	0.06					0.06	0.03	0.17	0.02	0.23	0.02	0.28	0.03	eh
Oreosoma atlanticum	Oxeye Oreo	0.05					0.05	0.02	0.23	0.04	0.31	0.06	0.38	0.08	
Species overriden by experts															
Centrolophus niger	Rudderfish	0.24		0.00		0.09	0.34	0.03	0.28	0.21	0.39	0.29	0.48	0.37	hM
Lepidopus caudatus	Southern Frostfish	0.14	0.00	0.01		0.10	0.26	0.06	0.29	0.14	0.39	0.20	0.48	0.26	h
Beryx decadactylus	Imperador	0.09	0.00	0.01		0.10	0.20	0.05	0.29	0.17	0.39	0.25	0.47	0.32	



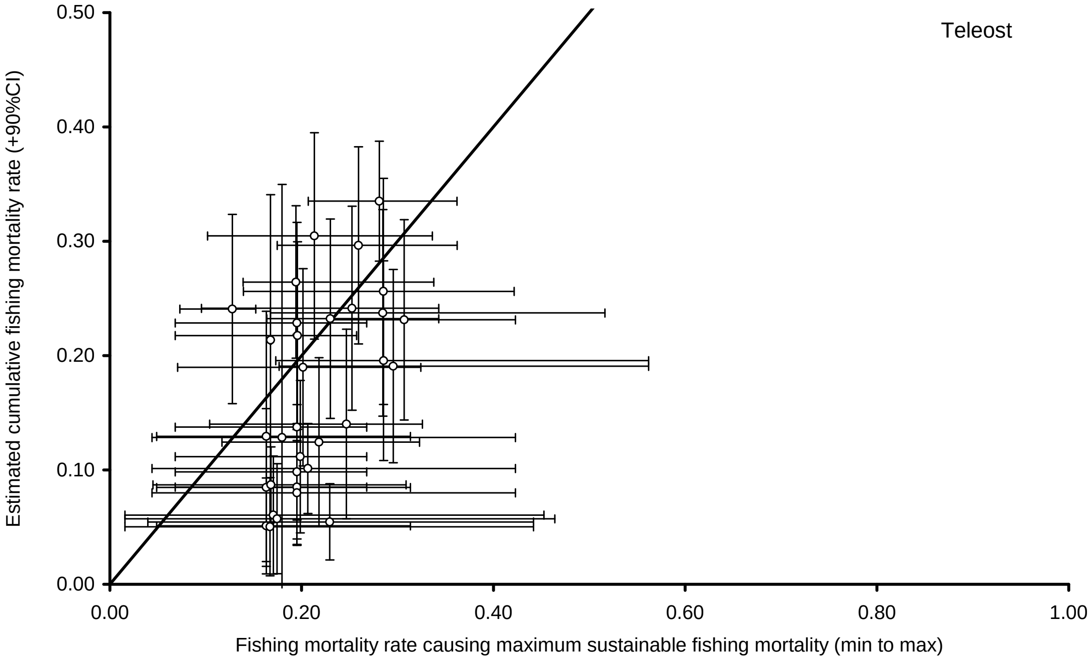
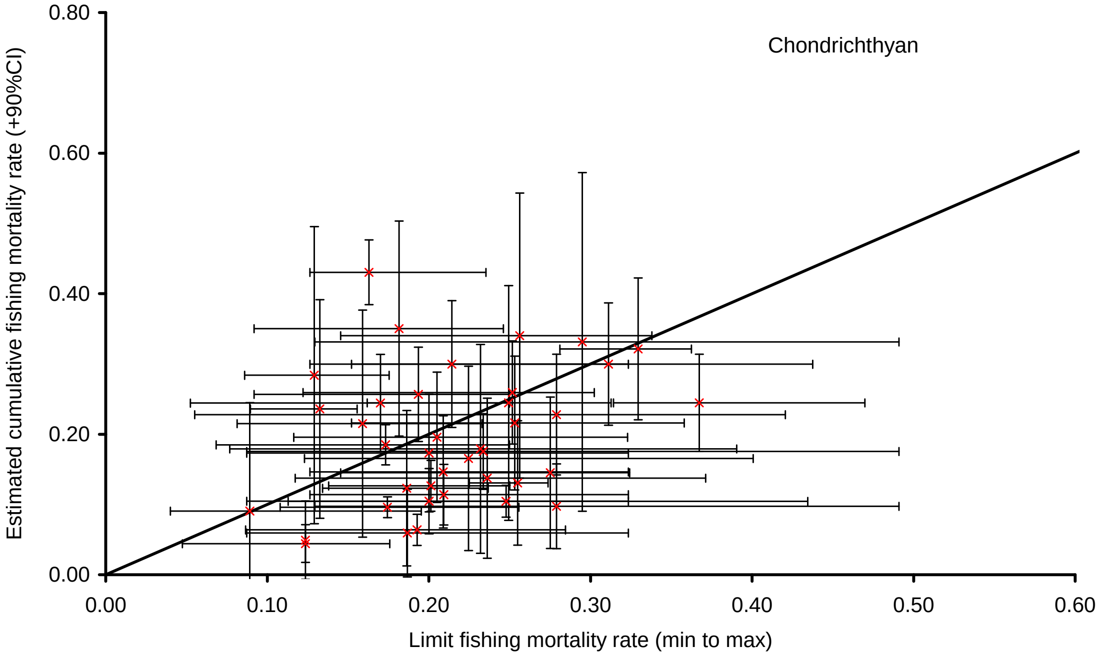


Figure 3-15. Comparison of estimated cumulative fishing mortality rate from five sub-fisheries within the fishery jurisdiction and the fishing mortality rate corresponding to the maximum sustainable fishing mortality for species caught in the SESSF. The diagonal line is where $u = u_{msm}$.



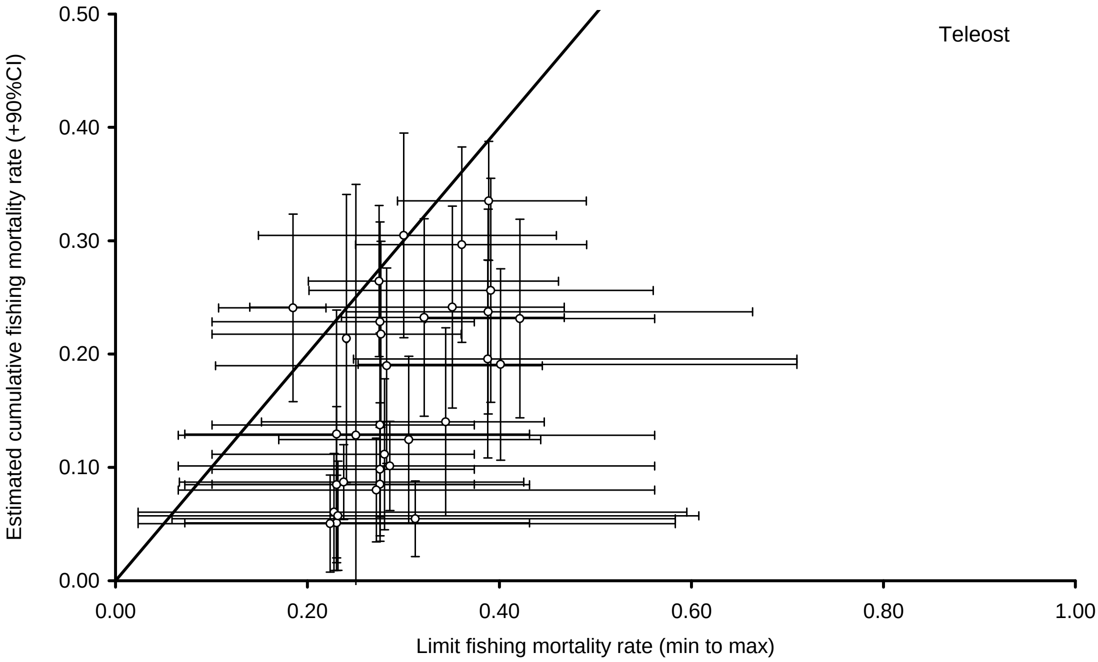


Figure 3-16. Comparison of estimated cumulative fishing mortality rate from five sub-fisheries within the fishery jurisdiction and the limit fishing mortality rate for species caught in the SESSF. The diagonal line is where $u = u_{lim}$.

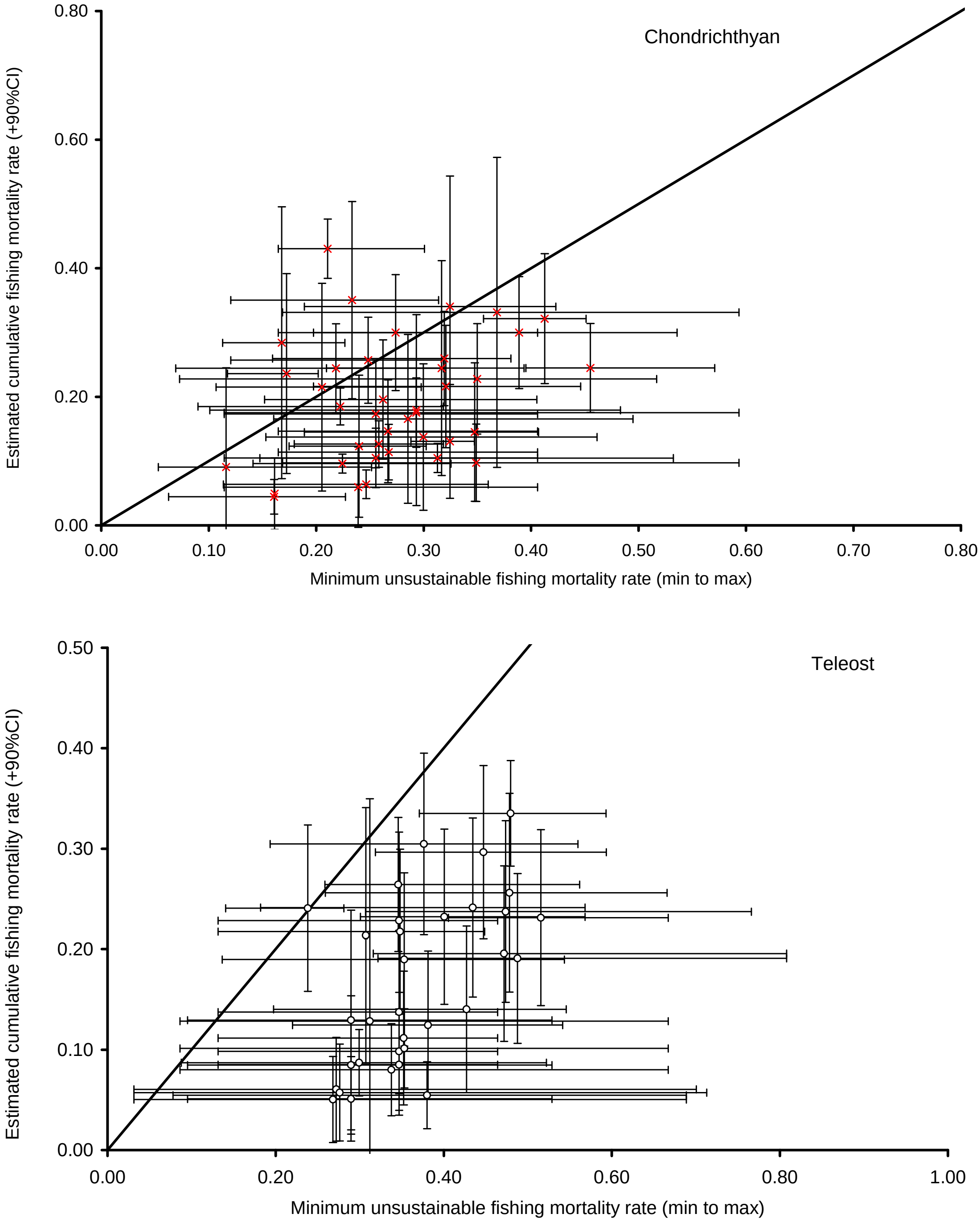


Figure 3-17. Comparison of estimated cumulative fishing mortality rate from five sub-fisheries within the fishery jurisdiction and the minimum unsustainable fishing mortality rate for species caught in the SESSF. The diagonal line is where $u = u_{crash}$.

CHAPTER 4. ASSESSMENT ON EASTERN TUNA AND BILLFISH FISHERY

4.1 Method

The ETBF fishery extends from Northern Queensland to Tasmania water (Figure 4-1). It reports catch and effort at high spatial resolution. We used the similar method developed in Chapter 2 for Auto Longling fishery to estimate fishing mortality rate for pelagic longline fishery.

We estimated effort area from shot start and end locations, analysed as an arc between the coordinates and overlayed on a 1 km^2 grid. Since gears are set at 30 to 400 m below the surface, we limited the species distributions to waters greater than 30 metres depth for estimating fishing efforts. The gear affected area will be certainly underestimated in this way if without correction. We included a correction factor ρ as in equation (8), Chapter 2. This parameter ρ can be considered as a correction factor adjusting actual gear affected area (due to drifting and bait odour dispersion) and gear efficiency. ETBF fishery targets five main species: yellowfin tuna, bigeye tuna, broadbill swordfish, albacore tuna, and striped marlin. Using estimated region-specific fishing mortality rates, observed catch, and estimated exploitable populations we were able to obtain the mean ρ = 1.48 (SE = 0.82) from the first four species (D. Kolody, CSIRO, personal communication).

The habitat-dependent encounterability q^h_i was set to 0.33, 0.66, and 1.0 for species with low, medium, and high scores encountering the fishing gear in the PSA analysis (Webb et al. 2007). We assigned the size-dependent catchability q^λ_i based on average length at maturity as in PSA: 0.33 for fish < 50 cm or > 500 cm, 0.66 for fish between 50 and 100 cm and between 400 and 500 cm, and 1.0 for fish between 100 and 400 cm. We used S_i = 0.00, 0.34, and 0.67 for species that have low, medium, and high probability of surviving after being caught and returned to the water.

There is a total of 207 species in PSA. Among which 158 have spatial distribution data. For the remaining pelagic species (including the main target species) we assumed they occur in all fishable areas where the depth is greater than 30 m. This total area equals 3,146,362 km^2 in the jurisdiction of ETBF fishery.

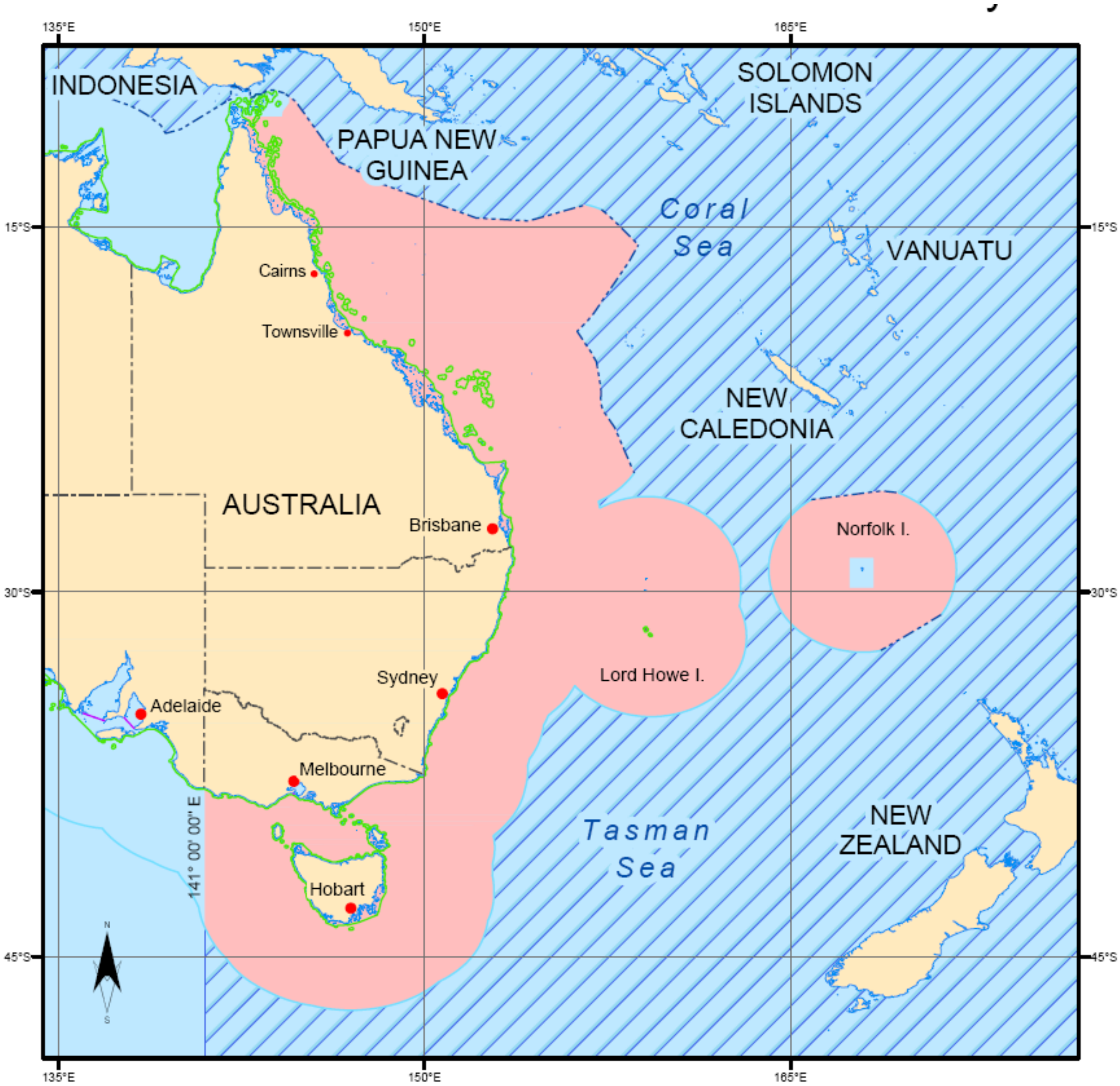


Figure 4-1. Area of the Eastern Tuna and Billfish Fishery.

4.2 Results

We assessed 207 fish species in ETBF fishery. We only present detailed information for species that are above the precautionary medium risk category. Eleven species (9 chondrichthyans and 2 teleosts) are at risk of potential overfishing (precautionary medium risk (m), either $E[u] \geq \min[u_{msm}]$ or $E[u]$ + 90%CI $E[u] \geq E[u_{msm}]$) (Table 4-1, Figure 4-2). Most species are included as precautionary risk due to uncertainty in the estimated fishing mortality rate or reference points. Only one species (*Zameus* *squamulosus*) has an estimated mean fishing mortality rate greater than its mean u_{msm}. Among these eleven species, eight are in the precautionary high risk category (either E[u] ≥ min[u_{lim}], or E[u] + 90%CI ≥ E[u_{lim}], Risk category h) (Figure 4-3). Further, four species are at precautionary extreme high risk category (either E[u] ≥ min[u_{crash}], or E[u] + 90%CI ≥ E[u_{crash}], Risk category e) (Table 4-1, Figure 4-4).

As for the SESSF, Table 4-1 includes all precautionary risk species when uncertainty is taken into account. The estimated impact (especially the upper 90% CI) may have been overestimated or the reference points underestimated (especially the minimum value of a reference points) for some species. Species that are overridden by experienced biologists or are felt uncertain are listed on the lower part of Table 4-1. Specific comments for these species are as follows.

Dalatias licha (Black shark): deepwater and mainly demersal species. They are likely accidental catches from cases where pelagic lines have sunk to the bottom and been recovered.

Centroscymnus plunketi (Plunket's shark): deepwater and mainly demersal species. They are likely accidental catches from cases where pelagic lines have sunk to the bottom and been recovered.

Deania calcea (Brier Shark): deepwater and mainly demersal species. They are likely accidental catches from cases where pelagic lines have sunk to the bottom and been recovered.

Rhincodon typus (Whale shark): plankton feeder and would only be caught via entanglement so likely at low risk.

Also, *Zameus sqamulosus* (velvet dogfish) is distributed mainly at depths below 300 m beyond the depths of pelagic longlines.

Biologists commented that several other species may be vulnerable to longline fishing impact, including *Makaira mazara* (blue marlin), *Makaira indica* (black marlin), *Xiphias gladius* (broad billed swordfish), *Prionace glauca* (blue shark), *Carcharhinus falciformis* (silky shark), *Carcharhinus longimanus* (oceanic whitetip shark). However, we estimated these species have low fishing mortality, mainly due to our assumption that they have wide distribution within the ETBF jurisdiction.

Table 4-1. Species, sorted by [u - u_{msm}], at potential risk of overfishing in the **ETBF fishery**. I_A = fraction of distribution area impacted; q = overall catchability, S = post-capture survival rate, Method = methods used for estimating the reference points. See text for risk category codes. Species without a risk code are in the precautionary medium risk (m) only.

					u		u_{msm}		u_{lim}		u_{crash}			
Scientific name	Common name	I_A	q	1-S	Mean	se	Mean	Min	Mean	Min	Mean	Min	Method	Risk
Isurus paucus	Longfin Mako	0.09	1.00	1.00	0.13	0.07	0.13	0.07	0.19	0.10	0.25	0.14	123456	h
Pseudocarcharias kamoharai	Crocodile Shark	0.09	1.00	1.00	0.13	0.07	0.17	0.17	0.24	0.24	0.31	0.31	5	h
Alopias pelagicus*	Pelagic Thresher	0.09	1.00	1.00	0.13	0.07	0.17	0.12	0.24	0.18	0.31	0.23	23456	h
Mola ramsayi	[an ocean sunfish]	0.09	1.00	0.33	0.04	0.03	0.11	0.02	0.15	0.03	0.20	0.04	356	eh
Mola mola	Ocean sunfish	0.09	0.33	1.00	0.04	0.03	0.11	0.02	0.15	0.03	0.20	0.04	356	eh
Alopias vulpinus	Thintail Thresher Shark,	0.03	1.00	1.00	0.05	0.03	0.17	0.05	0.24	0.08	0.30	0.10	23456	
Species overriden by experts														
Zameus squamulosus	Velvet dogfish	0.19	0.66	1.00	0.19	0.11	0.13	0.06	0.19	0.09	0.25	0.11	123456	ehM
Dalatias licha	Black Shark	0.28	0.33	1.00	0.14	0.10	0.14	0.06	0.20	0.09	0.26	0.11	23456	eh
Centroscymnus plunketi**	Plunket's shark	0.21	0.11	1.00	0.03	0.04	0.09	0.07	0.13	0.11	0.17	0.14	46	
Deania calcea	Brier Shark	0.20	0.22	1.00	0.07	0.05	0.14	0.09	0.20	0.14	0.26	0.18	23456	
Rhincodon typus	Whale shark	0.10	0.33	1.00	0.05	0.04	0.06	0.03	0.09	0.04	0.12	0.05	13456	h

Note:

*Pelagic thresher might be a misidentified species as this species is only observed in north-western Australia.

**the name of this species has changed to *Proscymodon plunketi*.

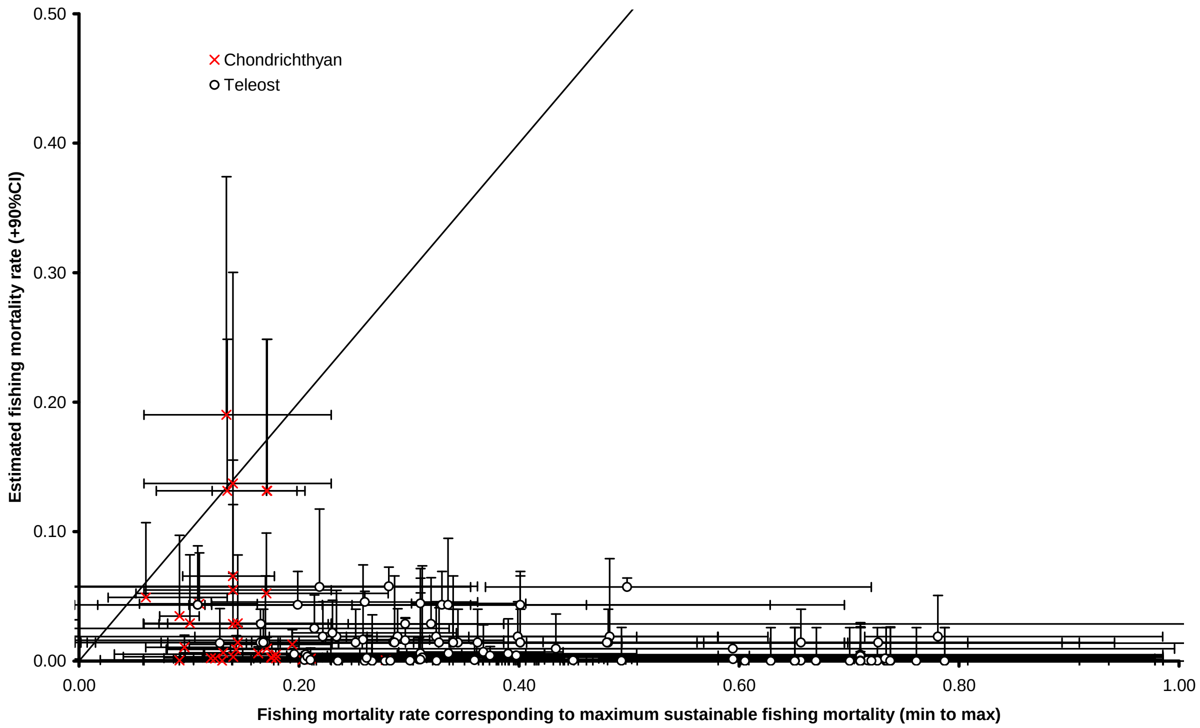


Figure 4-2. Comparison of estimated fishing mortality rate within the fishery jurisdiction and the fishing mortality rate corresponding to the maximum sustainable fishing mortality for fish species caught in ETBF fishery. The diagonal line is where $u = u_{msm}$.

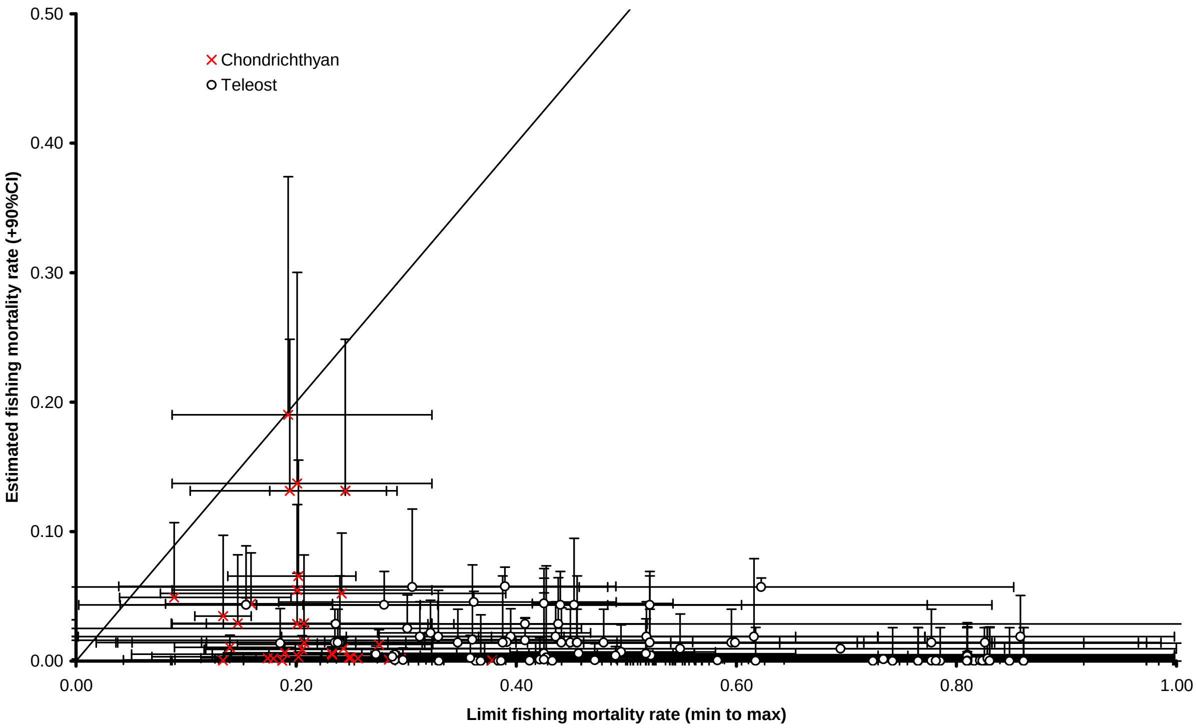


Figure 4-3. Comparison of estimated fishing mortality rate within the fishery jurisdiction and the limit fishing mortality rate corresponding to limit biomass $B_{lim} = 0.5B_{msm}$ for fish species caught in ETBF fishery. The diagonal line is where $u = u_{lim}$.

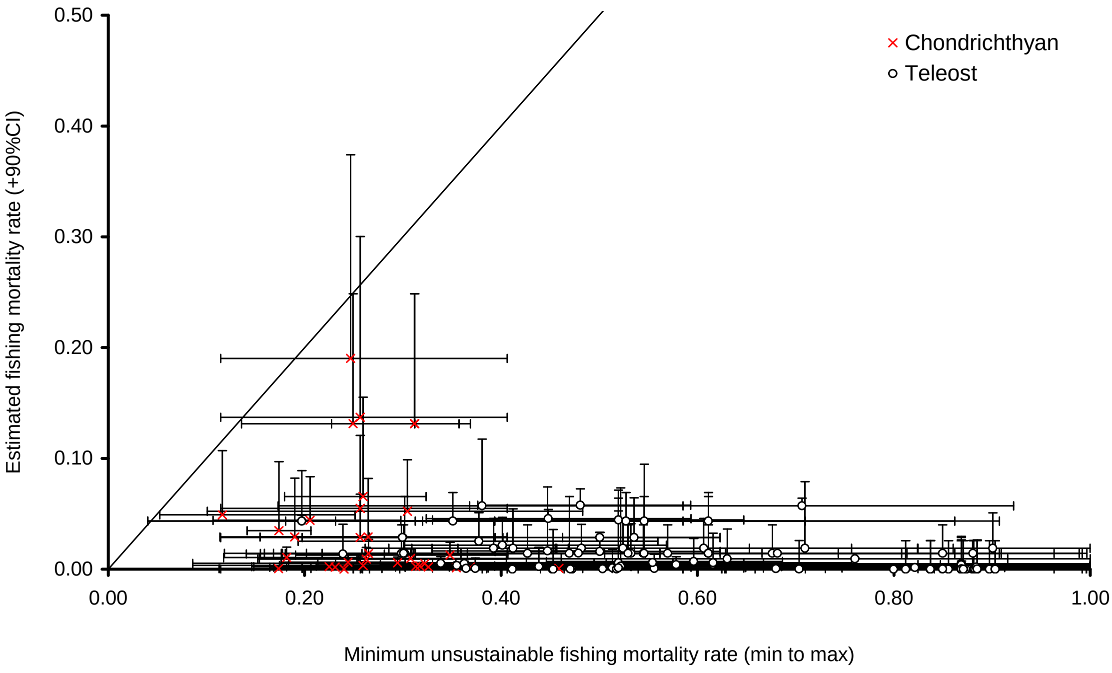


Figure 4-4. Comparison of estimated fishing mortality rate within the fishery jurisdiction and the minimum unsustainable fishing mortality rate for fish species caught in ETBF fishery. The diagonal line is where $u = u_{crash}$.

References

Webb, H., A. Hobday, J. Dowdney, C. Bulman, M. Sporcic, T. Smith, I. Stobustzki, M. Fuller, D. Furlani (2007) Ecological Risk Assessment for the Effects of Fishing: Eastern Tuna & Billfish Fishery: Longline Sub-fishery. Report for the Australian Fisheries Management Authority.

CHAPTER 5. REVIEW ON DATA AVAILABILITY FOR LEVEL 3 SAFE FOR OTHER COMMONWEALTH FISHERIES

In this project we have conducted SAFE analyses for three major Commonwealth fisheries: SESSF fishery (including 5 sub-fisheries), ETBF fishery, and the NPF. We also reviewed other Commonwealth fisheries for data availability for quantitative sustainability assessment. Two sets of data are necessary for such assessment: one set for estimating fishery impact and the other set for establishing reference points. Table 5-1 presents our assessment of the availability of suitable data for other fisheries.

Table 5-1. Data availability for quantitative risk assessment in other Commonwealth fisheries

Fishery	Sub-fishery or gear	Data for fishing impact	Data for reference point	L3 assessment
Western Tuna and Billfish Fishery	Pelagic Longline	1. Species spatial distribution. 2. Fishing effort 3. No target species F 4. Approximate catchability	Basic life history parameters	Possibly yes
Small Pelagic Fishery	Purse seine	1. Species spatial distribution. 2. Fishing effort 3. Approximate catchability	Basic life history parameters	Yes
	Mid-water trawl	1. Species spatial distribution. 2. Fishing effort 3. Approximate catchability	Basic life history parameters	Yes
Bass Strait Scallop Fishery	Dredge	1. Species spatial distribution. 2. Fishing effort 3. Approximate catchability	Basic life history parameters	Yes
Southern Bluefin Tuna Fishery	Purse seine	1. Species spatial distribution. 2. Fishing effort 3. Approximate catchability	Basic life history parameters	Yes
Southern Squid Jig fishery	Jig	1. Species spatial distribution. 2. Fishing effort 3. No target species F 4. Approximate catchability	Basic life history parameters	Low risk fishery. Possibly yes
Western Deepwater Trawl Fishery	Trawl	1. Species spatial distribution. 2. Fishing effort 3. Approximate catchability	Basic life history parameters	Yes
North West Slope Trawl fishery	Prawn trawl	1 Species spatial distribution. 2. Fishing effort 3. Approximate catchability	Basic life history parameters	Yes
Torres Strait Fishery	Prawn Fishery	1. Species spatial distribution. 2. Fishing effort 3. Approximate catchability	Basic life history parameters	Yes
	Tropical Rock Lobster			Low risk fishery

Sub-Antarctic Fishery	Macquarie Island trawl	1. No spatial distribution. 2. Fishing effort 3. Approximate catchability	Basic life history parameters	Possibly no
	HIMI demersal trawl	1. No spatial distribution. 2. Fishing effort 3. Approximate catchability	Basic life history parameters	Possibly no
	HIMI mid-water trawl	1. No spatial distribution. 2. Fishing effort 3. Approximate catchability	Basic life history parameters	Possibly no
	HIMI longline	1. No spatial distribution. 2. Fishing effort 3. Approximate catchability	Basic life history parameters	Possibly no
Coral Sea Fishery	Aquarium Sub-fishery (Hand collection)			Low risk fishery. Assessment unimportant
	Auto longline Sub-fishery	1. Species spatial distribution. 2. Fishing effort 3. Target species F NA 4. Approximate catchability	Basic life history parameters	Possibly yes
	Demersal longline Sub-fishery	1. Species spatial distribution. 2. Fishing effort 3. No target species F 4. Approximate catchability	Basic life history parameters	Possibly yes
	Demersal Trawl Fishery	1. Species spatial distribution. 2. Fishing effort 3. Approximate catchability	Basic life history parameters	Yes
	Finfish Trap Trials Sub-fishery	1. Species spatial distribution. 2. Fishing effort 3. Approximate catchability	Basic life history parameters	Possibly yes
	Lobster and Trochus Sub-fishery			Low risk fishery. Assessment unimportant
	Other Line sub-fishery			
	Sea Cucumber Sub-fishery			Low risk fishery. Assessment unimportant
Skipjack Tuna Fishery	Purse seine	1. Species spatial distribution. 2. Fishing effort 3. Approximate catchability	Basic life history parameters	Yes

This review indicates that we have enough data to carry out quantitative risk assessment for the majority of Commonwealth managed fisheries, assuming we will use the similar methods developed in this report. To apply existing method, the essential data may include: species spatial distribution, annual fishing effort and its spatial distribution, relative catchability (i.e., habitat-dependent encounterability, size-dependent selectivity), intrinsic population growth rate, natural mortality rate, growth parameter, longevity, age at maturity, exploitation rate estimate for target species in the same fishery, etc. Note that it is unnecessary to have all these data. Existing methods may need to be modified for new gear types.

One exception in Table 5-1 is the sub-Antarctic fishery, which includes four sub-fisheries. This fishery currently lacks species spatial distribution information. Therefore, it is difficult to apply the existing method to estimate fishing impacts. However, albeit may be possible to estimate species distribution or to develop new methods for this fishery given sufficient time. Alternative methods have recently become available for generating species distributions which may be utilised (e.g. http://www.aquamaps.org/).

For fishing gears that use bait to attract fish, i.e., hook and line fishery and trap fishery, it is preferable that fishing mortality rates have been estimated for some species (generally these are target species). The known fishing mortality rates can then be used to improve the accuracy of fishing mortality rate estimation for non-target species.

CHAPTER 6. DISCUSSION

In this project, we undertook a rapid quantitative species-by-species approach to assessing the effects of fishing on the sustainability of hundreds of data-poor target and non-target fish species. As discussed in Zhou and Griffiths (in press) and Zhou et al. (in review), this may be a simple and feasible approach to help achieve the objectives of EBFM by allowing an assessment of whether the majority of species are being impacted within sensible limit reference points. We summarize the pros and cons of this approach as follows.

Major advantages include:

* Less data demanding: this approach does not require fishery time-series data. Only one or a few life history parameters will be sufficient for establishing sustainability reference points. By making some key simplifying assumptions, it circumvents the need for full stock assessments on large numbers of impacted species by using limited information.
* Flexible: it focuses on one single indictor – fishing mortality rate. This allows alternative methods to be used to estimate fishing impact depending on available data while the measurements are in the "same currency" for easy comparison and possible assessment of cumulative impact.
* Scientific: the concept and method are based on existing fishery knowledge.
* Comprehensive: it can assess all species including target and non-target species in a batch process.
* Precautionary: the method is more scientifically rigorous as uncertainty in both indicator and reference points can be quantified.
* Cost effective: resource requirements on data and analytical time are minimal. It is a one-step process to assess all fish species impacted by a fishery.
* Transparent: all processes in estimating fishing mortality rate and reference points are quantitatively formulated.
* Impacts additive: assessing cumulative fisheries impacts is straightforward.
* Management application: results can be easily incorporated into fishery management plan, because this framework is similar to the typical management regimes used for target species.

The main cons and challenges are:

* Estimated fishing mortality rate may have high uncertainty and may not be accurate for a range of species.

* Relationship between sustainability and life history parameters may differ among taxonomic groups/species.

We eliminate detailed discussion on general characteristics of this approach, but focus on specific challenges encountered in this project.

6.1.1 Comparison between SAFE framework and management policy for target species

The generic framework of SAFE is in line with fishery management approaches for target species. It is essentially the same as the Tier 3 harvest control rule for the SESSF (DAFF 2007), though using different methods to estimate fishing mortality. In Tier 3 harvest strategy, fishing mortality F is estimated from catch curves (age-length data). The decision rules are based on F/M ratio. A recommended biological catch (RBC) is calculated as a proportion of average total catch over the previous four years. This proportion is set to zero when current fishing mortality is more than double natural mortality ($F > 2M$), set to 1 when $M > F > 0.75M$, and set to 1.2 when $F < 0.5M$. The first two cut-off points are similar or close to our F_{crash} and F_{msm} reference points.

The SAFE framework is also very similar to the widely-applied approach for marine mammal assessment and management. For example, the United States and the International Whaling Commission use the following model to estimate potential biological removal level, PBR (Sainsbury et al. 2000; Taylor et al., 2000; Wade, 1998):

$$PBR = N_{min} \times 0.5 \times R_{max} \times F_K$$

where: N_{min} = the minimum population estimate of the stock, R_{max} = the maximum theoretical or estimated net productivity rate of the stock at a small population size, and F_K = a recovery factor between 0.1 and 1. Here, R_{max} is equivalent to our intrinsic population growth rate r, PBR/N_{min} is equivalent to our fishing mortality rate u, and we use $F_K = 1$ for bycatch species.

6.1.2 Estimating fishing impacts

We developed feasible methods to estimate fishing impacts by different gear types using limited information. However, all methods involve similar steps and include similar components: fishery distribution, fishing gear affected area, species distribution, habitat-dependent encounterability, size-dependent catchability, and post-capture mortality. Each component in the model contains uncertainty, which will contribute to the final bias and uncertainty of the estimated fishing mortality rate. We have considered various sources of uncertainties and have taken into account as many sources as possible in the estimation, but it is not possible to incorporate them all.

Area overlap of fishing effort with species distribution, defined as availability in the Level 2 ERAEF (Smith et al. 2006), is critical for estimating fishing impact. We use species bioregionalisation data and refined core ranges for estimating species distribution (Commonwealth of Australia 2005; Heap et al. 2005). No variation or uncertainty is considered for this parameter. The true species distribution range may be underestimated by using core distribution range, since we found significant catches of some species occur outside core distribution range. Distributions of fishing activities are recorded in commercial logbooks. We explicitly include variability in effort distribution in our estimation of fishing mortality rates. By using area overlap of species distribution with fishing effort, we assume that individuals of fish randomly or homogeneously distribute within their distribution range, and fish densities are the same between fished and unfished areas within species distribution range. We believe it would be more accurate if we have data on relative abundance or density between fished and unfished area, as it has been applied for the NPF bycatch risk assessment (Zhou and Griffiths, in press). To some extent habitat preference information can improve resolution of this issue.

One potential source of bias is the estimation of the gear affected area. To improve the accuracy, we have developed separate methods for each type of fishing gear. The methods are based on available data and our best knowledge from literature on performance of each type of gear. Estimating affected area for gears that use bait to attract fish (i.e., longline) was the most difficult. We borrow information from target species to improve estimation of gear affected area and gear efficiency, but behavioural response will of course vary across species. Obviously, the reliability of fishing mortality estimates for target species will affect the accuracy of other species in our assessment.

The overall catchability parameter is the combination of encounterability and selectivity. Encounterability is the likelihood that a species will encounter fishing gear deployed within its range. We use habitat information from FishBase, modified by bathymetric information to assign approximate values for encounterability. To be more conservative, we chose the maximum value from each of the three categories for all species in that encounterability category, i.e., 0.33 for 0-0.33, 0.66 for 0.34-0.66, and 1.0 for 0.67-1.0. This is simplistic as no account is made of the difference in encounterability due to the different gear configurations used to target different species. For example, in the ETBF fishery when targeting yellowfin tuna the longline is usually set shallow (above the thermocline) whilst when targeting albacore tuna the longline is usually set very deep (down to 400m). Changing the configuration of the longline will significantly influence encounterability. As we lack detailed fishery information we did not differentiate gear configurations. Indeed, there needs to be another dimension added to the analysis to account for changes in the vertical profile of the species distribution in the future research.

The size-dependent selectivity is a measure of the likelihood that the species will be caught by the gear. Factors affecting selectivity may include gear specification, fish body size, fish morphology, and fish behaviour. However, body size in relation to gear size is the most important attribute for this aspect. To be more conservative, we also choose the maximum value from each of the three categories for all species in that selectivity category, i.e., 0.33 for 0-0.33, 0.66 for 0.34-0.66, and 1.0 for 0.67-1.0.

To account for uncertainty in overall catchability parameters, we assume that the capturing process is a Bernoulli trial: being caught or not. Therefore, we use the binomial distribution to estimate the variance. We recognize that including statistical uncertainty alone may underestimate the true variance in this parameter.

For species that are caught by the gear, post capture survival rate measures the survival probability of the species after being returned to the water. The value of this parameter is mainly based on independent field observations or expert knowledge. We choose the minimum value from each of the three categories for all species in that survival category, i.e., 0 for 0-0.33, 0.34 for 0.34-0.66, and 0.67 for 0.67-1.0. We also use the binomial distribution to account for uncertainty in the post-capture survival rate, i.e., a fish either survives or dies after returning to the water.

The practice of using the maximum value from each of the three categories for all species in the catchability and post-capture mortality categories may overestimate the actual fishing impacts.

6.1.3 Cumulative impacts

The overall impact on ecological sustainability is the cumulative fishing mortality rate from all sources of human activities. Yet, cumulative impact is one of the most difficult subjects to deal with by using other ERA methods. Our approach makes cumulative impact easy and straightforward to estimate. For example, the SESSF fishery is a complex multi-sector, multi-gear and multi-species fishery. As we have estimated fishing mortality rate for each individual sub-fishery and this quantity is measured in the same unit, it is straightforward to sum it across all sub-fisheries to obtain the total cumulative fishing mortality rate. We have accomplished this extra task, which was not planned for the project.

6.1.4 Reference points derivation

We use a total of six alternative methods to derive reference points to increase our confidence in the assessment result and to measure uncertainty. Theoretically, the method based on intrinsic population growth rate is the most defensible approach. However, as we know this parameter is difficult to obtain for many fish species, we have collected as many *r* values as possible from the literature. Since we are not assured of its reliability, we do not solely depend on intrinsic population growth rate and give this method the same weight as other methods. Similarly, since no single method appears to be the best, we treat all six methods as of equal importance.

6.1.5 Using expert opinions

At the end of our quantitative assessment for each fishery, we solicited expert opinion from biologists who have first-hand knowledge about the fishery and species on the risk list. This process is similar to that in the level 2 PSA. However, experts differed between the two processes and experts may have different experience and opinions. For this reason, the species where overrides were applied may not be exactly the same as in the previous PSA assessment. The documentations on the reasons for the overrides are included in the body of this report.

6.1.6 Comparison with PSA

It is interesting to compare the results from this project with previous assessment using qualitative or semi-quantitative methods. However, direct comparisons are difficult because of the following differences between the SAFE and PSA (Stobutzki et al. 2001; Stobutzki et al. 2002; Hobday et al. 2007):

- The SAFE framework is similar to the conventional management of target species in that it applies explicit reference points to estimates of fishing mortality rates (DAFF

2007). While the PSA method is based on similar information on productivity and susceptibility of species, it does not provide quantitative estimates of mortality.

- SAFE provides a clear definition of the type of ecological risk (comparison of fishing mortality rate to reference points) while such a definition is unclear in PSA. For example, the High risk in PSA does not necessarily mean the species is unsustainable or at risk of overfishing, and does not correspond directly to any risk category in SAFE.

- As for some target species management, SAFE considers natural mortality to be very important in establishing reference points and uses published methods to expressly quantify these reference points; PSA does not consider natural mortality but includes a range of life history traits as surrogates and treats all the attributes as of equal importance in determining productivity scores.

- SAFE expressly quantifies uncertainty for both fishing mortality rates and reference points while PSA has difficulties to include uncertainty in the attributes.

The reliability of the semi-quantitative method has been evaluated in the NPF (Grifffiths et al. 2006). Although here we provided simple comparison of the results from the two approaches, caveats should be taken for the practical usefulness of such a comparison.

SESSF-Otter trawl sub-fishery: PSA resulted in a total of 154 fish species (58 chondrichthyans and 96 teleosts) at high risk (Wayte et al. 2006). Our SAFE indicated that 21 species are at risk of potential overfishing after taking over-rides by experts into account (Precautionary medium risk, $E[u] \geq \min[u_{msm}]$, or $E[u]$ + 90%CI $\geq E[u_{msm}]$). Among these 21 species, 19 were categorized as high risk and 25 as medium risk in the PSA.

SESSF- Great Australian Bight Trawl sub-fishery: PSA resulted in a total of 57 fish species (34 chondrichthyans and 23 teleosts) at high risk (Daley et al. 2007a). Our SAFE indicated that there is no species at risk of potential overfishing. However this needs to be treated with some caution as the fishery is expanding rapidly onto the upper slope and so changes in distribution of effort will need to be monitored closely.

SESSF-Shark Gillnet sub-fishery: PSA resulted in a total of 16 fish species (all are chondrichthyans) at high risk (Walker et al. 2007). Our SAFE indicated that there are 11 species at risk of potential overfishing after taking into account of over-rides by experts ($E[u] \geq \min[u_{msm}]$, or $E[u]$ + 90% CI $\geq E[u_{msm}]$). These are all chondrichthyans and all were categorized as high risk in PSA.

SESSF-Danish Seine sub-fishery: No species is considered as high risk in PSA (Wayte et al. 2007). Our SAFE agrees with PSA assessment that there is no species at risk of potential overfishing.

SESSF-Automatic Longline sub-fishery: PSA resulted in a total of 28 fish species (20 chondrichthyans and 8 teleosts) at high risk (Daley et al. 2007b). Our SAFE indicated that there are 13 species at risk of potential overfishing after experts' overriding ($E[u] \geq u_{msm} -$ 90%CI, or $E[u]$ + 90%CI $\geq u_{msm}$), including 9 chondrichthyans and 4 teleosts. Among these 13 species, 12 were categorized as high risk and 1 as medium risk in PSA.

ETBF: PSA resulted in a total of 5 fish species (all are chondrichthyans) at high risk (Daley et al. 2007b). Our SAFE indicated that there are 7 species at risk of potential overfishing after experts' overriding ($E[u] \geq u_{msm} -$ 90%CI, or $E[u]$ + 90%CI $\geq u_{msm}$), including 5 chondrichthyans and 2 teleosts. Among these 7 species, only 1 was categorized as high risk, 4 at medium risk, and 1 as low risk in PSA.

6.1.7 Recommendations

As mentioned at the beginning of this chapter, one of the challenges in using the SAFE method is the potential bias and uncertainty in the estimated fishing mortality rate. Accuracy can be improved by collecting additional data, such as species distribution within fished and unfished areas, observation on catchability and post-capture survival rate, target species fishing mortality estimation, etc.

The other challenge is that the relationship between sustainability and life history parameters may differ among taxonomic groups or species. Setting $F_{msm} = M$ may not be appropriate for every species. Further research is needed to investigate the more reliable relationship between sustainability and simple life history parameters for different taxonomic groups of fish.

So far we have undertaken quantitative risk assessments on three major Commonwealth fisheries: SESSF, ETBF, and NPF. Similar assessments for most of the remaining Commonwealth fisheries are also feasible.

The main objective of this project is to evaluate fishing impact on sustainability of fish species in selected Commonwealth fisheries. The assessment focuses on the current impact -- averaging fishing effort over the last four years from 2003 to 2006. For the purpose of practical management on bycatch species, we suggest that assessment be carried out on a regular basis, which may be annual if the distribution or amount of fishing effort is changing rapidly in the fishery. Such assessments will allow management rules, such as the preliminary rules suggested in Table 2-1, to be implemented in practical management of the fishery.

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APPENDIX 1. SUSTAINABILITY ASSESSMENT FOR FISHING EFFECTS (SAFE): A NEW QUANTITATIVE ECOLOGICAL RISK ASSESSMENT METHOD AND ITS APPLICATION TO ELASMOBRANCH BYCATCH IN AN AUSTRALIAN TRAWL FISHERY

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Abstract

We present a quantitative approach to the ecological sustainability assessment for fishing effects (SAFE) of diverse and data-poor bycatch assemblages. The method estimates fishing impact and compares the impact with sustainability reference points based on basic life-history parameters. We demonstrate the effectiveness of this method by assessing the impact of Australia’s Northern Prawn Fishery on the sustainability of 51 elasmobranch bycatch species. We estimated the proportion of the population distributed within trawled areas, from detection-nondetection data collected from scientific surveys. This estimate of species’ abundance was then included in a model incorporating catch rate and escapement probability to give an estimate of the fishing mortality rate of each species. To guide management of bycatch species, we established two reference points based on natural mortality rate and growth rate: maximum sustainable fishing mortality rate and minimum unsustainable fishing mortality rate. The proportion of the 51 species’ populations distributed within the fished area ranged between 0.02 and 1.00 (mean 0.36 ± SD 0.31). Our results indicated that fishing impacts may have exceeded the maximum sustainable fishing mortality rate for 19 species, and exceeded the minimum unsustainable fishing mortality rate for 9 species. However, the estimates were highly uncertain for

some species. SAFE can also be used by scientists and fishery managers to focus monitoring programs on potentially at-risk species to obtain additional data for further sustainability evaluation. Because the framework of SAFE is compatible with the management of target species, it can be incorporated into existing fishery management strategies, and may fulfil emerging ecosystem-based fishery management objectives.

Keywords: Ecological risk assessment, bycatch, fishing effects, sustainability, detection- nondetection

1. Introduction

Ecosystem approaches to fisheries management are being developed worldwide to conform to increasingly stringent environmental and fisheries legislation. There is a growing body of evidence that fishing activities adversely affect populations of non-target species (i.e. bycatch) and physical damage to habitats (Hall and Mainprize, 2005). These impacts can lead to changes to biodiversity and ultimately change the overall functionality of the ecosystem (Pitcher and Chuenpagdee, 1994; Crowder and Murawski, 1998; Pauly et al., 1998; Dulvey et al., 2000; Harrington et al., 2005). Although broad management policies and objectives exist for ecosystem-based management (FAO, 2003), translating them into action is difficult. Fishery scientists and managers often do not have the information required to properly assess fishery impacts on non-target species and communities, and to develop management measures to ensure the fishery operates in an ecologically sustainable manner. To move closer towards fulfilling the broad objectives of ecosystem-based fisheries management, approaches need to be developed that can cope with the high species diversity and limited data that is typical of many fisheries worldwide, especially in tropical regions.

The Northern Prawn Fishery (NPF, Fig. 1) is one of the first Australian fisheries to attempt to tackle the challenge of demonstrating the ecological sustainability of its supporting ecosystem through ecological risk assessment (ERA) (Milton, 2001; Stobutzki et al., 2001a, 2002). The fishery targets six prawn (shrimp) species with twin demersal trawl gear and operates as three spatially and temporally distinct sub-fisheries. However, the main trawl impact is from the tiger prawn fishery, which primarily targets Penaeus semisulcatus and P. esculentus only during the night from August to November. Because tiger prawns are generally widely dispersed, the trawls are generally long (~3 h), which often results in large catches of unwanted bycatch, most of which is discarded dead. Pender et al. (1992) estimated that around 30,000 t of bycatch was discarded annually in this fishery. The bycatch, which often comprises more than 95% of the catch, is diverse, including invertebrates (234 taxa), teleosts (366 spp.), elasmobranchs (51 spp.), turtles (8 spp.) and seasnakes (13 spp.) (Stobutzki et al., 2001a; Griffiths et al., 2004).

An semi-qualitative, attribute-based ecological risk assessment technique developed concurrently by Stobutzki et al. (2001b, 2002) and Milton (2001) was applied to teleost, elasmobranch and seasnake bycatch in the NPF. The relative sustainability of bycatch species was examined by ranking species with respect to their susceptibility to capture; mortality due to prawn trawling; and capacity to recover once the population becomes depleted. However, the attribute-based method has a number of drawbacks. It provides only a relative measure of risk among the group of species examined, and gives no indication of whether the populations of the highest-risk species are truly unsustainable, or the lowest-risk species are truly sustainable. Furthermore, the term “risk” and “sustainability” are not clearly defined, thus providing no basis on which to assess the status of individual bycatch species. Griffiths et al., (2006) recently demonstrated that this method is not sensitive to changes in the size selectivity of species as a result of changes in fisheries management strategies, and can inadequately reflect even the most obvious changes in risk to individual species.

In this paper we describe a practical method for assessing the impact of fishing on large numbers of non-target, low economic value and data-poor species, and to establish sustainability reference points that management can use at an operational level. We refer to this method as a Sustainability Assessment for Fishing Effects (SAFE). We use SAFE to assess the fishing impact on the sustainability of elasmobranch bycatch in the NPF tiger prawn trawl sub-fishery as a test case for our method. This group is of particular concern because of their slow growth, low natural mortality rates and low reproductive potential, which can make their populations vulnerable to decline from overfishing (Stevens, 1997; Walker, 1998; Baum et al., 2003).

2. Materials and methods

2.1 Data sources

Over 70 scientific voyages have been undertaken in the Northern Prawn Fishery (NPF) managed area between 1979 and 2003, mostly by CSIRO Marine and Atmospheric Research and a few by state fisheries agencies. Together, the surveys covered the entire NPF, although not in any one voyage (Fig. 1). We used data from these surveys to assess fish distribution. Where no catch was recorded, we could not estimate abundance by conventional techniques. Since this is a common problem in fisheries worldwide, we pooled the data from all scientific surveys to maximize the sample sizes and geographical coverage, but used detection-nondetection information to estimate bycatch species' distribution and abundance in the region.

To model the abundance of bycatch species using detection-nondetection data, we defined a sampling unit as a 6 by 6 nautical mile grid, which is currently used in NPF logbooks for reporting purposes. There are a total of 6,963 grids in the NPF-managed area. The composition of bycatch species varies spatially within the NPF (Blaber et al., 1990, 1994; Stobutzki et al., 2001a; Tonks et al., in press), as well as with sampling effort. Therefore, we stratified the NPF-managed area into five bioregions based on established bioregions for fishes (IMCRA, 1998) and expert opinion (Fig. 1). During the surveys, a total of 5,835 samples were taken in 924 grids, using trawl gear of various types. Some grids were repeatedly surveyed over a number of years. The sampling rate in bioregion 4 was higher than in the other bioregions because it had a higher fishing effort and consequently was surveyed more often to investigate fishery-related problems.

Fishing effort varies spatially over time in NPF. We obtained data regarding the spatial distribution of fishing effort from compulsory fishery logbooks in the tiger prawn fishery from 1999 to 2003 in order to assess the impact of this fishery on elasmobranch bycatch species during this time period.

2.2 Estimating fishery impacts

Species-specific fishing-induced mortality rate was derived from a number of variables: the proportion of the entire management area trawled; the relative abundance of individual bycatch species in trawled areas compared with the total area; the probability of a fish on a trawl track entering the net; and the probability of a fish escaping from the trawl after it has entered the net.

We defined the "fished area" as grids where the total fishing effort recorded in logbooks was >5 boat days over 5 years between 1999 and 2003. Five days of fishing effort is equivalent to about 10% of sea floor within the grid being systematically swept by prawn trawls in 5 years, assuming trawling occurs for 12.3 hours per day (Rawlinson, 2003) at a speed of 3.24 knots (Bishop, 2003) with a headrope length of 14 fathoms and a 0.66 spread ratio (Bishop and Sterling, 1999). This criterion may overestimate fishing impact, as trawls are unlikely to sweep the entire "fished area". Further, because trawl tracks often overlap, the actual impact was expected to be less than 10% (Stobutzki and Pitcher, 1999). We treated grids where the fishing effort was ≤ 5 boat-day as the "unfished area".

The distributions of individual bycatch species and their spatial overlap with the trawled area indicates which species are most likely to be affected by the fishery. Although the true impact of the fishery on a species' population would be best determined by taking into account its entire distribution, our main interest is the local sustainability within the approximate 700,000 km^2 area of the NPF.

The proportion of a species' population in the fished area relative to the entire NPF-managed area is an indicator of a fishery's impact on the species' distribution. We derived this parameter by a new quantitative method to estimate the abundance of each species through detection-nondetection data (Zhou and Griffiths 2007), which are easier and more cost-effective to collect than count data. These data are also more widely available, and hence, our approach is more easily transferred to other data-poor fisheries.

Theoretical and field studies indicate that the pattern of presence and absence over a geographic area closely reflects actual animal abundance (Kunin, 1998; Kunin et al., 2000; He and Gaston, 2000a, 2000b; MacKenzie and Kendall, 2002; Nielsen et al., 2005). Estimating the proportion of a geographical area occupied by a particular species from such data has been considered useful in long-term monitoring programs and metapopulation studies (Azuma et al., 1990; MacKenzie et al., 2004). A particular concern of using detection-nondetection data is the presence of false-negative (or false absence) errors. This can occur if a sample does not capture/detect a species when it is, in fact, present. To avoid this, the probability of false-negative errors should be incorporated into models of binary data (Bayley and Peterson, 2001; MacKenzie and Kendall, 2002; MacKenzie et al., 2002; MacKenzie et al., 2003; Tyre et al., 2003; Royle and Nicholes, 2003; Gu and Swihart, 2004), as we have done in our model to estimate abundance.

After stratifying the NPF into five bioregions and fished and unfished areas within each bioregion, individuals of each elasmobranch species were assumed to be randomly distributed within each strata (bioregion as well as within the fished or unfished areas). We believe that this assumption is appropriate for tropical elasmobranchs, because they are generally not encountered in large aggregations in the NPF. The probability that a surveyed grid is occupied by a particular species was held to be directly related to the total abundance of the species in the entire study area. Conditional on the species actually existing in a surveyed grid, it may not be captured in every survey. Therefore, the result for any given survey can be considered as two binomial processes working simultaneously: the probability that a species is present in the grid, and the probability that one or more individuals of that species are captured given the species is indeed present in the grid. Repeated surveys within the same grid allow the estimation of the total abundance, or mean density, of a species in the study area. Because the scientific surveys spanned many years, we assumed that the relative abundance of each bycatch species between fished areas and unfished areas remained constant during this period (see Discussion). The model of Zhou and Griffiths (2007) has two components: firstly, for grids where one or more individuals were detected in at least one survey; and secondly, for grids where the species was never detected, but may actually be present. For each surveyed grid i, assume that a total of m_i surveys have been conducted, of which a species has been captured in n_i surveys ($n_i \leq m_i$). The combined likelihood across all surveyed grids within a specific region R is:

$$L(N_F, D \mid m_i, n_i, A_F, a_i) = \prod_{F=0}^{1} \prod_{\substack{i=1 \\ n_i>0}}^{C_1} \left\{ \binom{m_i}{n_i} D^{n_i} (1-D)^{m_i - n_i} \left[1 - \left(1 - \frac{a_i}{A_F} \right)^{N_F} \right] \right\} \times \prod_{\substack{i=1 \\ n_i=0}}^{C_0} \left\{ (1-D)^{m_i} \left[1 - \left(1 - \frac{a_i}{A_F} \right)^{N_F} \right] + \left(1 - \frac{a_i}{A_F} \right)^{N_F} \right\} \quad (1)$$

where N_F = total abundance in area F (F = 0 unfished, F = 1 fished), D = the probability of detecting (capturing) one or more individuals, A_F = the size of area F of the study region, a_i = the size of area surveyed in grid i, C_1 = total number of grids where $n_i > 0$, and C_0 = total number of grids where n_i = 0. Unknown parameters N_F and D can be solved by maximizing the likelihood in this equation. We assumed that fish density differed between fished and unfished areas within each bioregion R. We also assumed that the probability of capturing a particular species was constant within each bioregion, but was specific to the fishing gear used. Eight gear types were used in the surveys, each with different species and size selectivities. These were: benthic sled, Engels trawl, Florida Flyer benthic trawl, Florida Flyer trawl with a bycatch reduction device (BRD), Florida Flyer trawl with a turtle exclusion device (TED), Frank and Bryce fish trawl, Julie Ann net, and a modified semi-pelagic Julie Ann net. Therefore, we used a logistic model to incorporate gear-specific probability of capture (D):

$$D = \frac{1}{\exp[-(\alpha + \boldsymbol{\beta}\mathbf{M} + \boldsymbol{\gamma}\mathbf{H})] + 1}, \quad (2)$$

where vector **M** is the sampling gear type, vector **H** is the area that each gear covers in each grid, and α, vectors $\boldsymbol{\beta}$ and $\boldsymbol{\gamma}$ are model parameters.

The proportion of a species' population that could potentially be impacted by trawling in the NPF, derived from the relative populations of each species within fished and unfished areas (P_N), was estimated as:

$$P_N = \frac{\sum_{R=1}^{5} N_{R,F=1}}{\sum_{R=1}^{5}\sum_{F=0}^{1} N_{R,F}}, \qquad (3)$$

where the subscript R is bioregion, F = 1 refers to the fished area, whereas F = 0 refers to the unfished area.

Finally, the fishing-induced mortality rate (u) was estimated by:

$$u = P_N q(1 - \mathrm{E}), \qquad (4)$$

where q is the species-specific catch rate and E is the species-specific probability of escapement after a fish enters the net. In this instance, the catch rate can be considered the probability of a fish entering the net along a track. The model (4) implies that we simplified the fishing process to uniformly sweep a grid once a year.

From commercial logbooks, we estimated that the average fishing effort in the fished areas from 1999 to 2003 (i.e. > 5 boat-day) was 43.5 boat-days per grid (SD = 46.1, n = 1,157). Using the same method and data described for P_A, this fishing effort could systematically sweep the seabed 0.89 times/year in the fished area (approximately 95% CI 0.80 – 0.99 times/year). Considering uncertainty and precaution, the assumption in equation (4) that assumes a grid is uniformly swept once a year is justified.

Catch rate estimates for each species were obtained directly from experimental trawl data (Pitcher et al., unpublished data). In cases where data were not available, we estimated the catch rate in one of three ways: (a) based on related species in the same genus for which catch rates were measured, since closely related species are likely to have similar vulnerability to capture, (b) based on values estimated by Blaber et al. (1990) for the same species; or (c) based on values of Blaber et al. (1990) for species having similar vertical distribution, size, and locomotory behaviour, referred to here as "ecomorphotypes" (Compagno, 1990; Bax et al., 1999).

We used the results of Brewer et al. (2004) to estimate the escapement of elasmobranch species from trawls due to turtle excluder devices (TEDs), which are compulsory in the NPF. Of the 51 elasmobranch species included in the present study, escapement estimates exist for 25 species. For the remaining 26 species, we assigned an escapement rate by averaging measured escapement rates from species in the same genus, family or the same ecomorphotype (see Compagno, 1990) that were measured by Brewer et al. (2006).

2.3 Uncertainty assessment

The estimated proportion of the NPF area fished is assumed to contain minimum uncertainty because the daily fishing locations are recorded in compulsory fishery logbooks. However, higher uncertainty may exist in the abundance estimates from scientific surveys, catch rates, and the probability of escapement due to TEDs. Since the abundance $\underline{N}_{R,F}$ and the resulting relative abundance ($\underline{P}_N$) are key factors affecting the fishing-induced mortality rate ($\underline{u}$), we evaluated uncertainty around these parameters. Approximate standard errors (SE) around $\underline{N}_{R,F}$ were derived from the square roots of the diagonal elements of the covariance matrix of the parameter estimates. This is the same as the inverse of the Hessian matrix (the matrix of second derivatives) of the likelihood. Variance of $\underline{P}_N$ was obtained from variance of $\underline{N}_{R,F}$ by a delta method (Zhou, 2002). Variance of $\underline{u}$ in equation (4) was also derived by the delta method from variances of $\underline{P}_N$, $\underline{q}$, and $\underline{E}$. Variances of $\underline{q}$ and $\underline{E}$ were calculated from binomial distributions, assuming both capture and escapement from trawl were binomial processes, i.e. $\theta \sim Bin(\underline{n}, E[\theta])$, where $\underline{n}$ is the sample size from field experiments or assumed samples and $E[\theta]$ is the expected probability of capture or escapement estimated from field studies.

2.4 Bycatch management reference points

For target fish species, one conceptual and general management goal is to achieve maximum sustainable yield (MSY). However, there is a lack of clear goals or practical guidelines for managing bycatch species at an operational level. We propose two reference points for bycatch species. The first is the maximum sustained fishing mortality (MSM), which is equivalent to MSY and a fishing mortality rate u_{msm} that corresponds to MSM. This reference point may be too conservative as a constraint for harvesting species of little economic value. The second reference point, or threshold, is the minimum fishing mortality rate that is expected to eventually render a population extinct in the long-term, referred to here as u_{crash}. This reference point corresponds to the management objective that the risk of possible extinction of any bycatch species should be avoided.

According to Graham-Schaefer's production model (Fletcher, 1978; Hilborn and Walters, 1992; Quinn and Deriso, 1999):

$$\frac{dB}{dt} = r(1-\frac{B}{B_\infty})B - FB = \frac{4m}{B_\infty}(1-\frac{B}{B_\infty})B - FB\,. \qquad (5)$$

This equation implies that the maximum instantaneous fishing mortality rate should not be greater than the intrinsic population growth rate $4m/B_\infty$. Therefore, we define:

$$F_{crash} = r = \frac{4m}{B_\infty} = \frac{2m}{B_m}\,. \qquad (6)$$

The instantaneous fishing mortality rate that corresponds to MSM is therefore:

$$F_{msm} = \frac{m}{B_m}\,. \qquad (7)$$

In the above equations, r = intrinsic growth parameter, B_∞ = pristine biomass, m = maximum productivity (equivalent to MSY), and B_m = biomass at which MSM occurs. Corresponding to instantaneous fishing mortality rate, a fraction of population loss due to fishing is: $u_{msm} = 1 - \exp(-F_{msm})$ and $u_{crash} = 1 - \exp(-F_{crash})$. For bycatch species in the NPF, there was insufficient information to conduct stock assessments to determine these parameters. The intrinsic ability of fish to sustain an extrinsic threat is fundamentally correlated with the life history traits of that species (Charnov, 1993; Jennings, 1998;Denney et al., 2002; Frisk et al., 2004; Reynolds et al., 2005; Goodwin et al., 2006). Among the many life history parameters that describe the life history strategy of a fish species, natural mortality M has widely been used as surrogate for F_{msy} for target species (Alverson and Peryra, 1969; Gulland, 1970; Quinn and Deriso, 1999). Therefore, in this first method, we set $F_{msm} = M$, and according to eqs. 5 and 6, $F_{crash} = 2M$. It has been argued that using M as a surrogate for F_{msy} may be risky for some target species (Garcia et al., 1989; Quinn and Deriso, 1999). Thompson (1993) suggested that a fishing mortality rate under 0.8M should keep a stock from collapsing in a model containing a depensatory spawner-recruit relationship. Deriso (1982) developed an upper bound for exploitation rates based on the delay-difference model:

$$u_{upper} \leq \sqrt{\frac{1}{\rho l^2 rv}}, \qquad (8)$$

where ρ = Brody's growth coefficient for weight, l = annual natural survival fraction for adults [l = exp(-M)], and $rv = [(1-\rho l)(1-l)]^{-1}$. In this second method we considered this exploitation rate to be equivalent to a fishing mortality rate that renders the population extinct, i.e. $F_{crash} = -\log(1-u_{upper})$. To be conservative, for each species the lower value of F_{crash} from these two methods was chosen as our reference point.

We obtained natural mortality M directly from the literature. In cases where M was not available, we obtained growth parameters from the literature and estimated M using one of five empirical equations depending on data available for a particular species:

(1) $\ln(M) = -0.0152 - 0.279\ln(L_\infty) + 0.6543\ln(k) + 0.4634\ln(T)$ (Pauly, 1980);

(2) $M = 10^{0.566-0.718\ln(L_\infty)} + 0.02T$ (www.Fishbase.org);

(3) $M = 1.6\,k$ (Jensen, 1996);

(4) $\ln(M) = 1.44 - 0.982\ln(t_m)$ (Hoenig, 1983); (9)

(5) $M = -\log_e(0.01)/t_m$ (Quinn and Deriso, 1999)

In these equations, k and L_∞ are von Bertalanffy growth parameters, T = average annual water temperature (in this case 28 °C), and t_m = maximum reproductive age.

3. Results

3.1 Fraction of area trawled

Prawn trawling activity was concentrated in a relatively small area, mainly in regions 4 and 5 (Fig. 1). During the last five years (1999-2003), the estimated mean annual impacts were 7% and 3% of NPF areas for effort > 0 boat-day and effort > 5 boat-day respectively. Because the spatial extent of the fished area often varies between years, the total impact (effort > 5 boat-day) in the last five years was about 6% of the NPF-managed area.

3.2 Trawling impacts on abundance distribution of bycatch species

The geographic distribution of the 51 elasmobranch species within fished areas and outside fished areas is shown in Table 1. Eight species were caught only in fished areas: Orectolobus ornatus, Carcharhinus leucas, Carcharhinus albimarginatus, Squatina sp. A, Taeniura meyeni, Urogymnus asperrimus, Himantura jenkinsii, and Rhinobatos typus, among which, four species were caught only once. Three abundant species were caught in more than 1,000 samples: Carcharhinus dussumieri, Himantura toshi, Gymnura australis.

The relative abundance of individual species within the fished area, P_N, ranged from 0.02 to 1.00 (mean 0.36 ± SD 0.31) (Fig. 2, Table 2). The eight species that were only caught on fished areas had an estimated P_N of 100%. An additional eight species had greater than 40% of their populations inside fished areas. However, the majority of species (30) had less than 30% of their population distributed in fished areas. The estimated relative abundance of some species was uncertain due to low detection rates.

3.3 Fishing-induced mortality rate

Estimated fishing impacts were reduced after we accounted for probabilities of capture and escapement. Most species (31) had a mean fishing mortality rate, u, of <10%. Only eight species had an estimated mean u > 30% (Fig. 3); these species were also shown to have experienced the highest impact, based on their distributions (P_N).

3.4 Sustainability assessment

(1) Maximum sustainable fishing mortality rate (u_{msm})

Based on natural mortality, the estimated fishing mortality rate at which a bycatch species can sustain the maximum fishing mortality rate (u_{msm}) ranged from 0.08 to 0.68, with a mean of 0.26 (± SD 0.12) (Table 2; Fig. 3). The majority of species (30) were estimated to capable of sustaining fishing mortality rates of between 20% and 40%. Sixteen species had u_{msm} less than 20%, whereas the remaining five species were estimated to be capable of sustaining fishing mortality rate greater than 40%. The fishing impacts on ten species may have exceeded u_{msm} (Table 2; Fig. 3). These were the same species that had a fishing impact (u) of greater than 45%, except Sphyrna mokarran, which had a very low u_{msm} (u_{msm} = 0.1, whereas u = 0.11 for this species). An additional nine species had an estimated 95% confidence interval (CI) of u that covered the estimated u_{msm} (Fig. 3).

(2) Minimum unsustainable fishing mortality rate (u_{crash})

Five species had an estimated u greater than u_{crash}: Carcharhinus albimarginatus, Orectolobus ornatus, Squatina sp. A, Taeniura meyeni, and Urogymnus asperrimus (Fig. 4). In addition, five species had an estimated 95% CI of u that covered the estimated u_{crash} value: Carcharhinus brevipinna, Carcharhinus leucas, Pristis microdon, Pristis zijsron, and Sphyrna mokarran.

4. Discussion

Assessing the sustainability of diverse trawl fishery bycatch species is a great challenge for researchers. Due to its indiscriminate nature, demersal prawn trawling has the potential to affect the populations of many non-target species (Kennelly, 1995). In tropical fisheries, this problem can be exacerbated by the enormous diversity of species and life histories of animals impacted, including sessile and motile invertebrates, teleosts, elasmobranchs, turtles and vertebrates such as seasnakes (Milton, 2001; Stobutzki et al., 2001a). As a result, in order to assess the ecological sustainability of bycatch species in a fishery, two problems must be resolved. First, the limited data that is generally available on bycatch species is a significant hindrance to assessing a population’s viability under existing fishing regimes, especially for elasmobranchs (Frisk et al., 2001). Second, unlike target species, there is a lack of clear guidelines and performance measures for assessing whether the fishing impacts on bycatch species are being managed at biologically sustainable levels. The SAFE method we propose can help to overcome these problems by using simple data and limited life history information on the species being impacted.

The SAFE method we propose for sustainability assessment of non-target species, has two main components: the fishery impact estimated from simple distributional information, and a sustainability benchmark established from life history traits. Because this framework is similar to the management strategy for data-limited target species, the approach can be directly incorporated into existing fishery management frameworks.

4.1 Quantifying fishery impacts on bycatch populations

The relative abundance of species inside the fished area ($\underline{P_N}$ in eq. 3) is an important parameter for SAFE. We used simple detection-nondetection data to estimate abundance. For many established fisheries, this type of binary data may already have been collected for other purposes, and if more data are needed, the time and cost of collecting detection-nondetection data are lower than for count data. A simulation study demonstrated that estimating abundance from detection-nondetection data has a low bias when a grid is repeatedly surveyed on average three or more occasions, the gear efficiency is ≥ 0.5, and the sampling rate is ≥ 5% (Zhou and Griffiths 2007). Furthermore, model (4) utilizes $\underline{P_N}$, the ratio between the abundance of a species inside fished areas relative to its abundance in the total NPF area, rather than actual abundance $\underline{N_{R,F}}$. Recent simulation results indicate that this relative quantity is less biased than the actual abundance itself (Zhou, unpublished data). Nevertheless, the actual sampling rate from the scientific survey may have been low. The scientific surveys had a nominal sampling rate of about 13%. However, the fishing gears sweep only a small proportion of a grid. Consequently, the actual sampling rate should be less than the nominal sampling rate, which may have contributed to the large uncertainty in the estimated fishery impact on the population of some species.

We used the scientific surveys from 1979 to 2003 to increase spatial coverage and sample sizes. This practice implies that we assume the proportion of a species' population in the fished area with respect to the total area remained unchanged during the entire study period. Two comparative scientific surveys conducted in the Gulf of Carpentaria in the1980s and again in recent years using identical gears and procedures found that abundance of fish species has largely remained the same (S.

Blaber, pers. comm.). If the prawn fishery reduces the population of a bycatch species inside the fished area and the population in the unfished area does not move into the fished area, then this assumption could be violated. This would most likely result in the relative abundance inside the fished area in recent years to be overestimated rather than be underestimated because the method uses data collected over a long period to derive the abundance ratio. Consequently, the recent trawl impact by the NPF on a population may be overestimated.

We encountered data limitation problems because many bycatch specieshave low economic value and therefore, poorly studied. Without actual catch and escapement rates for bycatch species, data from previous studies (e.g. Blaber at al., 1990; Brewer et al., 2004) is required to provide estimates for closely related species, or those species that generally have similar ecomorphotypic characteristics as well-studied species. If the values used from other studies are incorrect, this will obviously bias the predicted fishery impact. However, given the common data limitations for individual bycatch species, we feel our approach of using values for similar species and expert opinion is adequate, although we see great value in undertaking further experimental work to increase the confidence in catch rate and escapement probability of species that have been little studied.

4.2 Management reference points for sustainability of bycatch

Ecosystem-based fishery management is currently neither well defined nor understood (Brodziak and Link, 2002). Although concepts such as biodiversity, ecosystem integrity and ecosystem function are frequently cited in management policy, they are difficult to interpret at an operational level (Garcia and Staples, 2000; Murawski, 2000; Mace, 2001). However, sustainability is one of the generally recognized objectives for ecosystem-based fishery management. We propose two risk reference points for bycatch species: risk of overfishing (F_{msm}) and risk of the population becoming unsustainable (F_{crash}). Our method for estimating the fishing mortality rate is simple, but fishing mortality rate alone does not indicate whether a stock is sustainable or not under current fishing pressure. A metric determining the optimal or threshold fishing mortality rate is required for evaluating the biological consequences of the estimated impact. Unfortunately, from a traditional stock assessment point of view, such benchmarks require a substantial understanding of population dynamics. For data-poor bycatch species, we recommend using reference points based on basic life history traits, as life history strategy generally has a close relationship with the resilience of a population exposed to fishing pressure (Jennings et al., 1999; Rochet, 2000; Frisk et al., 2001).

For target fish species, natural mortality rate has been used widely for the optimal fishing mortality rate since the 1960s (Alverson and Pereyra, 1969; Gulland, 1970). Research has shown that instantaneous natural mortality rate (M) is a reasonable surrogate for F_{msy} for some stocks, although it can be too high for other stocks (Francis, 1974; Deriso, 1982; Garcia and Csirke, 1989). For example, Clark (1991) showed that from calculations made with a range of life history parameter values typical of demersal fish and using a range of realistic spawner–recruit relationships, the optimal harvest rate is often close to the natural mortality rate M. There are also numerous examples of demersal stocks that have sustained fishing mortality rates well above $2M$ for long periods (Clark, 1991). On the other hand, for stocks with little or no growth data, a maximum fishing mortality rate of 80% of the natural mortality rate has been suggested as a precautionary approach (Thompson, 1993). Walters and Martell (2002) suggested that any assessment that results in $F_{opt} > 0.5M$ must be carefully justified.

Another issue of using natural mortality is the uncertainty around this parameter itself. Because we gathered most values of M from other studies and the literature, where often no variance was provided, we had difficulty including uncertainty around this parameter. Uncertainty of M and the resulting reference points can be incorporated in the assessment when more data are available.

Of the two risk reference points, we consider that optimal fishing mortality rate may not be the most appropriate management goal for bycatch species. Ecological sustainability is likely to be more acceptable to multiple users (Garcia and Staples, 2000), whereby the fishery does not aim to maximise the yield of a bycatch species, but ensures that its fishing impacts does not drive the population to very low levels. A stock is technically overfished when its biomass is lower than a biomass that produces maximum sustainable yield or is fished at a rate where yield-per-recruit is lower than the maximum level. However, such a stock is not necessarily unsustainable (Hilborn, 2002).

Elasmobranchs are among the species most vulnerable to overfishing, whether as target or bycatch, mainly due to their low capacity to recover once depleted (Stevens, 1997; Walker, 1998; Baum et al., 2003). Consequently their management should be precautionary, especially given the uncertainty in biological parameters, and the fact that bycatch species are rarely recorded in fishery logbooks to provide an indication of their long-term population viability. Smith et al., (1998) used total mortality Z = 2M to assess the rebound potential of 26 species of Pacific sharks and recommended that populations should not be fished at mortalities greater than the intrinsic rate of increase at a mortality level chosen as twice the natural mortality rate, which are generally very low, ranging from 0.017 to 0.136. Walker (1998), who recommended using MSY as a management reference point, showed the fishing mortality rate required to achieve MSY for a temperate shark, Mustelus antarcticus, is between 12 and 15%, but can be as low as 5-6% for other temperate species such as Galeorhinus galeus. However, these management recommendations were aimed at optimising economic profits from commercially harvested species. Tropical elasmobranch bycatch species, which have generally higher production rates than temperate species (Smith et al., 1998), could sustain a higher fishing mortality if the management objective is to maintain ecological sustainability rather than fishery profits.

4.3 At-risk elasmobranch bycatch species

The results from SAFE indicate that out of the 51 species, 19 species (when uncertainty is taken into account) would be potentially at risk of overfishing and 9 species (with uncertainty) at potential risk of being unsustainable. In particular, five species have point estimates of $u > u_{crash}$ (Carcharhinus albimarginatus, Orectolobus ornatus, Squatina sp. A, Taeniura meyeni, and Urogymnus asperrimus). These species were rarely recorded, and exclusively caught within the fished region, meaning that 100% of their population could be exposed to trawling in the NPF, although they are also reported to occur outside the fishery (Last and Stevens, 1994). Possibly, they were recorded only within the fished area because those areas were of the focus of the surveys. Among these five species, Carcharhinus albimarginatus is a widely distributed pelagic species and is rarely caught in prawn trawls; the fishing impact on this species is likely to be overestimated. The other four species are relatively slow-moving benthic species, which are likely to have a high catchability by a trawl; hence they were each assigned the highest relative catch rate of 1. Since these rays are also typically relatively small (< 1 m disc width), they were each assigned a low probability of escaping through TEDs because they are small enough to pass through the spaces between the TED bars (Brewer et al., 2006). Also, the estimated fishing mortality rates of these species contain high uncertainty. Considering uncertainty associated with the estimated reference points, which is not included in this paper, we can only conclude that these species are potentially at risk of being unsustainable. Several other species have high distributional overlaps with the fishery, such as Carcharhinus leucas, Rhincobatus typus and Himantura jenkensii (100% overlap). However, these species are considered to be not at risk because their catch rates are relatively low (47% for C. leucas) or their escapement is high (100% and 69% for Rhincobatus typus and H. jenkensii, respectively).

This study assessed the impact of the NPF on bycatch species but did not consider the impact of other fisheries in the region that target elasmobranchs or catch them as bycatch. Populations of sawfishes (Pristidae) and other elasmobranchs may be sustainable while being exposed to the impact of the NPF alone. However, their populations could potentially be at risk from the cumulative impacts of the state-regulated and illegal gillnet fisheries in the region. As a result, there is an urgent need to assess the cumulative impacts of fisheries on elasmobranch populations. The estimated fishing impacts in the present study are additive, so our SAFE method has the potential to study the cumulative impacts from fisheries and possibly other anthropogenic activities.

In conclusion, our SAFE is a novel method for quantitative ecological risk assessment on fishing effects. This method would be most effective for fisheries management when used in conjunction with an ongoing monitoring program. Because the method can quantify the fishery impact on hundreds of species, it may serve as a ‘filtering’ mechanism, identifying species potentially at risk, which can become candidates for monitoring. Ongoing monitoring of these species would provide additional data on the population and allow more sophisticated stock assessment to be undertaken in future.

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Table 1. Total observed detections and the number of grids where each species was recorded. Sample size: 4,441 for fished areas and 1,394 for unfished areas. Total grids surveyed: 233 for fished areas and 691 for unfished areas.

	Fished area		Unfished area	
Species	Grids	Detections	Grids	Detections
Aetobatus narinari	13	16	7	9
Aetomylaeus nichofii	56	89	28	36
Aetomylaeus vespertilio	9	9	1	1
Anoxypristis cuspidata	37	54	13	13
Atelomycterus fasciatus	3	4	2	2
Carcharhinus albimarginatus	1	1	–	–
Carcharhinus amboinensis	1	1	6	6
Carcharhinus brevipinna	1	1	1	1
Carcharhinus dussumieri	154	942	315	433
Carcharhinus fitzroyensis	2	2	1	1
Carcharhinus leucas	1	1	–	–
Carcharhinus limbatus	45	141	18	24
Carcharhinus macloti	9	10	10	12
Carcharhinus sorrah	46	68	26	31
Carcharhinus tilstoni	108	634	52	61
Chiloscyllium punctatum	101	499	23	26
Dasyatis annotata	58	510	47	52
Dasyatis brevicaudata	4	4	18	18
Dasyatis thetidis	5	8	16	16
Dasyatis kuhlii	47	108	98	110
Dasyatis leylandi	88	436	46	75
Eusphyra blochii	9	11	3	3
Galeocerdo cuvier	4	7	4	4
Gymnura australis	149	901	179	221
Hemigaleus microstoma	122	488	135	152
Hemipristis elongata	20	22	20	20
Himantura fai	2	3	7	10
Himantura granulata	8	10	9	10
Himantura jenkinsii	5	7	–	–
Himantura sp. A	8	12	1	1
Himantura toshi	156	1063	124	182
Himantura uarnak	28	41	71	77
Himantura undulata	44	71	15	16
Narcine westraliensis	1	1	7	13
Nebrius ferrugineus	6	9	2	2
Negaprion acutidens	4	4	8	9
Orectolobus ornatus	1	1	–	–
Pastinachus sephen	61	97	12	12
Pristis microdon	1	2	2	3
Pristis zijsron	6	7	5	5
Rhina ancylostoma	29	34	7	7
Rhinobatos typus	10	11	–	–
Rhizoprionodon acutus	91	468	113	162
Rhizoprionodon taylori	10	14	12	12
Rhynchobatus djiddensis	146	787	123	135
Sphyrna lewini	40	71	35	35
Sphyrna mokarran	7	7	5	5
Squatina sp. A	1	1	–	–
Stegastoma fasciatum	54	126	29	29
Taeniura meyeni	4	4	–	–
Urogymnus asperrimus	4	4	–	–

Table 2. Estimated trawling impact on bycatch species' abundance distribution (P_N), probabilities of capture q and escapement E used to derive fishing mortality rate u, and comparison with reference points u_{msm} and u_{crash}. Numbers underlined are actual measurements from field studies.

Species	P_N	SE[P_N]	q	E	u	SE[u]	u_{msm}	u_{crash}
Aetobatus narinari	0.33	0.19	1.00	1.00	0.00	0.00	0.17	0.32
Aetomylaeus nichofii	0.32	0.06	1.00	0.00	0.32	0.06	0.32	0.53
Aetomylaeus vespertilio	0.67	0.27	1.00	1.00	0.00	0.00	0.28	0.48
Anoxypristis cuspidata	0.41	0.00	1.00	0.73	0.11	0.05	0.14	0.24
Atelomycterus fasciatus	0.08	0.11	1.00	0.00	0.08	0.11	0.33	0.55
Carcharhinus albimarginatus	1.00	0.00	0.47	0.00	0.47	0.16	0.17	0.32
Carcharhinus amboinensis	0.07	0.20	0.47	0.00	0.03	0.10	0.19	0.34
Carcharhinus brevipinna	0.68	0.36	0.47	0.00	0.32	0.20	0.29	0.49
Carcharhinus dussumieri	0.14	0.00	0.47	0.00	0.07	0.02	0.37	0.60
Carcharhinus fitzroyensis	0.15	0.27	0.47	0.00	0.07	0.13	0.34	0.56
Carcharhinus leucas	1.00	0.01	0.47	0.00	0.47	0.16	0.28	0.48
Carcharhinus limbatus	0.23	0.02	0.47	0.00	0.11	0.04	0.32	0.54
Carcharhinus macloti	0.50	0.16	0.47	0.00	0.23	0.11	0.26	0.46
Carcharhinus sorrah	0.24	0.00	0.47	0.00	0.11	0.04	0.45	0.69
Carcharhinus tilstoni	0.14	0.01	0.47	0.00	0.07	0.02	0.20	0.36
Chiloscyllium punctatum	0.37	0.02	1.00	0.27	0.27	0.02	0.37	0.61
Dasyatis annotata	0.11	0.01	0.83	0.40	0.05	0.01	0.39	0.62
Dasyatis brevicaudata	0.19	0.01	0.83	0.23	0.12	0.02	0.14	0.26
Dasyatis kuhlii	0.02	0.01	0.83	0.23	0.02	0.01	0.14	0.26
Dasyatis leylandi	0.07	0.01	0.83	0.00	0.05	0.01	0.36	0.59
Dasyatis sp. A	0.11	0.01	0.83	0.30	0.06	0.01	0.34	0.56
Eusphyra blochii	0.40	0.16	0.47	0.00	0.19	0.10	0.28	0.48
Galeocerdo cuvier	0.03	0.04	0.47	0.00	0.01	0.02	0.09	0.13
Gymnura australis	0.19	0.00	0.60	0.00	0.11	0.03	0.23	0.41
Hemigaleus microstoma	0.14	0.02	0.47	0.00	0.07	0.02	0.36	0.59
Hemipristis elongata	0.36	0.08	0.47	0.00	0.17	0.07	0.23	0.41
Himantura fai	0.03	0.04	0.07	0.69	0.00	0.00	0.28	0.48
Himantura granulata	0.25	0.14	0.07	0.42	0.01	0.01	0.31	0.52
Himantura jenkinsii	1.00	0.00	0.07	0.69	0.02	0.03	0.30	0.51
Himantura sp. A	0.48	0.28	0.07	0.69	0.01	0.01	0.43	0.68
Himantura toshi	0.22	0.00	0.07	0.42	0.01	0.01	0.33	0.55
Himantura uarnak	0.14	0.05	0.07	1.00	0.00	0.00	0.27	0.46
Himantura undulata	0.22	0.06	0.07	0.91	0.00	0.00	0.15	0.27
Narcine westraliensis	0.07	0.01	1.00	0.39	0.04	0.01	0.68	0.90
Nebrius ferrugineus	0.06	0.13	1.00	1.00	0.00	0.00	0.25	0.44
Negaprion acutidens	0.26	0.03	0.47	0.00	0.12	0.04	0.15	0.27
Orectolobus ornatus	1.00	0.00	1.00	0.39	0.61	0.11	0.21	0.37
Pastinachus sephen	0.30	0.04	1.00	0.98	0.01	0.01	0.26	0.45
Pristis microdon	0.23	0.26	1.00	0.73	0.06	0.07	0.10	0.18
Pristis zijsron	0.31	0.18	1.00	0.73	0.08	0.06	0.10	0.18
Rhina ancylostoma	0.56	0.11	1.00	1.00	0.00	0.00	0.16	0.29
Rhinobatos typus	1.00	0.00	1.00	1.00	0.00	0.00	0.08	0.15
Rhizoprionodon acutus	0.15	0.03	0.08	0.00	0.01	0.01	0.44	0.68
Rhizoprionodon taylori	0.46	0.12	0.08	0.20	0.03	0.03	0.45	0.70
Rhynchobatus djiddensis	0.18	0.00	0.04	0.39	0.00	0.01	0.14	0.26
Sphyrna lewini	0.26	0.04	0.47	0.00	0.12	0.05	0.12	0.23
Sphyrna mokarran	0.24	0.09	0.47	0.00	0.11	0.06	0.10	0.18
Squatina sp. A	0.89	0.04	1.00	0.39	0.54	0.10	0.23	0.41
Stegastoma fasciatum	0.14	0.02	1.00	1.00	0.00	0.00	0.23	0.41
Taeniura meyeni	1.00	0.00	1.00	0.42	0.58	0.11	0.29	0.49
Urogymnus asperrimus	1.00	0.00	1.00	0.42	0.58	0.11	0.33	0.55

Figure 1. Distribution of samples taken in scientific surveys in NPF from 1979 to 2003 (+) and grids where tiger prawn fishing effort was greater than 5 boat-days from 1999-2003 (■). The NPF managed area is stratified into 5 bioregions based on the bioregions of IMCRA (1998).

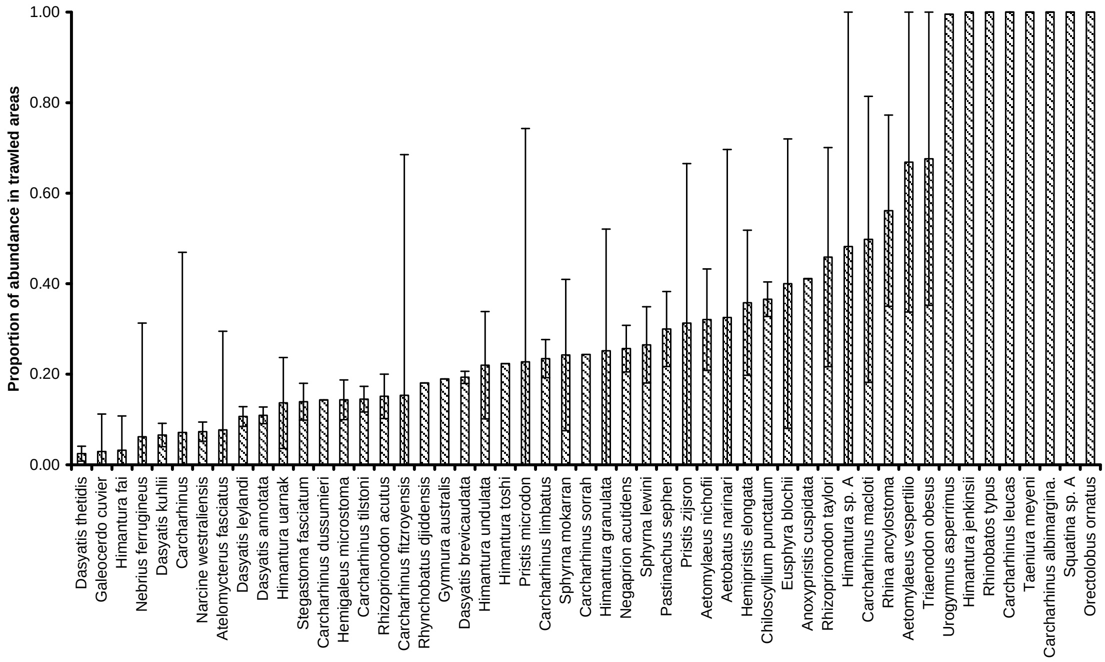

Figure 2. Estimated proportion of abundance within fished areas and 95% confidence intervals for 51 bycatch species.

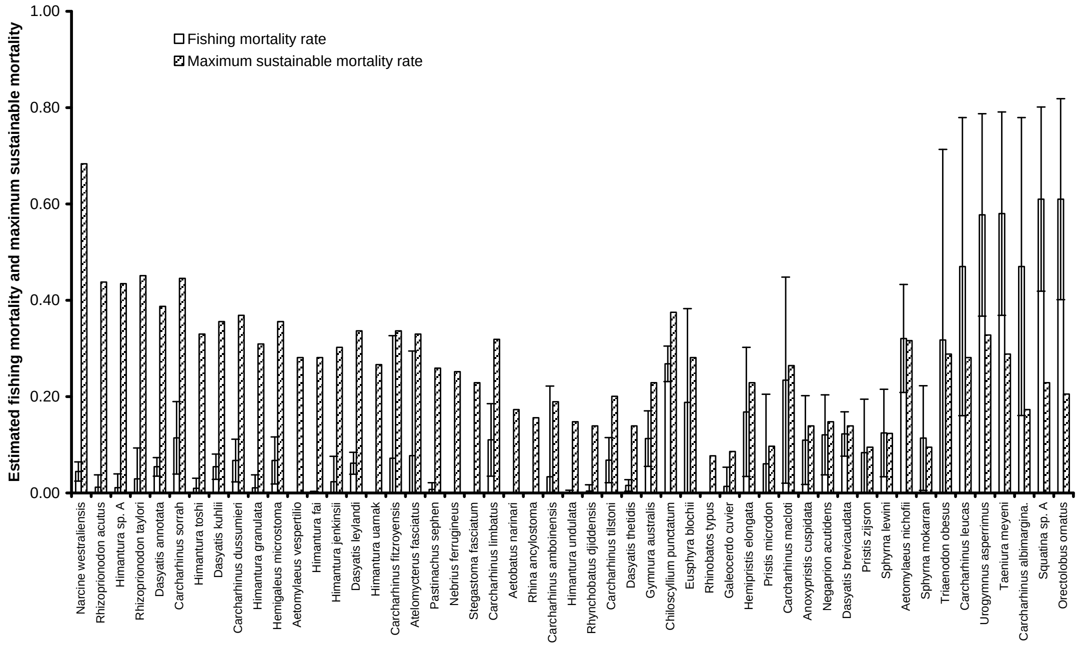

Figure 3. Comparison between estimated fishing mortality rates (u) (±95% confidence intervals) from prawn trawling and the maximum sustainable fishing mortality rates $\underline{u}_{msm}$ for the 51 bycatch elasmobranchs.

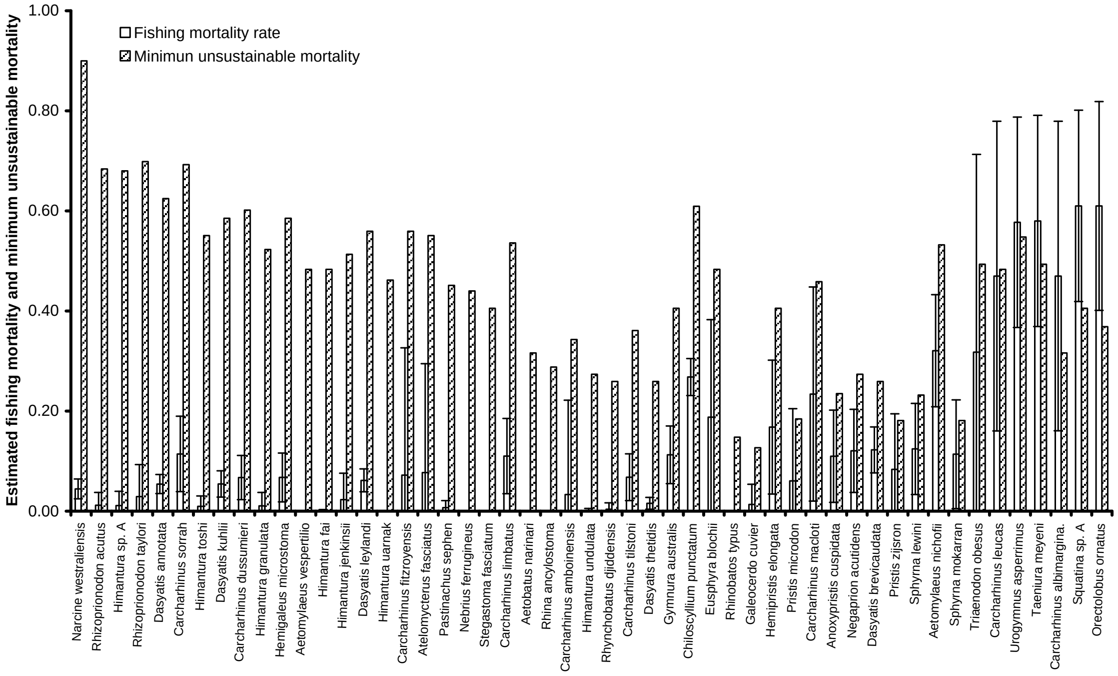

Figure 4. Comparison between estimated fishing mortality (u) from prawn trawling and the minimum unsustainable mortality rate (u_{crash}).

APPENDIX 2. SUSTAINABILITY ASSESSMENT FOR FISHING EFFECTS (SAFE) ON HIGHLY DIVERSE AND DATA-LIMITED FISH BYCATCH IN A TROPICAL AUSTRALIAN TRAWL FISHERY

Shijie Zhou*, Shane P. Griffiths, and Margaret Miller

Abstract

We uses a new sustainability assessment for fishing effects (SAFE) method to assess the ecological sustainability of 456 teleost bycatch species in Australia's Northern Prawn Fishery. This method can quantify the risk from fishing for large numbers of species with limited data. First, we estimated the fishing mortality rate of each species based on its spatial distribution (estimated from detection-nondetection data) and catch rate. Second, we assessed the sustainability of each species by using two biological reference points based on life history parameters: maximum sustainable fishing mortality and minimum unsustainable fishing mortality. The point estimates indicated that only two species (but 12 when uncertainty was included) had estimated fishing mortality rates greater than fishing mortality rate corresponding to the maximum sustainable fishing mortality. These two species also had their upper 95% confidence intervals (but not their point estimates) greater than their minimum unsustainable fishing mortality rates. That the large number of species is sustainable can be attributed mainly to their wide distributions into unfished areas, low catch rates within the fished area, and short life spans (high sustainability). This study demonstrates how SAFE may be a cost-effective quantitative assessment method to support ecosystem-based fishery management objectives.

Key words: ecological risk assessment, distribution, fishing mortality, prawn fisheries, sustainability, teleost bycatch

Introduction

The management paradigm of marine fisheries has traditionally been dominated by target species. However, over the past decade the concept of ecosystem-based fishery management (EBFM) has begun to infiltrate the management approaches for many fisheries worldwide. Generally, EBFM is considered as a holistic approach to ensuring the sustainability of the species, communities and habitats that support fisheries (Larkin 1996). Such broad ecosystem objectives may be attractive fishery policies, but they do not provide a practical means by which to manage the balance between maintaining ecosystem integrity and function, and optimising fishery yields (Link 2002).

Two conceptually different approaches may be used to implement EBFM: a single species approach that assesses species of interest in isolation, and a holistic approach that uses the entire ecosystem (Link 2002). Between these two extremes, there is an increasing development of multispecies models with different levels of complexities (Hollowed et al. 2000). Due to the enormous complexity in understanding ecological relationships, few practical tools available can assess the sustainability of all species impacted by fisheries. Quantitative ecosystem models, including Atlantis (Fulton et al., 2004) and Ecopath (Christensen and Pauly, 1992) have great potential in the EBFM arena, but they are data- and labour-intensive and the results from these models are yet to be validated. In contrast, traditional quantitative single species models are more useful for assessing sustainability of individual species, via the use of well-established reference points, but they are data intensive and have limited application to data-poor, non-target species. As an alternative, some ecological risk assessment approaches can be applied to all species and may provide a critical and practical first step towards achieving ecosystem-based fishery management.

In recent years, a number of ecological risk assessment approaches have been developed, consisting of qualitative (Fletcher et al. 2005; Astles et al. 2006; Hobday et al. 2006) and semi-quantitative attribute-based models (Milton 2001; Stobutzki et al. 2001b; Cheung et al. 2004; Walker 2004;

Hobday et al. 2006), which have primarily been designed for data-limited fisheries. Unfortunately, these methods provide only a relative ranking of risk for each species, and cannot quantify the fishery impact on individual species, or assess whether the species is at risk or not from fishing activities. Furthermore, some of these methods are not sensitive to changes in size or species selectivity of a fishery and can fail to reflect even the most obvious change in species sustainability due to management intervention (Griffiths et al. 2006). As a result, more quantitative approaches that can also use limited data and assess absolute risk are desirable for fishery managers.

Two recent studies addressed this issue for non-target species. Pope et al. (2000) performed length-cohort analysis on catch-at-length data, and used a simple swept-area method to estimate the current overall fishing mortality rate for two non-target species in the North Sea. They then assessed the capacity of these populations to withstand the given fishing mortality, by estimating the fishing mortality that would reduce the spawning-stock biomass per recruit to an arbitrary, but supposedly sustainable, level (5%) of its unfished biomass. Their methods for estimating fishing mortality rates are useful for fisheries where detailed catch-at-length data are available, or the density of non-target species in fished and unfished areas can be assumed equal, and catch rates along a trawl track are 100%. Their method for estimating the capacity of the population to withstand fishing mortality requires both life history parameters and fishery information, including natural mortality, growth rate, length infinity, age at recruitment, age at capture, fishing mortality rate. However, some of these parameters may be difficult to obtain, especially in tropical fisheries where hundreds of bycatch species may be impacted (Stobutzki et al. 2001). In addition, setting an arbitrary value of 5% of the virgin spawners-per-recruit as a reference point needs further deliberation.

Zhou and Griffiths (in press) developed a rapid quantitative method, Sustainability Assessment for Fishing Effects (SAFE), to assess the sustainability of 51 data-poor elasmobranch bycatch species in a tropical Australian trawl fishery. They used binary detection-nondetection data to estimate fishing mortality rates, and established two reference points based on one or two life history parameters (natural mortality and growth rate). In this paper, we extend the SAFE method to assess the sustainability of 456 teleost species caught as prawn trawl bycatch in one of Australia's largest and most valuable fisheries, the Northern Prawn Fishery (NPF). We define two types of risks: risk of overfishing and risk of extinction in the long-term.

Materials and methods

In order to assess fishery impacts over the entire NPF managed area we used data collected from over 70 scientific voyages between 1979 and 2003. Details of these collections are described by Zhou and Griffiths (in press). Fourteen gear types were used in the surveys, each having different species and size selectivity. These gears included a benthic sled, Engels demersal fish trawl, Engels trawl fitted with codend cover, Florida Flyer benthic trawl, Florida Flyer trawl with codend cover, Florida Flyer trawl with a bycatch reduction device (BRD), Florida Flyer trawl fitted with BRD and codend cover, Florida Flyer trawl with a turtle exclusion device (TED), Frank and Bryce fish trawl, modified semi-pelagic Frank and Bryce trawl, Julie Ann net, modified semi-pelagic Julie Ann net, twin Florida flyer trawl with Texas drop-chain rig, and Yankee Doodle 10 Fathom Prawn net. Although different fishing gears were used in these surveys, the most frequently used gears were the Florida Flyer prawn trawl, the Frank and Bryce fish trawl, and Yankee Doodle 10 fathom prawn net. We divided the NPF managed area into 6,963 sampling units of 6 by 6 nautical mile grids. Sampling occurred in 1,380 of these grids where a total of 7,095 samples were taken. The distribution of species commonly caught as bycatch varied spatially within the NPF (e.g. Blaber et al. 1990, 1994, Stobutzki et al. 2001a; Tonks et al. In press). Therefore, we stratified the NPF-managed area into five bioregions based on established bioregions for fishes (IMCRA, 1999) and expert opinion.

In the NPF, the tiger prawn (*P. semisulcatus* and *P. esculentus*) fishery extends from August to November and trawling takes place only during the night. Because the fishery targets dispersed tiger prawns and uses long trawl hours (3-4 h), the bycatch is often caught in large volumes (>300 kg/trawl;

Stobutzki et al. 2001a). In this paper, we defined the fished area as any grid where the total fishing effort recorded from commercial logbook data was ≥ 5 boat-days in any one year between 1999 and 2003. Five boat-days of fishing effort is equivalent to about 10% of sea floor within the grid being systematically swept by prawn trawls in 5 years, assuming trawling occurs for 12.3 hours per day (Rawlinson 2003) at a speed of 3.24 knots (Bishop 2003) with a headrope length of 14 fathoms and a 0.66 spread ratio (Bishop and Sterling 1999). Because trawl tracks often overlap, the actual impact is probably less than 10% of sea floor (Stobutzki and Pitcher 1999, Dichmont et al. 2001).

Fishing induced mortality rate

The fishing-induced mortality rate of individual species was estimated from: 1) their relative abundance within trawled areas compared to the entire NPF managed area, 2) the estimated proportion of fish in the path of the trawl that enters the trawl opening (termed "catch-rate"), 3) the proportion of fish escaping through a Turtle Excluder Device (TED) or a Bycatch Reduction Device (BRD) after entering the trawl opening (termed "escapement rate"), and 4) the proportion of landed fish surviving when returned to the sea (termed "post-capture survival rate"). This can be represented as:

$$u = \frac{N_1}{N_1 + N_0} q(1 - \mathrm{E})(1 - s) \qquad (1)$$

where N_1 and N_0 are is the abundance of a species inside and outside trawl areas, respectively; q is the catch rate, E is the escapement rate, and s is the post-capture survival rate. This formula implies that we simplified the fishing process to uniformly sweep a grid once a year. From commercial logbook data, we estimated that the average fishing effort in the fished areas could systematically sweep the seabed 0.89 times/year in fished area (approximately 95% CI 0.80 – 0.99 times/year).

The key component of Eq. (1) is the relative abundance exposed to trawling, $N_1/(N_1 + N_0)$. We used the model of Zhou and Griffiths (2007) to estimate N_1 and N_0 from detection-nondetection data. The model assumes that after stratification of the NPF into bioregions, individuals were randomly distributed within fished and unfished areas within each bioregion, and fish density differed between fished and unfished areas within each bioregion. The probability that a surveyed grid is occupied by a particular species is directly related to the total abundance of the species in the fished or unfished area in the bioregion. Because the survey data were collected over a 24-year period, we assumed the relative abundance of each bycatch species between the fished and unfished areas remained constant during the study period. We also assumed that the probability of capture of a particular species remained constant across all surveyed grids within each bioregion, but was specific to the fishing gear used. We used a logistic model to incorporate gear-specific catchability into the model as described by Zhou and Griffiths (in press).

We obtained species-specific catch rates using one of following methods: i) from field studies (Pitcher et al. 2002); ii) based on related species in the same genus for which measurements were made, since closely related species are likely to have similar vulnerability to capture; iii) based on values estimated by Blaber et al. (1990) for the same species; and iv) based on values of Blaber et al. (1990) but for species having similar vertical distribution, size, and locomotory behaviour, or “ecomorphotypes” (Compagno 1990; Bax et al. 1999). The first method was our preferred choice because estimates came from actual field studies. However, the data are reported as a “relative catch rate” estimated from catch-per-hectare data for each gear type for each species obtained from deployment of multiple sampling gears at the same sites (Pitcher et al. 200). If the prawn-trawl yielded a lower catch-per-hectare than either the epi-benthic sled or the fish trawl, then the relative catch rate was the fraction: prawn trawl catch/ha over highest catch/ha. If the prawn-trawl yielded the highest catch per hectare then the relative catch rate was 1, even though the actual catch rate may in reality have been less than 1. For these reasons, the relative catch rate applied herein should be considered a maximum.

Brewer et al. (2006) found that the compulsory use of TEDs and BRDs in the NPF reduced the teleost bycatch by 8%. They did not measure species-specific escapement rates, which are likely to differ between species. Therefore, we were conservative and assumed escapement rates of zero for all species. For the same reason we assumed a post-capture survival rate of zero for all species. These treatments may contribute to an overestimate in fishing impacts on individual species, but in the absence of empirical data we chose to be conservative.

Uncertainty assessment

Quantifying uncertainty is important for assessing risk. We estimated variances for the parameters *N* and *q*. Approximate standard errors (*SE*) of *N* were derived from the square roots of the diagonal elements of the covariance matrix of the parameter estimates. This is the same as the inverse of the Hessian matrix (the matrix of second derivatives) of the likelihood. Variances of *q* were calculated from binomial distributions, assuming capture in trawls is a binomial process, i.e., q~Bin(n, E[q]), where *n* is the sample size from field experiments or assumed samples and E[*q*] is the expected probability of capture estimated from field studies or the literature. Variance of fishing mortality rate *u* was obtained from the variance of *N* and *q* by the delta method of Zhou (2002).

Management reference points

Theoretically, a fish population can be sustainable, that is, maintaining a certain population size into the long term, at numerous alternative states. However, we were particularly interested in three states: i) pristine state where the population is at its carrying capacity and human impact on the population are minimal, ii) population size that supports maximum productivity, and iii) the point where the fishing mortality rate equals the intrinsic population growth rate. Since one of the primary goals of EBFM is to support sustainable exploitation by fisheries, restoring the ecosystem back to the pristine state is often not a valid objective. The two other states relate to classical biological reference points used in the management of target species: the maximum sustainable fishing mortality (F_{msm}) as a target reference point, and the fishing mortality that may eventually drive a stock to collapse as a limit reference point to avoid (F_{crash}). These are the two biological reference points proposed by Zhou and Griffiths (in press) for managing non-target, and low economic value elasmobranch species. The first reference point, the maximum sustainable fishing mortality (MSM), is equivalent to MSY and a fishing mortality rate (u_{msm}) corresponding to MSM. One of the methods that define the reference point is to set u_{msm} = 1 - exp(-F_{msm}) = 1 – exp(-M), where F_{msm} is the instantaneous fishing mortality rate and M is natural mortality rate. However, setting $F_{msm} = M$ is not considered conservative, especially for species of high natural mortality (Garcia et al. 1989; Thompson 1993; Quinn and Deriso 1999). Using this generic approach, the estimated u_{msm} could be greater than 0.99 for some short-lived tropical species having high natural mortality rates. If this total annual fishing mortality rate is spread out over the entire year, species that have a short life span and continues to spawn over extended periods may still be sustainable under such a high fishing mortality rate. However, if the fishing season is relatively short, a pulse of high fishing mortality rate may trigger overfishing or render the population unsustainable. Therefore, in this paper, we set $F_{msm} = \omega M$, where the scaling parameter ω is a function of M of between 0.5 and 1:

$$\omega = 1 - 0.5\frac{M - M_{\min}}{M_{\max} - M_{\min}}. \qquad (2)$$

In this equation, M_{min} and M_{max} are the minimum and maximum instantaneous natural mortalities of all species in the study.

The second reference point or threshold, u_{crash}, is the minimum fishing mortality rate that leads to unsustainable stock in the long-term. The terms of “sustainable ecosystem (or fishery)” and “sustainability” have been widely used in resource management. However, these terms are often not clearly defined or ill-defined in many cases, and may cause confusing between technical application and policy level usage. For example, in fishery sustainability is commonly interpreted as equivalent to the concept of Maximum Sustainable Yield (Garcia and Staples 2000). However, sustainability is generally defined as a characteristic of a process or state that can be maintained at a *certain level* indefinitely (for example, see definition in encyclopaedia such as http://en.wikipedia.org). The term of "sustainable" refers to systems to be productive indefinitely. In fishery, as long as a stock can produce surplus production at a certain level indefinitely, that stock should be regarded as sustainable. Although a stock can be sustainable at countless levels, from the deterministic process point of view, there is only one theoretical level when the stock can support a maximum sustainable surplus mortality and only one level when a stock can sustain a minimum unsustainable mortality. The latter expressed in fishing mortality rate is what we defined as u_{crash}. According to the Graham-Schaefer production model (Fletcher 197.

8; Hilborn and Walters 1992; Quinn and Deriso 1999), $F_{crash} = 2F_{msm}$, i.e., $u_{crash} = 1 - \exp(-2F_{msm})$.

Natural mortality (M) was derived from the literature or estimated using the empirical equations:

i) $\ln(M) = -0.0152 - 0.279\ln(L_\infty) + 0.6543\ln(k) + 0.4634\ln(T)$ (Pauly 1980);

ii) $M = 10^{0.566-0.718\ln(L_\infty)} + 0.02T$ (www.fishbase.org) and

iii) M = 1.6 k (Jensen, 1996)

In these equations, k and L_∞ are the von Bertalanffy growth parameters, and T = average water temperature (in this case 28 °C).

Results

Spatial distribution of bycatch species

Of the 456 teleost species recorded as trawl bycatch in the NPF, six and 94 species were caught only in fished and unfished areas, respectively. Except for the six species only caught in the fished area, no species had more than 50% of their population recorded inside the fished area (Fig. 1). Most species had less than 30% of their populations recorded inside the fished area. However, considering that only

6% of the entire NPF managed area is actually trawled, the distribution of most species occurs outside of fished areas even though their densities may be higher inside fished areas.

Fishing mortality

Prawn trawling does not catch all fish within the path of the trawl, due to their vertical distribution in the water column and gear avoidance. The estimated fishing mortality rates ranged from 0 to 0.43 with a mean of 0.05 (± SD 0.07). Nearly half (48%) of the species had fishing mortality rates of ≤ 0.03, while 95% of species had fishing mortality rate ≤0.20 (Fig. 2). Typically, the higher the proportion of a species' population distributed in the fished area, the higher the fishing mortality rate.

Reference points for assessing species sustainability

(1) Fishing mortality rate corresponding to maximum sustainable fishing mortality (u_{msm})

Based on natural mortality, the estimated fishing mortality rate at which a bycatch teleost species can sustain maximum sustainable mortality (u_{msm}) ranged from 0.10 to 0.93, with a mean of 0.55 (*SD* = 0.22; Table 1 and Fig. 3). Only a few species had u_{msm} less than 0.20 or greater that 0.90. The estimated fishing mortality for only two species, *Dendrochirus brachypterus* and *Scorpaenopsis venosa*, exceeded u_{msm} (Table 1, Fig. 3). If uncertainty in estimated fishing mortality rate is taken into account, the 95% confidence intervals of u for 21 species exceeded u_{msm} (Fig. 3).

(2) Minimum unsustainable fishing mortality rate (u_{crash})

The estimated minimum unsustainable fishing mortality rate ranged from 0.19 to 0.99 (mean 0.75 ± SD 0.20) for the 456 teleost species. No species had a mean estimated fishing mortality rate greater than u_{crash}. However, if uncertainty in the estimated fishing mortality rate is considered, the upper 95% CI for five species exceeded u_{crash} (Table 1, Fig. 4). These were *Dendrochirus brachypterus*, *Hemiramphus robustus*, *Lutjanus rufolineatus*, *Parascolopsis tosensis*, and *Scorpaenopsis venosa*.

Discussion

The estimated fishing mortality rates for 21 species of teleost bycatch species exceeded the u_{msm} biological reference point when uncertainty in estimates were taken into account. After close examination of these species, we determined that nine species should be excluded from the list for a number of reasons. Catch rates of 100% from field studies by Pitcher et al. (2002) appeared to be overestimated for six species: *Cottapistus cottoides*, *Lepidotrigla argus*, *Onigocia spinosa*, *Parupeneus barberinoides*, *Richardsonichthys leucogaster*, and *Torquigener hicksi*. These fish have small body sizes (maximum length < 30 cm). When we used catch rate values estimated by Blaber et al. (1990), our estimated fishing mortality rates plus 95% CI did not exceed u_{msm} for these species. An additional three species were included in the high-risk species table because of extremely high uncertainty in their estimated fishing mortality rates. *Hemiramphus robustus* (Robust garfish) had a point estimate of u = 0.01. However, since it was captured in only a very small number of trawls, the upper 95% confidence interval was 100%. This species is a small pelagic fish distributed mainly in inshore regions and therefore its capture in trawls was infrequent, probably only occurring during the net deployment and retrieval periods. As a result, it is unlikely that this species was truly at risk of overfishing. *Parascolopsis tosensis* and *Lutjanus rufolineatus* had point estimates of u = 0.01 and 0.00, respectively. They are both small demersal species and were also caught in low numbers in the surveys. The large uncertainty gives the false impression that the fishery had a high impact on these species. Published occurrence records indicate that *Parascolopsis tosensis* is only distributed outside the NPF, mainly along the Great Barrier Reef on the east coast of Australia. Similarly, *Lutjanus rufolineatus* is primarily restricted to the north-western Australia, outside of the NPF managed area. Therefore, these two species were either misidentified or represent rare specimens well outside their normal geographic distribution. For the same reasons, three species (*Hemiramphus robustus*, *Lutjanus rufolineatus*, *Parascolopsis tosensis*) that have estimated fishing mortality + 95% CI exceeding the u_{crash} reference point and were therefore excluded from the list.

In this study, we undertook a rapid simple quantitative species-by-species approach to assessing the effects of fishing on the sustainability of 456 data-poor bycatch species. This may be a simple and feasible approach to achieving the objectives of EBFM by allowing all species to be impacted within sensible limit reference points. We demonstrated that this approach, initially applied to elasmobranch bycatch (Zhou and Griffiths in press), can be easily applied to highly diverse and data-limited fish assemblages and may be easily transferable between fisheries. This approach can circumvent qualitative assessments or full stock assessments on large numbers of impacted species by using fishery or research data in its simplest form (i.e. presence-absence) and life history parameters that can be relatively easy to estimate or obtain from the literature. Because this framework is similar to the typical management regimes used for target species, the approach can be directly translated and incorporated into existing fishery management strategies.

The SAFE approach is flexible and transferable between fisheries regardless of size or fishing methods used. Unlike qualitative methods (e.g. Stobutzki et al. 2001; Fletcher, 2005; Astles et al. 2006), SAFE focuses on one metric – fishing mortality rate. This allows different methods to be used to estimate fishing impact depending on available data. Although in this paper we use detection-nondetection data and catch rates to estimate fishing mortality rates, other methods, such as using age or length data, or expert opinion when little data are available, can be used. For example, Pope et al. (2000) used a swept-area method to estimate fishing mortality. Pearce and Boyce (2006) summarised methods for modelling distribution and abundance using presence-only data, which may be used to estimate the relative fishing mortality rate, similar to our approach in his paper. If more information is available, more common techniques, such as catch curves, length-cohort analysis (Jones 1981; ICES 1988), catch-age methods (Quinn and Deriso 1999), or virtual population analysis can be used to estimate fishing mortality.

The SAFE approach uses one single life history parameter, natural mortality, for setting sustainability reference points. Natural mortality can be more easily and cost-effectively obtained compared to population parameters such as abundance, population growth rate and density dependent parameters (Pauly 1980; Quinn and Deriso 1999). Methods that use natural mortality and growth parameters to determine the limit reference point were not used in this study, since they are generally less conservative (e.g. Deriso 1982; Zhou and Griffiths in press). The sustainability of a species' population depends on its intrinsic ability to tolerate external pressures, which is directly related to their life history traits (Charnov 1993; Jennings 1998; Froese and Binohlan 2000; Denney et al. 2002; Reynolds et al. 2005; Goodwin et al. 2006). For target fish species, natural mortality has been widely used as a surrogate for optimal fishing mortality since the 1960s (Alverson and Pereyra 1969; Gulland 1970). Because finer relationships between natural mortality rate and optimal fishing mortality rate may vary between taxonomic groups (Francis 1974; Deriso 1982; Garcia and Csirke 1989; Clark 1991), we applied a scaling parameter that is a linear decreasing function of the natural mortality rate. Further study is needed to establish a more rigorous relationship between biological reference points and life history parameters.

Many fish populations can be exposed to fishing mortality from numerous sources. However, it is difficult to assess the cumulative impacts of numerous sources on species using previous semi-qualitative methods (e.g. Stobutzki et al. 2001b). In contrast, because SAFE uses one single fishing mortality rate as the standard measure of fishing impact, fishing mortalities from each source can be simply summed to estimate the total impact. Such a cumulative impact can then be evaluated against reference points to determine whether the species can sustain the total impact.

We assessed two types of risks using two biological reference points for non-target bycatch species: maximum sustainable fishing mortality and minimum unsustainable fishing mortality. The first corresponds to the maximum sustainable yield (MSY) for target species. F_{msm} is a new concept for bycatch since non-target species are not required to be managed in a manner that maximises yield.

It is known that for target species the sum of single species MSY is greater than MSY for the system, and is therefore energetically impossible to simultaneously maximize yield for multiple species, especially if they have strong interactions (May et al. 1979; Link 2002). Because our assessment was conducted at the species level, the reference point for each species may be overestimated in the context of whole community. Furthermore, we do not intend to discuss what the best outcome for the resource itself is, and are precautionary in advocating F_{msm} as a management goal for non-target species. Theoretically, applying F greater than F_{msm} (but less than F_{crash}) will eventually reduce the stock to a level that can support a surplus mortality less than MSM but not endanger its ecological sustainability. However, when the estimated fishing mortality is greater than F_{msm}, it indicates potential overfishing for such a species. For the 12 species whose estimated u (including uncertainty) was greater than u_{msm}, we recommended their inclusion in a long-term monitoring program in order to collect additional data that may allow the sustainability of their populations to be assessed using more rigorous population modelling approaches.

The second reference point is the minimum unsustainable fishing mortality that is expected to render population extinct. There are numerous instances where exploitation of target species have resulted in local extinctions of target and bycatch species (Dulvy et al. 2004). Criteria have been established and used to define the risk of extinction in marine fishes (IUCN 1994; Musick 1999). Extinction can be defined as the point at which the last member of a species has died (Purvis et al. 2000). However, extinction risk is difficult to measure for marine fishes since the population behaviour at densities close to extinction can be complicated by many factors, such as Allee effects. One of the goals of EBFM is to prevent populations from declining long before the potential risk of extinction is identified (Sainsbury et al. 2000). However, fishery scientists often attempt to quantify the relative threat or quasi-extinction risk to a fish species within different risk categories rather than the risk of absolute extinction (Musick 1999). The large number of non-target species and their scarcity of biological and catch data make the task extremely difficult.

In the absence of established or practical reference points for non-target species (see Diamond, 2004; Hall and Mainprize, 2004) we recommend the reference points of u_{msm} as a safeguard and u_{crash} as a threshold for guiding fishery management of non-target bycatch. Theoretically, applying F greater than F_{crash} year after year will eventually drive the population extinct. With adequate knowledge and data, fishing pressure should immediately be reduced when the estimated current F is greater than F_{crash}. However, despite the large number of species impacted by the NPF (see review by Griffiths et al. 2004), our results indicate that few species are at risk of becoming unsustainable due to fishing. Two main factors contributed to this outcome: 1) the aggregated fishing area is small relatively to entire management area (~5%), which limits the proportion of the population exposed to fishing, and 2) most teleost bycatch species have high resilience, having short life spans, small body sizes, fast growth rates, and high natural mortalities (Jennings 1998; Denney et al. 2002; Frisk et al 2004; Reynolds et al. 2005).

In spite of many positive attributes of the method, we recognize that there are some strong assumptions in estimating fishing impact (species distribution, population trend, using detection-nondetection data, catch rate and escapement rate, etc.) and in deriving reference points (linking sustainability to natural mortality, uncertainty in parameter estimation, etc.). Zhou and Griffith (in press) addressed these potential weak points. We concur with their views and omit further discussion in this paper.

In conclusion, 12 species have upper 95% confidence intervals of estimated fishing mortality rates greater than the fishing mortality rate corresponding to their maximum sustainable fishing mortality. This means these species are at potential risk of overfishing. Amongst them, two species, *Dendrochirus brachypterus* and *Scorpaenopsis venosa*, have upper 95% confidence intervals of estimated fishing mortality rates greater than their minimum unsustainable fishing mortality rates. These two species had reasonable sample sizes, model parameter estimates, and their detection in the NPF concurred with known distributions of these species. Therefore, they should be the priority species for the further research. The stock status of these species should be analysed by more rigorous methods when necessary data become available.

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Table 1. Parameters for species that have estimated upper 95% confidence intervals of fishing mortality rates greater than u_{msm} (fishing mortality rate corresponding to the maximum sustainable fishing mortality). Two underlined species have point estimates u greater than u_{msm}. Five species in bold have estimated upper 95% confidence intervals of fishing mortality rates greater than the minimum unsustainable fishing mortality rate u_{crash}. P_N = proportion of abundance in fished area; q = catch rate; M = instantaneous natural mortality.

Species	P_N	SE[P_N]	q	M	u	u+95%CI	u_{msm}	u_{crash}
Bathophilus nigerrimus	1.00	0.00	0.30	2.12	0.30	0.58	0.57	0.81
Benthosema pterotum	1.00	0.00	0.30	2.88	0.30	0.58	0.42	0.67
Cottapistus cottoides	0.08	0.05	1.00	1.48	0.08	0.17	0.17	0.31
<u>***Dendrochirus brachypterus***</u>	0.38	0.13	0.92	1.68	0.35	0.59	0.33	0.55
Epinephelus malabaricus	0.22	0.05	0.47	0.26	0.10	0.18	0.13	0.24
Hemiramphus robustus	0.03	11.12	0.30	1.10	0.01	1.00	0.58	0.82
Johnius australis	1.00	0.00	0.30	1.03	0.30	0.58	0.53	0.78
Lepidotrigla argus	0.13	0.07	1.00	1.58	0.13	0.27	0.26	0.45
Lutjanus johnii	0.08	0.06	1.00	0.66	0.08	0.19	0.19	0.34
Lutjanus rufolineatus	0.01	11.70	0.17	1.50	0.00	1.00	0.64	0.87
Onigocia spinosa	0.18	0.21	1.00	1.28	0.18	0.60	0.44	0.69
Parascolopsis tosensis	0.01	11.71	0.97	2.44	0.01	1.00	0.79	0.95
Parupeneus barberinoides	0.12	0.11	1.00	1.09	0.12	0.33	0.28	0.48
Richardsonichthys leucogaster	0.00	0.10	1.00	1.48	0.00	0.19	0.17	0.31
Scolopsis vosmeri	0.45	0.24	0.97	1.56	0.43	0.89	0.82	0.97
Scomberoides commersonnianus	0.38	0.03	0.47	0.43	0.18	0.30	0.21	0.38
Scorpaenopsis macrochir	0.35	0.22	0.30	1.18	0.10	0.27	0.23	0.40
<u>***Scorpaenopsis venosa***</u>	1.00	0.18	0.30	1.28	0.30	0.60	0.25	0.43
Sphyraena jello	0.32	0.14	0.30	0.37	0.09	0.22	0.15	0.27
Torquigener hicksi	0.29	0.23	1.00	2.03	0.29	0.74	0.70	0.91
Triacanthus nieuhofi	0.48	0.36	0.30	1.18	0.14	0.39	0.37	0.60

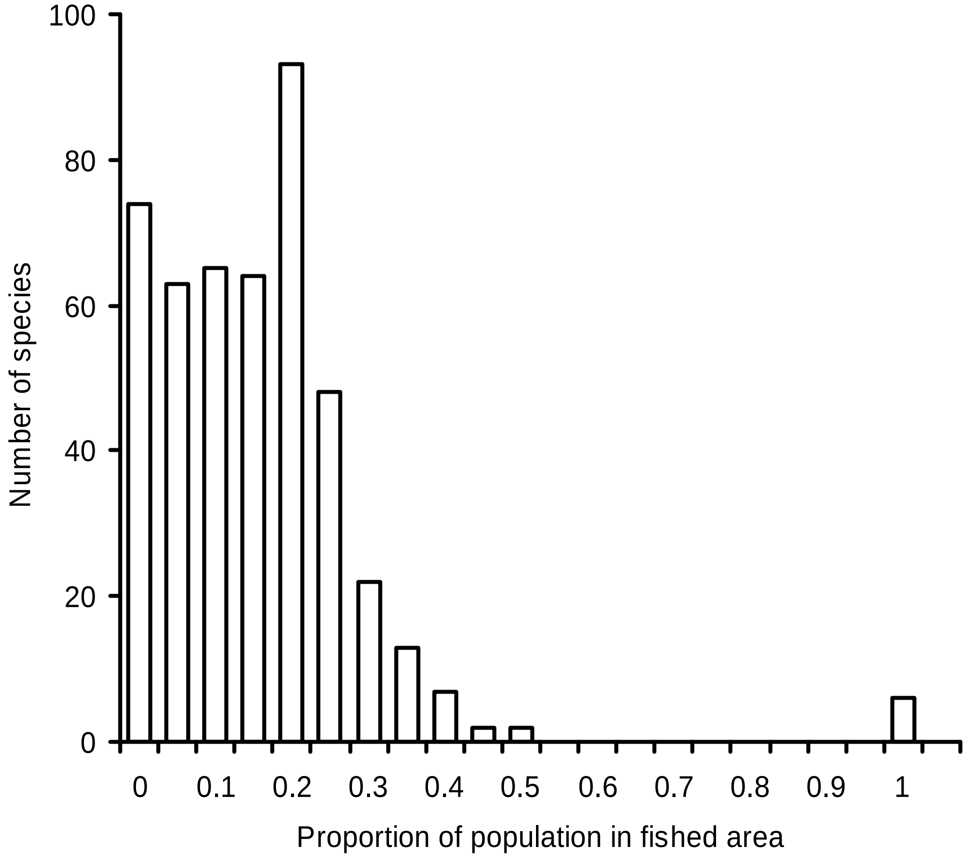

Figure 1. Spatial distribution expressed as mean proportion of population in fished area for 456 teleost bycatch species in NPF.

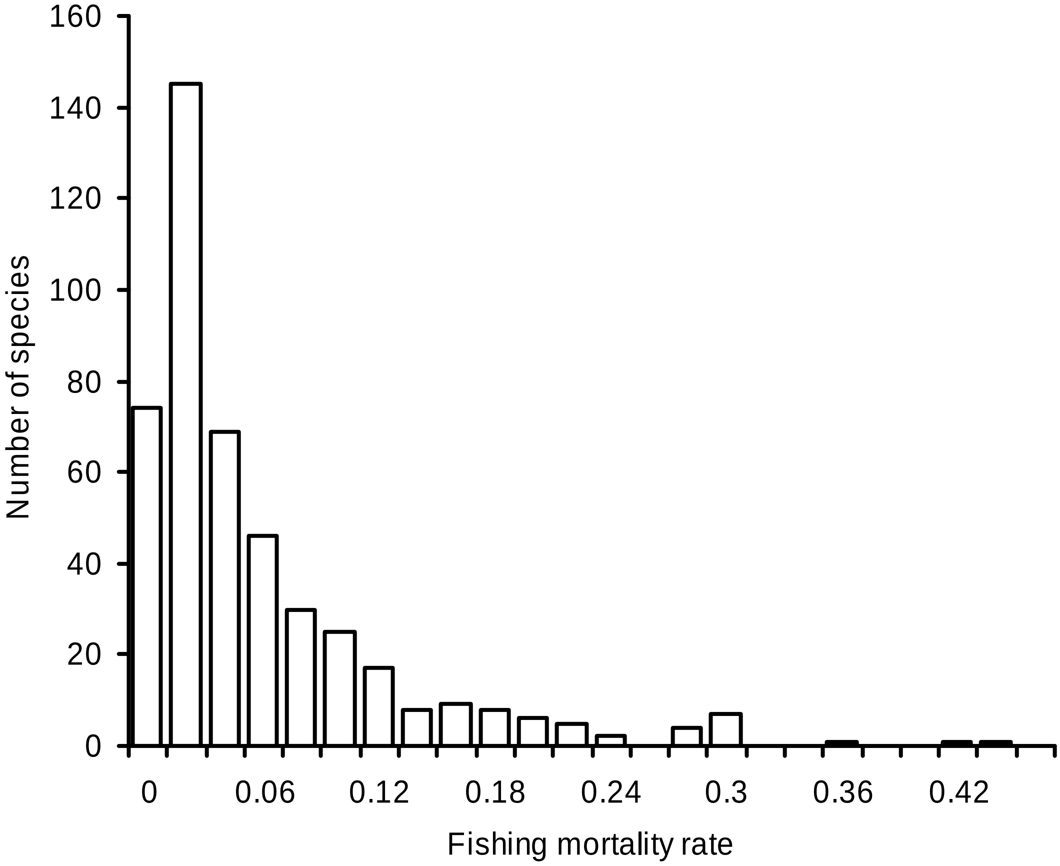

Figure 2. Distribution of the estimated mean fishing mortality rates for the teleost bycatch species.

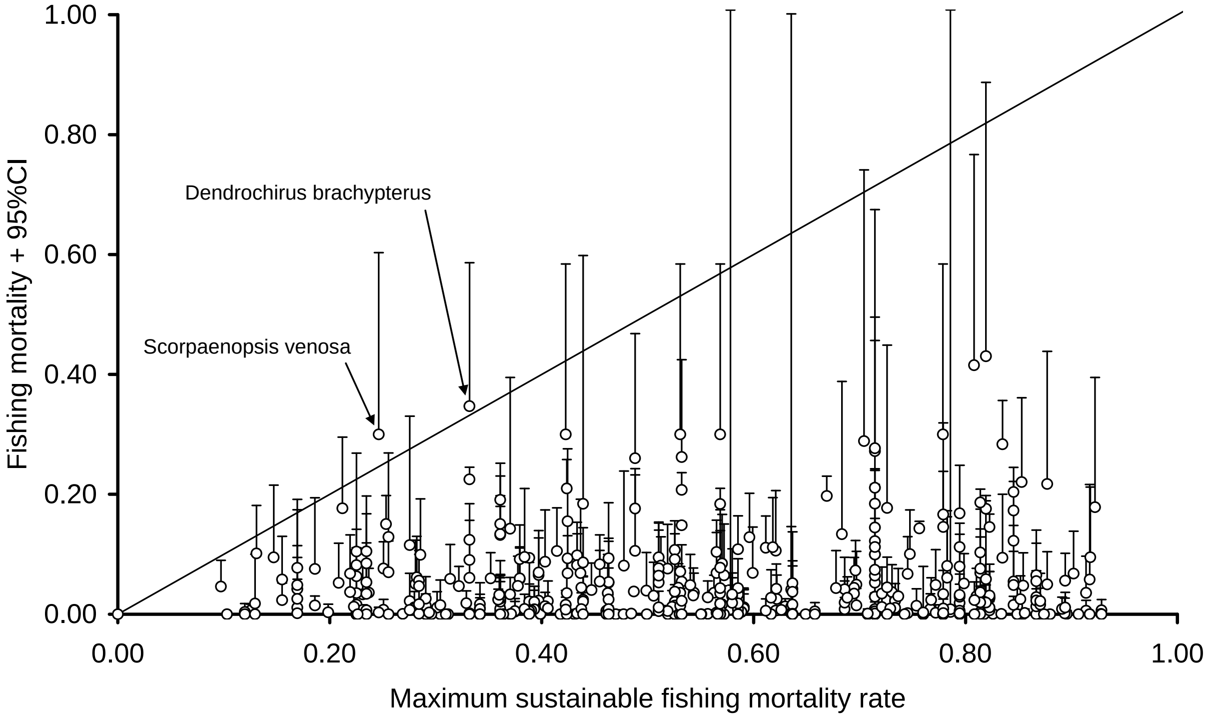

Figure 3. Comparison between estimated fishing mortality rates (u + 95% confidence intervals) from prawn trawling and the fishing mortality rate corresponding to maximum sustainable fishing mortality (u_{msm}) for the 456 bycatch teleost species. The diagonal line is $u = u_{msm}$.

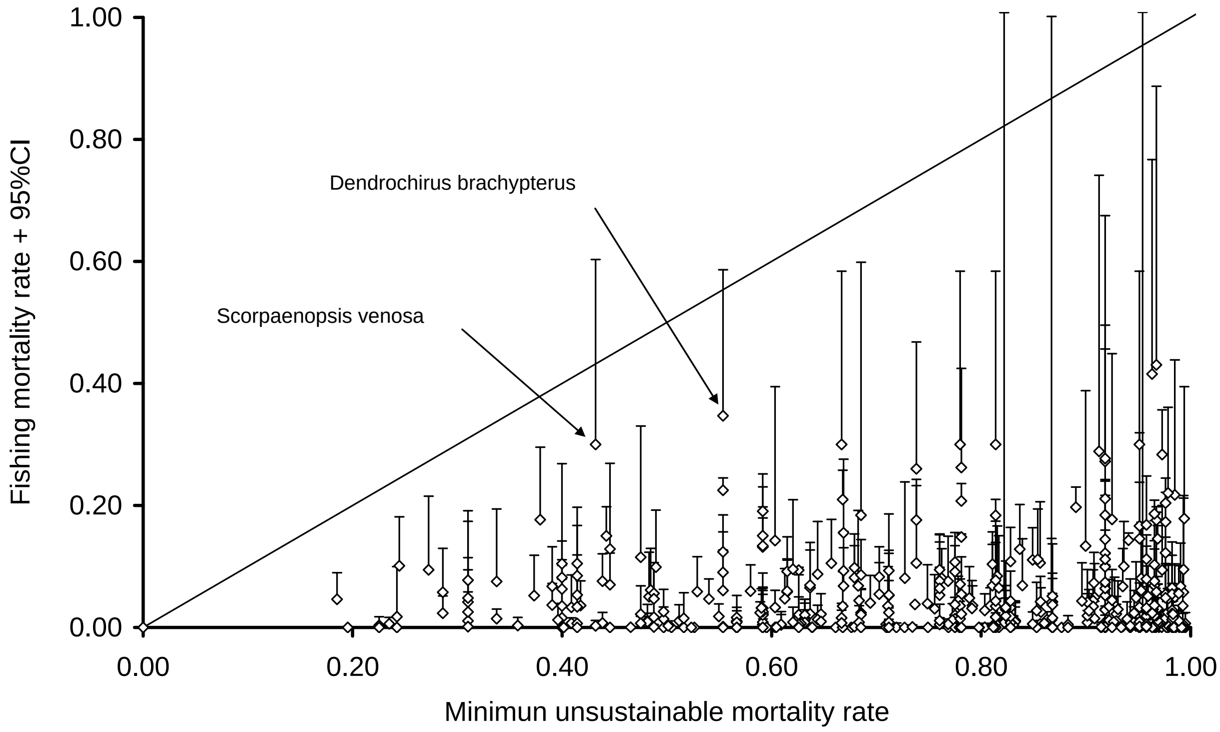

Figure 4. Comparison between estimated fishing mortality rates (+ 95% confidence intervals) from prawn trawling and the minimum unsustainable mortality rate (u_{crash}). The diagonal line is $u = u_{crash}$.