Ensuring monitoring and management of bycatch in Southern Rock Lobster Fisheries is best practice

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Abbreviations

DEE: Department of energy and the environment.

EPBC: Environmental protection and biodiversity conservation (Act).

ERAEF: Ecological risk assessment for the effects of fishing

EZ: Eastern zone. Eastern management zone in the Victorian rock lobster fishery.

FAO: Food and agricultural organisation (of the United Nations).

IRI: Index of relative importance. Used in assessing importance of bycatch species.

NZ: Northern zone. Northern management zone in the South Australian fishery.

PCM: Post capture mortality.

SA: South Australia.

SARLF: South Australian rock lobster fishery.

SZ: Southern zone. Southern management zone in the South Australian rock lobster fishery.

SRLF: Southern rock lobster fishery.

TAS: Tasmania.

TRLF: Tasmanian rock lobster fishery.

TCS: Tier classification system (used in the assessment of bycatch monitoring programs by the United States National Marine Fisheries Service)

TEPS: Threatened endangered and protected species.

VIC: Victoria

WZ: Western zone. Western management zone in the Victorian rock lobster fishery.

Executive Summary

Bycatch is an important issue in fisheries worldwide, with the impacts of fishing activities on non-targeted species and the wider marine environment receiving increasing public attention. Issues such as the potential wastage of resources through discarding of unwanted catch, ecological impacts on non-targeted species and the possibility of negative impacts on Threatened Endangered and Protected Species (TEPS) have led to an expectation that government and other managers will report on the status and impacts on these species. In order to do this effectively, well designed monitoring programs need to be in place. Effective bycatch monitoring programs allow researchers to understand which species are important as bycatch across a fishery, how the quantity of bycatch is changing through time, and any potential risks to bycatch species.

This report provides the most in-depth analysis of bycatch across the entire Southern Rock Lobster Fishery (SRLF) to date, involving researchers, stakeholders and managers across South Australia, Victoria and Tasmania. We use information from independent scientific observer programs and scientific research cruises collected over a period of greater than 15 years to:

(i) Explore the important bycatch species in each state and management zone;
(ii) Conduct a critical appraisal of the current monitoring programs by comparing them to international best practice;
(iii) Help inform a risk assessment for all bycatch species through workshops held in each state involving key stakeholders including researchers, fishers, fisheries managers, scientific observers involved in the monitoring programs, scientific experts and ecologists.;
(iv) Explore quantities and trends in bycatch for species deemed to be at moderate risk from fishing activities.

Important bycatch species in terms of frequency, total number and weight varied across the states and management zones but generally included wrasse, leatherjackets, perch, octopus, crabs (hermit, velvet and giant) and sharks (Draughtboard Sharks in Tasmania and Eastern Victoria, and Port Jackson Sharks in Western Victoria and South Australia). Undersized Rock Lobsters, which are also considered bycatch, formed a large component of the overall bycatch particularly in Tasmania.

The current bycatch monitoring program was assessed against international best standards defined under the United States Tier Classification Scheme developed by the National Marine Fisheries Service. We found that each state managed program fell into a Tier 2 classification out of five possible tiers ranging from 0 to 5. This score was reasonable when comparing the programs in other fisheries given the size of the SRLF. However, areas for improvement in the observer programs were identified and recommendations on how to improve the ongoing monitoring program are made in light of our findings.

The risk assessment found that no bycatch species was at high risk from fishery operations of the SRLF. Species that were identified as having a medium potential risk were a subset of those that are kept as byproduct either for consumption, sale or bait. Barotrauma was also identified as a risk factor for some finfish species with swim bladders as these species when brought up from depth may suffer injury or be unable to descend and thus more susceptible to predation. Also, missing life history information for a number of species meant that precautionary higher risk scores were assigned to these species until more information is obtained. Rates of encounter with gear of Threatened, Endangered and Protected Species (TEPS) were found to be low, and consequently direct threats from fishery operations likely to be low; however, ongoing monitoring of TEPS interactions is a necessary component of best practice.

A short list of ten species identified as being more susceptible to risk from the SRLF were given further analyses. These ten species included Draughtboard Shark, a number of leatherjacket species, Ocean Perch, Blue-throat Wrasse and Conger Eel. Analysis of these species and groups allowed for estimates of total catch of these species and trends in catch through time. These estimates provide a baseline for ongoing monitoring and the setting of reference points for management action for these species.

Based on the findings of this report, it is recommended that:

- Improvements are made to the observer programs including increasing the number of vessels participating, creation of consistent reporting methods, improved species identification
- Information is collected for bycatch species with missing life history parameters to allow increased confidence in future risk assessments

- Species identified in this report as being of primary or secondary importance as bycatch in the SRLF are prioritised for ongoing reporting and monitoring, with a periodic census of all bycatch species (perhaps every 5 years) used to detect any trends in overall bycatch composition
- Due to the considerable noise in bycatch data, longer-term trends are used as management trigger points
- Further research is conducted into reducing the amount of undersized Rock Lobster

Introduction

General

Bycatch has become a major issue in fisheries science, monitoring and management due to several reasons including the perceived wastage of resources and its potential to affect both exploited and non-exploited species. Such issues have resulted in significant changes in the way fisheries are managed throughout the world, including a policy shift towards ecosystem-based approaches, which take account of the effects of fishing beyond the targeted species. In recent decades, bycatch issues have also contributed to changes in expectations in terms of the ecological sustainability of fisheries, of the general public (owners of fisheries resources), governments (charged with the stewardship of those resources on behalf of the public) and seafood markets through eco-labelling initiatives (that buy and sell these resources).

Despite this increasing awareness of the importance of bycatch, clear guidelines regarding its monitoring and reporting are lacking for many fisheries throughout the world - including Australia (Kennelly 2018), where a FAO estimate indicates that 55% of catches are discarded (Kelleher, 2005). While there have been several policies developed in Australia about bycatch for Australia's Commonwealth fisheries (DAFF 1999; DAFF 2000; DAWR 2017), many jurisdictions lack such instruments. Furthermore, because bycatch impacts are fishery- and gear-specific, there is a significant need for detailed monitoring and management of bycatch in individual fisheries across Australia, together with an identification of areas for improvement.

Definitions

The term "bycatch" is used in a diverse way in the literature, so it is important for any study about bycatch to establish its own particular scope and definitions. Here we adopt a definition for bycatch, as the unintended catch, or interaction with, species that are not retained for sale, while targeting particular species (or sizes of species). In many fisheries, a portion of non-target species are kept for the purposes of eating, selling or bait which we classify here as "byproduct", and we include such organisms in our definition of "bycatch". We further categorise bycatch as species that are returned to the sea as "discards" (including target species that are undersized or subject to other restrictions like over-quota, etc); and that part of the catch that does not reach the deck but is affected by “interactions” with the fishing gear. Typically, discards are the main focus of studies into bycatch as they are perceived as wastage. Both discards and non-capture interactions may involve threatened, endangered or protected species (TEPS) and, in Australia, are subject to reporting and review under the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act). Impacts on TEPS are often of particular concern to conservation groups and the wider community and therefore attract higher profile scrutiny than species that are more common and/or less charismatic.

The Southern Rock Lobster Fishery

Australia’s Southern Rock Lobster Fishery (SRLF) is an economically important fishery that operates across southern Australia. Annual catches range from 3500 – 4000 tonnes, with a gross revenue of greater than AUD$200 million. The fishery is managed on a jurisdictional basis between South Australia (SA), Victoria (VIC) and Tasmania (TAS). Programs exist in each state to manage the fishery, including regular stock assessments, quota monitoring and licensing. In terms of mitigating the levels of bycatch in the fishery, management measures include the use of mandatory escape gaps in pots across most management zones in the fishery, spatial closures, limited soak times for pots and devices for the exclusion of seals from pots. Escape gaps are designed to ensure that the majority of undersized lobsters and smaller bycatch species can escape from the pots. Legislated requirements in terms of mitigation measures vary between jurisdictions, and sometimes within management zones, such as the use of seal exclusion devices only being mandatory in certain areas.

Each state has systems in place for monitoring bycatch and TEPS interactions. Bycatch in the SRLF is recorded through a combination of voluntary reporting by fishers (which has generally low participation rates), on-board observer programs (where trained observers go aboard commercial trips and collect bycatch data along with other data such as size and sex of rock lobster) and research cruises (that use lobster pots but often with closed escape gaps and thus may not be directly comparable with commercial catches). Byproduct kept or used by fishers is subject to mandatory reporting in dedicated logbooks; however comparisons to observer data has shown that byproduct is likely to be underreported, at least in some jurisdictions, particularly for species used as bait (Hartmann, Gardner & Hobday 2013). Reporting of any interactions with TEPS is mandatory under the EBPC Act, with reports to be filed with the Department of the Environment and Energy.

Periodic assessments of the SRLF under the *Environment Protection and Biodiversity Conservation Act 1999* (the EPBC Act) are made every ten years on a jurisdictional basis by the Department of Environment and Energy (DEE) (formerly the Department of Environment and Heritage). These assessments examine the ecological sustainability of the fishery, including impacts on bycatch, byproduct and TEPS within the framework of the EPBC Act under the Guidelines for the Ecologically Sustainable Management of Fisheries (Fletcher *et al.* 2003). Such assessments generally conclude that impacts on bycatch and byproduct species are relatively minor for this fishery; however, the need to develop formal decision rules around levels of bycatch that are linked to management responses has been consistently noted, at least for one jurisdiction (Tasmania) (e.g. DEH 2004; DEE 2016). Jurisdictional stock assessments and risk assessments also regularly report on bycatch and byproduct.

Despite these systems, a recent project indicated that there are areas for improvement in the performance of bycatch monitoring and reporting in this fishery relative to industry best-practice criteria (Kennelly 2018). The need for improvements in bycatch reporting, assessment and management for the SRLF has also been discussed at Crustacean Research Advisory Committees/groups from South Australia, Victoria and Tasmania. Numerous small issues have been identified that require action. For example, it has been noted that data collection should be made consistent across jurisdictions to enable sharing of data to measure total impacts across all parts of the fishery. This includes details like making decisions on whether to measure bycatch as weight and/or numbers, using consistent identification and naming at the species level, and whether details like individual size, sex or vitality are recorded. Furthermore, a formal risk assessment for bycatch species has not been conducted for the fishery as a whole. Outputs of a risk assessment could be used to aid in identifying a subset of bycatch species that may be at increased risk. An analysis of the historical catch of these species could then be used to develop performance indicators to assist management of these species across the fishery.

This current project provides a detailed examination of all aspects of bycatch and byproduct in relation to the SRLF in Australia in order to ensure monitoring and management standards are best practice. An analysis of current data collection practices is conducted across each jurisdiction and appraised in order to define a consistent protocol across the fishery, a need that has been previously identified (see Linnane & Walsh 2011). We also provide the results from a Productivity and Susceptibility (PSA; Hobday *et al.* 2007) risk assessment conducted for bycatch, byproduct and TEPS at workshops held within each jurisdiction. Based on the risk assessment outputs, a subset of higher-risk species is examined in more detail through the use of data poor stock assessment approaches. Management performance indicators for these species are examined and reference points suggested. Based on an assessment of the current monitoring program, we also make recommendations to improve ongoing monitoring and management of bycatch in the SRLF. Our overarching goal is to provide a framework for bycatch monitoring that is of the highest standard possible, capable of being held up to scrutiny by eco-labelling certification bodies such as the Marine Stewardship Council (MSC), or by future assessments for export certification by DEE.

Need

As for any fishery, documenting and assessing the impact of fishing on bycatch is required for rock lobster fisheries to enable the appropriate management of ecosystem interactions. This is also needed to satisfy obligations for assessments under the EPBC Act, jurisdictional reporting as well as international instruments such as the UN FAO's Guidelines for Bycatch Management and Discard Reduction. Bycatch information is currently collected in all jurisdictions that manage Australia's SRLF but improvements are required if they are to meet the standards required for rigorous certification such as those required under MSC's Principle 2. Furthermore, demonstration of adopting best practice can provide benefit to fisheries in terms of community acceptance.

Whilst we consider that bycatch issues are not severe for this fishery (nor for many other lobster fisheries using pots), there is nonetheless room for improvement, especially for such valuable fisheries of this size. Specific issues include poor quality of byproduct reporting in logbooks, a lack of combined assessments of bycatch risks (and cumulative impacts of such risks) across jurisdictions, reporting systems not consistent with standard and/or best practice (e.g. recording numbers and not weights), different risk-based assessment methods being used across jurisdictions, and no agreed/implemented approach for monitoring the status of species that are at high to moderate risk. Bycatch management ideally should be integrated into the harvest strategy for a fishery and this notion will be examined through this project through the development of reference points for relevant species. Additionally, there is a need for transparent assessment and reporting of bycatch and TEPS interactions that are best practice to improve community acceptance of the SRLF.

Objectives

The objectives of this project are:

1. To define consistent data collection methodology of bycatch and TEPS that is best practice and can be verified across South Australia, Victoria and Tasmania.
2. To do a risk assessment pooled across all jurisdictions which will comprise all current information on bycatch and TEPS in the SRLF.
3. To develop best practice ongoing bycatch monitoring and reporting for the SRLF based on an assessment of the current program against international best practice.
4. To conduct quantitative assessments of selected byproduct species ranked at higher risk in objective 2, including data poor stock assessment methods where appropriate.
5. To provide guidance around establishment of appropriate performance indicators for moderate and high risk bycatch species and associated management strategies.

Methods

1. Comparison of current bycatch data collection and monitoring protocols and practices with industry best practice and exploratory data analysis

In accordance with Objective 1, we assessed the current bycatch programs operating in each jurisdiction across the fishery against international best practices in bycatch monitoring and reporting. We conducted exploratory analyses in order to inform this assessment and to gain insights into the key bycatch species.

1.1 Comparing the consistency of current bycatch reporting compared to international best practice data collection standards

The quality of bycatch data collection programs currently operating in each jurisdiction was assessed using the Tier Classification System (TCS) developed by the US National Marine Fisheries Service (NMFS 2011). This system uses a series of quantifiable criteria such as the longevity of observer programs and sampling design (number of vessels, trips, hauls observed), spatial and temporal coverage, etc. and gives a greater weight to observer data compared to data collected by fishers. Scores under the various criteria are then summed, and the overall score places the bycatch data collection program into one out of a possible five tiers (0 – 4, with a higher rating being better). Project members from each state used a combination of their expert opinion and the outputs of the exploratory analysis (see section 1.2) to score their respective jurisdiction under each criterion.

1.2 Statistical exploration of the temporal and spatial consistency of bycatch data collection across the fishery and identification of key species, trends and patterns

Exploratory analyses of the observer data were conducted on a jurisdictional basis, using a standardised approach for each jurisdiction. For SA and VIC, where distinct zones are managed within the jurisdiction, analyses were conducted for each management zone.

Bar plots or tables of the number of monitored pots in each month over the fishing season across the length of the observer program were produced for each jurisdiction. The length of the observer program and its temporal consistency varied between jurisdictions and their associated management zones and were explored through summary statistics and plotting. The spatial distribution of bycatch sampling effort was also explored through the creation of summary statistics or maps.

An “Index of Relative Importance” (IRI) was developed to quantify important bycatch species, taking into account numbers and weights of species. Results examine all data pooled and annual temporal patterns where sufficient data existed. For brevity, only pooled summaries are presented in this report. For SA, IRIs were calculated separately for the Southern Zone (SZ) and Northern Zone (NZ) management areas. For VIC, IRIs were calculated separately for the Western Zone (WZ) and Eastern Zone (EZ) management areas. For TAS, only data from years with greater than 100 pot lifts were included, which was 2009 - 2014. We produced plots that both include and exclude undersize lobster, as these are a major component of the bycatch but mask the importance of other species when included.

A ratio of “fishing effort:bycatch” was calculated for each jurisdiction. This was achieved by using counts of bycatch species in each pot from observer data. We converted counts to weights based on an average length for each species (see Appendix A). Observer recorded lengths were available for a subset of the Tasmanian observer bycatch data as well as from observer bycatch data from 2005 to 2007 in Victoria, where observers recorded the length of all bycatch species. Where length data was not available for a species, the expert opinion of two marine ecologists was sought and the average of these two lengths was used as the basis for calculating mean length. Details of the mean lengths, standard deviations of these lengths and the number of observations used in calculating the mean length are provided in Appendix A. The ratio was calculated for bycatch that both included and excluded undersized Rock Lobster. Weights of bycatch per pot were then used to calculate a mean ratio of bycatch per pot across the respective span of bycatch data in each state management zone, thereby giving a mean ratio of effort:bycatch (i.e. kg of bycatch per potlift). Standard errors were calculated by bootstrapping across mean ratios, with years as strata. This ratio was then used to calculate a total mean bycatch amount in each management zone, by multiplying the ratio in a given fishing season by the commercial effort in the respective zone/season and averaging across seasons. The associated standard deviation was calculated by using the formula for the product of variances (Goodman 1960), where the bootstrapped variances of the ratio were combined with variance in commercial effort across seasons.

2. Risk assessment for bycatch, byproduct and TEPS across the Southern Rock Lobster Fishery

For objective 2, a risk assessment for all bycatch, byproduct and TEP species was conducted across the SRLF. We used the ecological risk assessment for the effects of fishing (ERAEF) approach developed by Hobday *et al.* (2007). Workshops were held in each jurisdiction in order to engage stakeholders in the risk assessment process. Details of the ERAEF approach and how we applied it to the SRLF are outlined below.

2.1 An overview of the ecological risk assessment for the effects of fishing

The ERAEF approach was developed by CSIRO in response to the need for an improved methodology when assessing the ecological risk posed by fisheries against the guidelines of the EPBC Act 1999. It has been applied to many of Commonwealth managed fisheries for over a decade. The methodology is hierarchical in nature, where low risk components are screened out at first thereby allowing analyses to focus on components that are identified as higher risk. These components may range from entire categories (e.g. bycatch or habitat impacts) at lower levels, to elements within these components (e.g. individual bycatch species) at higher levels. The hierarchy consists of:

(i) Scoping: identification of objectives and potential hazards;

(ii) Level 1: Scale Intensity Consequence Analysis (SICA): an analysis focussed on the most vulnerable element within each component (e.g. habitat, bycatch, byproduct, TEPS) to assess whether any risk is posed and further analysis needed;

(iii) Level 2: Productivity and Susceptibility Analysis: A semi-quantitative approach designed to assess the relative risk to individual elements (i.e. species) within each component (e.g. byproduct, bycatch and TEPS). This is the methodology used in this report and is outlined in more detail below;

(iv) Level 3: Fully quantitative approaches including stock assessment for species where this is deemed necessary from PSA outputs;

A scoping exercise or a level 1 (SICA) assessment was not undertaken as part of the project as it was assumed that at least some risk was posed to each of the components of bycatch, byproduct and TEPS to warrant a level 2 analysis (ie. a PSA). This decision was made as the present project was developed with the intention of proceeding to higher levels in this hierarchy (i.e. level 3 – fully quantitative approaches) for any species found to be at medium-high risk at level 2. This therefore led us to conclude that, at the minimum, a level 2 analysis would be necessary.

2.2 Productivity and Susceptibility Analysis (PSA) - an overview

PSA is a semi-quantitative ecological risk assessment where risk scoring is conducted for each species on an individual basis, with final overall risk ratings for each species being classified as high, medium or low. It should be noted that this approach assesses potential rather than actual risk, because analyses do not take into account the level of catch, size of the populations, likely exploitation rates or any potential management actions already in place. The aim at this level of analysis is to filter out a list of any species that are under potential risk in order to take a more detailed analysis at level 3 (i.e. more quantitative approaches).

The method scores each impacted unit (species or higher taxonomic grouping) based on:

Productivity – biological characteristics of the species such as fecundity, reproductive strategy, size/age at maturity, etc.

Susceptibility – how susceptible the species is to the fishing method

Both productivity and susceptibility result in final scores between 1 (= low risk) and 3 (= high risk). The final ranking is a score on each of these two axes (productivity and susceptibility) resulting in an overall score that falls within bounds of high, medium and low risk (Figure 1). The primary components considered in PSA are summarised in Table 1.

Table 1. Attributes scored in the PSA analysis (taken from Hobday et al. 2007).

	Attributes
Productivity	Average age at maturity
	Average size at maturity
	Average maximum age
	Average maximum size
	Fecundity
	Reproductive strategy
	Trophic level
Susceptibility	Availability considers overlap of fishing effort with species distribution
	Encounterability considers the likelihood that a species will encounter fishing gear that is deployed within the geographic range of that species (based on two attributes: adult habitat and bathymetry)
	Selectivity considers the potential of the gear to capture or retain species
	Post capture mortality considers the condition and subsequent survival of a species that is captured

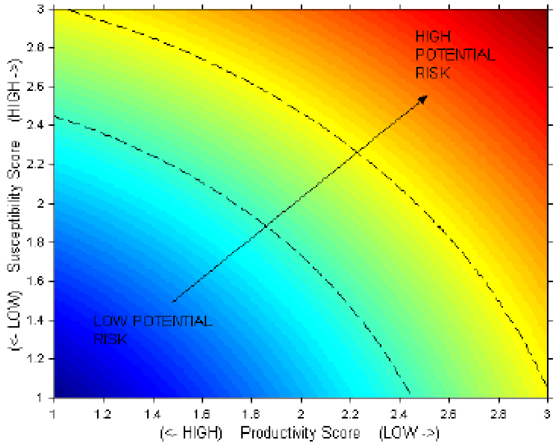

Figure 1. The axes on which the risk to ecological units is plotted. The x-axis, productivity, includes attributes that influence the productivity of a species, or its ability to recover from fishing impacts. The y-axis includes attributes that influence the susceptibility of the species to impacts from fishing. Contour lines show how final scores on each axis partition species into overall potential risk scores (taken from Hobday et al. 2007).

The approach employs the precautionary principle - where missing data results in a higher risk value being assigned. This is particularly relevant where a species' biological information is missing, which will result in a default higher risk score for that species on the productivity scale, and therefore a greater chance of an overall high risk rating. This approach is therefore quite conservative and likely to generate more false positives for high risk (species assessed to be high risk when they are actually low risk) than false negatives (species assessed to be low risk when they are actually high risk).

A necessary component of the PSA approach is the involvement of stakeholders in the process. Open discussions of the results and the inclusion of expert opinion can be used as part of the final scoring process to "override" scores where data is documented and available and is deemed necessary or appropriate to take this data into account in the scoring. For example, where information is missing, but a rigorous independent observer program is in place and recorded information from this program is available, observer input can be used to determine certain susceptibility scores.

2.3 PSA for the Southern Rock Lobster fishery

A PSA was completed for the entire SRLF through consideration of the ecological impact of the fishery on a comprehensive list of byproduct, bycatch and TEP species. The process consisted of acquisition of the PSA worksheets, compilation of a comprehensive species list across the fishery, initial scoring of each species by researchers on the project and a series of workshops held in each jurisdiction to discuss the results and solicit feedback from observers and the fishing industry on the scoring of a refined subset of species.

2.3.1 Data collection for the PSA worksheets

PSA excel worksheets that were specifically designed for use in implementing the methodology were obtained from CSIRO. The worksheets are a stand-alone analysis tool that, once populated with all the necessary information, calculate risk score for each species. The worksheets came ‘unpopulated’ in terms of species data for the SRL fishery.

A species list for use with the PSA worksheets was compiled by examining data sets of bycatch and byproduct from each state, and TEPS databases from the Commonwealth Department of Energy and the Environment. Species lists for bycatch and byproduct species included historical data sets from observer reporting, research cruises and fishery dependent pot sampling from each state. Where higher taxonomic groupings were reported (e.g. “wrasse” or “leatherjacket”) rather than species, all possible species within the geographic range were included. If a species had ever been historically reported as byproduct it was listed. Similarly, any TEPS that had any geographical overlap with the range of the fishery was also included in the species list. This process resulted in a list of 251 species comprising 75 byproduct species, 42 bycatch (i.e. discard) species and 134 TEPS.

Detailed life-history, distributional and habitat/depth preference information is required for completing the PSA, with the most important attributes outlined in Table 1. For some species, data provided from a previous PSA on the Tasmanian scalefish fishery was able to be used to populate the worksheet fields. Where this was not available, data was sought from a variety of sources. For fish species, fishbase (https://fishbase.org/) was used as a source for many of the productivity attributes. Literature searches were also done for many of the species which were more dominant in the data sets. These data formed the basis for the automatic scoring of productivity risk in the PSA worksheets.

2.3.2 Scoring of the susceptibility attributes

Scoring of the susceptibility attributes (availability, encounterability, selectivity and post-capture mortality (PCM)) followed the method outlined in Hobday et al. (2007). The method uses a combination of data regarding species geographical ranges, habitat and depth preferences and sizes, along with expert opinion. Expert opinion can be used to override susceptibility scores where this is justified, with the reasons for this override recorded as part of the PSA methodology. Expert opinion was gathered through open workshop discussions held in each jurisdiction and included ecologists, fishers, managers, researchers and observers. These workshops were held in Hobart on the 15th February 2019 at the Institute of Marine and Antarctic Studies Taroona; Adelaide on the 19th February at the South Australian Research and Development Institute (SARDI) (Aquatic Sciences); and on the 21st February 2019 at the Victorian Fisheries Authority in Queenscliff.

The overall susceptibility score is calculated by multiplying the four attribute scores together and averaging so that a final score between 1 and 3 (low = 1, medium = 2, high = 3) is obtained. Therefore, if a score in any one of the attributes was low, overall susceptibility risk could not be high. This makes intuitive sense and is the logic for scoring susceptibility as a multiplicative factor. For example, if a species has a high overlap (high availability), inhabits the same habitat and depth range as the fishery (high encounterability), is highly selected for by the fishing gear (high selectivity), but is almost always released in good condition (low PCM) then overall risk will be low-medium (1.65 out of 3).

In order to score availability, species distribution maps were obtained for the byproduct and bycatch species from the Atlas of Living Australia (https://www.ala.org.au). These distributional maps were used to assess the geographic overlap of the fishery (the combined

South Australian, Victorian and Tasmanian fisheries) with the distribution of each species. The cut-offs for scoring availability were: high (> 30%), medium (10-30%) and low (<10%).

Encounterability (how likely a species is to encounter the fishing gear) was based on habitat and depth preferences for each species gathered as part of the data collection prior to the workshops. Consideration was given to the fact that the SRLF targets rocky reef habitat, generally in depths less than 110 metres. Scoring for encounterability first considered the habitat preference of a species (hard bottom, soft bottom, bentho-pelagic, meso-pelagic, epi-pelagic). Species associated with hard bottom scored high, those that range across hard and soft bottom medium, and other categories or combinations scored low. Subsequently a “bathymetry check” was used to potentially override the score, based on the depth preference of the species. High risk was given to species that inhabit 0-110 metres, medium to species that range 0-250 metres and low to all other depth ranges. The lower of the habitat or depth score was used to score encounterability as this takes into account potential habitat or depth refuge from fishing pressure.

Selectivity, the potential of a gear to catch or retain a species, provided a particular challenge for this project. Because the PSA approach has been primarily applied to trawl fisheries, the method has used a combination of mesh size and size at maturity of a species to determine selectivity. However, for a pot fishery like the SRLF, selectivity is determined by factors such as the attraction of a species to the bait, its swim speed and home range, whether it is likely to enter/exit pots during deployment and how likely it is to get out of escape gaps or other gaps in the pots. As this type of selectivity is not currently quantified and no method currently exists to do it rigorously, it was decided that expert opinion gathered through open discussion in the workshops provided the most sensible approach to scoring this attribute.

PCM, the likelihood of survival once caught, was also scored through an open discussion process in the workshops. While observers record bycatch, the fate of bycatch, for example whether it is kept as byproduct or has suffered barotrauma, is not typically recorded. The PSA methodology allows for observer input in this respect: where a long-term observer can verify that > 2/3 of a species is returned to the water in good condition, PCM can be rated as low; where the species is kept as byproduct or returned in poor condition between 1/3 and 2/3 of the time a medium risk is given; and when a species is kept or returned in poor condition > 2/3 of the time a high risk score is given.

An initial workshop to discuss the risk assessment method and initial scoring was held on February 4th and 5th 2019 and included all jurisdictional co-investigators on the project. The entire species list was discussed and an initial scoring of the susceptibility attributes of each species conducted. This process was precautionary in nature and left scores of higher risk when scoring was uncertain. Based on this initial scoring, a short list of 39 of the highest ranked byproduct and bycatch species was prioritised for detailed discussion in the workshops (Table 2). These 39 species were decided upon as they were ranked as the highest risk through the initial scoring process. It was necessary to narrow the list down in this way due to each workshop being of one-day duration which precluded the discussion of the entire list of species.

Table 2. The list of 39 species discussed in detail at the workshops held in each state. The susceptibility scoring for each species was discussed and stakeholder input considered as part of the risk assessment process.

Common name	Species
Draughtboard Shark	*Cephaloscyllium laticeps*
Barber Perch	*Caesioperca rasor*
Blue-throat Wrasse	*Notolabrus tetricus*
Purple Wrasse	*Notolabrus fucicola*
Gummy Shark	*Mustelus antarcticus*
Conger Eel	*Conger verreauxi*
Marblefish	*Aplodactylus arctidens*
Southern Octopus	*Octopus australis*
Maori Octopus	*Macroctopus maorum*
Gloomy Octopus	*Octopus tetricus*
Port Jackson Shark	*Heterodontus portusjacksoni*
Bridled Leatherjacket	*Acanthaluteres spilomelanurus*
Toothbrush Leatherjacket	*Acanthaluteres vittiger*
Velvet Leatherjacket	*Meuschenia scaber*
Degen's Leatherjacket	*Thamnaconus degeni*
Mosaic Leatherjacket	*Eubalichthys mosaicus*
Six spine Leatherjacket	*Meuschenia freycineti*
Horseshoe Leatherjacket	*Meuschenia hippocrepis*

Continues…

Brownstriped Leatherjacket	*Meuschenia australis*
Ocean Jacket	*Nelusetta ayraud*
Green-eyed Dogfish	*Squalus chloroculus*
White-spotted Dogfish	*Squalus acanthias*
Velvet Crab	*Nectocarcinus tuberculosus*
Gurnard Perch	*Neosebastes scorpaenoides*
Ocean Perch	*Helicolenus percoides*
Butterfly Perch	*Caesioperca lepidopterus*
Ribaldo	*Mora moro*
Latchet	*Pterygotrigla polyommata*
Sergeant Baker	*Latropiscis purpurissatus*
Red Cod	*Pseudophycis bachus*
Swallowtail	*Centroberyx lineatus*
Harlequin Fish	*Othos dentex*
Western Blue Groper	*Achoerodus gouldii*
Eastern Blue Groper	*Achoerodus viridis*
Snapper	*Chrysophrys auratus*
Nannygai	*Centroberyx australis*
Knifejaw	*Oplegnathus woodwardi*
Oilfish	*Ruvettus pretiosus*
Eastern Orange Perch	*Lepidoperca pulchella*

2.3.3 Risk scoring for TEPS

Scoring for all TEPS defaulted to high risk scores for susceptibility attributes. Overrides were considered where a well-established independent observer program exists. As long-term observer programs in the SRL fishery exist in all state jurisdictions, observer input was available for use in scoring TEPS susceptibility attributes. In Tasmania, TEPS interactions, whether positive, negative or benign are recorded by observers. In Victoria, an intensive observer program was conducted between 2005 and 2007 to specifically report bycatch and TEPS interactions. In addition, the frequency of TEPS sightings was also reported. In South Australia, TEPS interactions are recorded by observers and where no observers are on board are required to be reported by fishers through Wildlife Interaction Forms. The results of an exploration of each of these data sets were presented at the workshops and used as a discussion point for the scoring of TEPS. For the purpose of the PSA, all forms of interactions, including sightings, are used to assess risk. For example, the proportion of trips where a TEPS was sighted could be used to help score availability, whereas the proportion of times that a TEPS was sighted and there was an interaction with gear was used to score encounterability.

For each of the four susceptibility attributes, if observer data and observers could confirm rates of less than 1/3, the attribute could be downgraded to low; between 1/3 and 2/3, medium; and greater than 2/3, high. For example, in scoring availability, if the particular species was observed on 1/3 to 2/3 of trips, availability was scored as medium. Similarly, if encounters of a species with the gear were observed less than 1/3 of the time, encounterability was scored as low.

2.3.4 Decision rules for prioritising species for level 3 analysis

Due to the precautionary nature of the PSA approach, many species may have higher than expected risk scores. In particular, missing species attributes will result in high risk productivity scores and therefore overall higher risk ratings for those species. The ERAEF methodology (Hobday et al. 2007) suggests the post-stratification of high-risk species in order to refine the reason for high risk scores to aid in discussion of prioritisation of further research. To be precautionary, it was decided that all species that had high risk ratings would be examined in more detail, and where sufficient data exists, a more detailed quantitative analysis would be undertaken.

After initial data collection for the worksheets, there were a large number of species with > 3 missing productivity attributes resulting in default high to medium risk scores. In order to prioritise medium risk species for more detailed analysis, further refinement was required. While life-history can denote real risk, for example in slow growing species with low reproductive output, this is usually restricted to certain taxonomic families. Therefore, we decided to rank medium risk species in terms of their overall susceptibility scores to aid in deciding which medium risk species should be subjected to further analysis. The logic used here was that these species would be those that were identified as having a higher level of relative potential impact from the fishery. A cut-off of 1.5 out of 3 (i.e. 50%) in terms of susceptibility was used as a decision criterion to shortlist species.

3. Quantitative analysis of risk posed to species identified as being at higher risk

Based on the PSA risk analysis conducted in objective 2, ten medium risk species were identified for further analysis. A more detailed analysis is conducted as a final stage (level 3) as part of the ERAEF process (Hobday et al. 2007) and includes approaches such as a quantitative stock assessment for species assessed as being at risk. Prior to this analysis it is recommended that suitable existing information to further understand the risks to higher risk species should be identified. The majority of our short list of ten species were species where stock assessments are conducted in each state as part of the scalefish fishery assessment, although this varied for species across the different jurisdictions. For example, stock assessments are conducted for Blue-throat Wrasse in each state, and leatherjackets are also assessed as a group in each state, whereas a small fishery exists for conger eel and Ocean Perch in Tasmania only. In order to assess whether the level of bycatch for our shortlisted species was likely to be significant compared to reported commercial catches of these species in targeted fisheries required an estimate of total bycatch within each jurisdiction. An estimate of total bycatch for each species through time was also necessary for conducting more detailed quantitative analyses including the data-poor stock assessments conducted here.

3.1 Scaling up estimates to the commercial fishery

In order to estimate total bycatch by season for each species we used the time-series of observer and research data available in each state. Unbiased estimates of total catch were made by applying a Generalized Linear Modelling (GLM) approach that took into account factors that may influence the catch per unit effort (CPUE) for each of our species. This approach is commonly taken when estimating bycatch in fisheries (e.g. Ortiz & Arocha 2004; Minami et al. 2007; Brodziak & Walsh 2013; Walsh & Brodziak 2015); however it is often noted that bycatch data are likely to contain a large number of zeros in the data and counts are likely to be over dispersed (standard deviation of counts is larger than the mean). Both these factors make modelling of bycatch data problematic as standard distributional assumptions of GLMs are likely to be violated. Therefore, we opted to use zero-inflated models for the bycatch data as these have been shown to perform well when modelling bycatch data in other fisheries (e.g. Minami et al. 2007; Walsh & Brodziak 2015). These models contain two components: first the probability of a 'false' zero count is modelled separately in a binomial model, and then the counts (i.e. expected number of bycatch species in a pot) are modelled as a separate process (which can also contain zero counts). Here a false zero means that a zero count was observed where there was a still a probability of an encounter. 'True' zeros are modelled as part of the count part of the model, typically by means of covariates. For example, where depth makes it very unlikely a species is encountered, a zero count may be modelled as a true zero. Covariates can be used to model both the binomial and count-based parts of the model. We tested both Poisson and negative binomial error distributions for the count part of the model in order to test whether accounting for overdispersion in the residuals (i.e. a negative binomial model) improved the model fits. Poisson models assume that the standard deviation of counts is equal to the mean, whereas negative binomial models include an additional factor (theta) that models overdispersion.

Models for bycatch estimation for each jurisdiction included main effects in the count model for fishing season, management zone, the interaction between zone and season and depth. Zone here refers to the management zones (Northern zone and Southern zone for SA, and Eastern zone and western zone for Victoria). It should also be noted that data confidentiality requirements in South Australia preclude presentation of data when collected from fewer than five fishers. For Tasmania we split the data into east and west coasts of Tasmania (longitude 146.5^0 used) to use a zone factor, as anecdotal evidence suggested potential differences in bycatch based on coast. Where depth was missing from the reported data, we used the mean reported depth for the recorded fishing block, with mean calculated over the entire time-series of commercial data available. For the zero inflated binomial part of the model we tested zone and depth as covariates. For the count part of the model we included fishing season, zone, a term for the interaction between season and zone and depth. This model structure makes intuitive sense as exploratory analysis (objective 1) indicated the probability of encountering certain species is likely to be lower in certain zones, and depth is known to be a key determinate for the distribution of marine species. Furthermore, data was available for these covariates in both the bycatch data and commercial effort data.

Three different model specifications were tested using both Poisson and negative binomial error distributions. An offset term was used for the number of pots, as the commercial data was typically by shot with shots having a differing number of pots. Models were compared using likelihood ratio tests, with the best model being used for subsequent prediction of total biomass across the commercial fishery. The zeroinfl function in the pscl package in R was used to fit all models (Jackman 2017). The three model specifications were:

$$\boldsymbol{count_i = f.season * zone + depth + offset(number\ of\ pots) \mid 1} \tag{1}$$

$$\boldsymbol{count_i = f.season * zone + depth + offset(number\ of\ pots) \mid zone} \tag{2}$$

$$\boldsymbol{count_i = f.season * zone + depth + offset(number\ of\ pots) \mid zone + depth} \tag{3}$$

Where $count_i$ denotes the count for the given species in the ith pot; $f.season * zone$ includes main effects for fishing season, zone and the interaction between fishing season and zone; $depth$ is the depth in metres for the pot or shot, $offset(number\ of\ pots)$ is an offset term used to account for the different number of pots; and **|** shows the separation between the two models, where everything to the left of the **|** symbol is the count model, and everything to the right is binomial model for false zeros. In model (1) the 1 in the binomial model indicates that only an intercept was fitted.

For Tasmania, there was a gap in the observer program and some spatial bias in sampling conducted under the observer program; however, there was also a time-series of research cruise data. Research cruise data is primarily collected with pots with closed escape gaps whereas the commercial fishery has open escape gaps. Therefore, to include the research data necessitated also modelling the effect of escape gaps in both the count and binomial parts of the model. Thus, all models above also included the term is_gap_closed in both parts of the model as a binary factor. Including this factor also allows for an exploration of the influence of escape gaps on bycatch for different species.

Model-based prediction of the total bycatch biomass for each of the species or grouped species was made for each fishing zone and season. It should be noted that total biomass here represents the estimated total bycatch, and not necessarily the total retained catch. Indeed, for some species such as Draughtboard Shark the majority of the total biomass caught is likely to be released in good condition. Estimated total biomass therefore represents the upper limit of what is likely to be caught and retained in a given season. Commercial effort data was available on a shot-by-shot basis, with depth recorded for the shot. Where depth information was missing the mean depth for the fishing block over the entire time-series was used. As the model was based on counts, predictions were in terms of expected counts within each shot. In order to scale up counts to biomass we multiplied the expected count by an estimated mean weight for the given species. The mean weight was calculated based on a mean length for the given species, and the 'a' and 'b' length-to-weight conversion parameters for that species (see Appendix A). Where possible, mean lengths were calculated from observer recorded length data for that species. Observer recorded lengths were available for a subset of the Tasmanian data as well as from 2005 - 2007 from Victoria, where observers recorded the length of all bycatch species. Where this data did not exist the expert opinion of two marine ecologists was sought and the average of these two lengths was used as the basis for calculating mean length. Detail of the mean lengths, standard deviations of these lengths and the number of observations used in calculating the mean length is provided in Appendix A.

In the Northern Zone management area in SA, pots used in the commercial fishery are required to have escape gaps. However, when observers are undertaking catch sampling operations in the Northern Zone, escape gaps are covered due to the historic focus on collecting size structure information (e.g. pre-recruit abundance) relating to SRL. Therefore scaling up estimates of bycatch in the Northern Zone using observer data would lead to positive biases in the estimates of total bycatch from this management area. In order to account for this bias, we applied the estimate of the escape gap effect for wrasse and leatherjacket species in the Northern Zone from research conducted pre and post introduction of mandatory escape gaps in 2003 (see Linnane et al. 2011). In the two years preceding the introduction of escape gaps (i.e. 2001 and 2002) the average CPUE of leatherjackets was 0.32 fish per potlift, whereas in 2003 it was 0.13, equating to a reduction of 59.4% for leatherjackets. For wrasse the average CPUE in 2001 and 2002 was 0.105, whereas in 2003 it was approximately 0.055 equating to a 47.6% reduction in bycatch. We use these estimates to adjust the estimated total of bycatch for leatherjacket and wrasse species from total estimates made for the Northern Zone. For Conger Eel an estimate of escape gap effects was not available for SA, so we used the estimate of the escape gap effect from the Tasmania analysis in order to adjust the estimated total bycatch in the Northern Zone in SA. We note that this method relies on the untested assumption that escape gap effects on bycatch of Conger Eel are equal in Tasmania and South Australia, however the method provides the most practical solution to estimating levels of bycatch of this species within the Northern Zone based on the information available.

We found that for a number of the key species there was insufficient data to model and predict total bycatch. This was due in some cases to low catch rates for those species (e.g. Draughtboard Sharks and Ocean Perch in SA), or due to the fact that bycatch was not identified to the species level. The lack of species level identification was particularly an issue for leatherjacket species where in many cases bycatch was just recorded as "leatherjacket". This was also the case for Blue-throat Wrasse, where bycatch was often recorded as just "wrasse". For leatherjackets, we modelled bycatch where there was sufficient data available from observers for Degen's Leatherjacket in Tasmania, and Horseshoe Leatherjacket in South Australia. We also modelled leatherjackets separately as a group in all jurisdictions. In Victoria there had been historic reporting of both bycatch and scalefish fishery catch as wrasse as a group. Expert opinion (Paul Hamer, VFA personal communication) and an analysis of the time-series of data indicated that "wrasse" was a mix of Blue-throat and Purple wrasse, with the proportion of Blue-throat Wrasse being typically around 90 percent. We therefore developed models for Blue-throat Wrasse in Victoria assuming that 90 percent of unidentified wrasse were Blue-throat Wrasse. This assumption was also used in Tasmania, where historical catches had also been reported as "wrasse", but data exploration also indicated ~ 90% of the commercial catch was Blue-throat Wrasse.

3.2 Data poor stock assessments

Data poor stock assessments: the Catch-MSY approach

Data poor stock assessments using the Catch-MSY approach (Matrtell & Froese 2013) were done for a subset of medium-risk species identified for further analysis in the previous section. The Catch-MSY approach was chosen as it is a relatively simple method that uses time series of catch for a species, which was the level of data available. The method is based on a Schaefer surplus production model with parameters r, the population growth rate, and K, the population carrying capacity or unfished biomass. The model requires potential initial and final values of the relative stock size (depletion levels) and a range of possible *r* and *K* values. The method assembles these prior levels of depletion and *r* and *K* by using ratios relative to the maximum catch, and then goes through year by year randomly taking *r-K* values from the parametric space containing these potential *r* and *K* values. They define the initial biomass from where the catches are taken and moving the stock dynamic forward by making biomass predictions. An extensive set of biomass trajectories, 50,000 in this case, are simulated by this process; however, not all of them are kept. All those trajectories that predict zero biomass or above the carrying capacity, *K* are discarded. The retained trajectories are used to estimate the mean values of B_0, MSY, etc. and make projections of biomass given different levels of catch.

The Catch-MSY approach has an underlying assumption that the catch is a direct reflection of the stock biomass. This assumption may be invalid, for example where catch is driven by market demand. Also, this method should ideally use the complete time series of catch for a species with at least 25 years of the catch history (Haddon *et al*., 2015), which was only available for Blue-throat Wrasse in VIC. Therefore, we apply this method with caution, and use the outputs as a relative reference to help facilitate the selection of reference points.

The Catch-MSY approach should include all sources of fishing mortality for a species when assessing sustainable future catch levels. We were therefore unable to apply this approach for all species due to a lack of species level information on catch from RL fishery bycatch or from other fishery sources. This was the case for all the leatherjacket species which are treated as a group in scalefish fishery assessments in each state and were typically also grouped in the bycatch data. Therefore, we analysed leatherjackets at the group level in VIC and TAS, but noting that improved data collection would allow for improved analysis in the future. For SA, data restrictions due to confidentiality, where < 5 fishers had operated in a management zone in a season, meant that there was not a sufficiently long time series of data to apply the Catch-MSY approach. Of the remaining short-listed species, data was only available for Blue-throat Wrasse in VIC and TAS. For the remaining species we analyse the time-series of estimated bycatch and make recommendations based on this analysis (see below).

For leatherjackets and Blue-throat Wrasse in VIC and TAS, data was collected on all sources of fishing mortality that were available in each jurisdiction, including estimated total biomasses of bycatch in the RL fishery from the previous section, catches from the scalefish fisheries, and estimated catches from the recreational fisheries (not available for Victoria). The intention was not to determine stock status for the species analysed or provide guidance for management of species which are targeted in other fisheries, but rather to carry out a risk analysis of potential increases in bycatch to aid in decision making and setting of reference points. In order to do this, we arbitrarily set a total allowable catch (TAC) of 90% of the Catch-MSY estimate. We then increased this TAC by adding additional catch which were fractions of average catch from the scalefish fishery catch in each state over the last 5 years in the time series. We tested fractions of 0%, 10%, 20%, 30%, 40%, 50% and 60%. The scalefish fishery catch was therefore used as a reference against which potential levels of bycatch could be assessed. The fate of bycatch is currently not well quantified, in particular the use of species for bait. As workshop input indicated that Wrasse and Leatherjackets are often used by fishers for bait, and are also susceptible to barotrauma, we set the PCM of leatherjackets and Blue-throat Wrasse at 75% for modelling purposes but note that improved estimates should be used in future modelling.

Modelling was done using the R package simpleSA (Haddon *et al.* 2019). Using the time series of catch data and the fractions of increase, trajectories were projected up until 2024 by using the coefficients r and K of the retained simulated trajectories of biomass. The projected biomass estimates were used to carry out a risk assessment, which involved the estimation of the probability of reaching the standard reference points of lower than 20% and greater than or equal to 40% of the estimated virgin biomass (B_0) as limit and target reference point respectively.

Alternative approaches for determining reference points for bycatch species

For species where there was insufficient data to conduct the Catch-MSY approach, we examined two criteria that have been suggested for monitoring bycatch in Commonwealth fisheries (see DAWE 2018):

- Catch/CPUE – greater than 20% (suggested) or 50% change compared to a maximum in the time-series in any one year.
- Catch/CPUE – statistically significant trend over the last 5 years (suggested) and over the entire time series of observer bycatch data.

We used the estimated catch (and associated CPUE) for each species and compared yearly estimates with the maximum estimate in the time series in order to examine variation in the time series. We chose to compare deviations compared to the maximum due to the considerable noise in the time series of data, making inter-annual variation much larger than the 20% or 50% cut-offs without normalisation.

Statistically significant trends in the last 5 years and the entire time-series of bycatch data was tested with a Generalized Additive Model (GAM). GAMs were used as they use a smoothing spline to estimate trends, which was considered necessary due to the noise in the data. We modelled the time-series of estimated total bycatch for each species using a "Tweedie" distribution, which is an appropriate distribution for biomass data (see Dunstan *et al.* 2013). The significance at $p = 0.05$ of the smooth of catch was used to test for significance. This criterion essentially tests whether the trend is statistically different to zero. We chose to test both the last 5 years (suggested under the Commonwealth guidelines) and the entire time-series, as a trend in a longer time-series is likely to be more evident, while the noise in a shorter 5-year time series makes trends more difficult to detect.

Results

1.1 Comparing the consistency of current bycatch reporting compared to international best practice data collection standards

Bycatch data across all jurisdictional fisheries were collected through a combination of four reporting systems:

1. Fisheries independent observer programs. Trained on board observers record bycatch numbers, but generally not the fate of bycatch (i.e. returned, retained for sale, used for bait etc). Interactions with TEPS are also recorded. The temporal and spatial scope of programs varied between jurisdictions.
2. Volunteer fishers (fishery dependent). Participating fishers record data on bycatch. Requirements and participation levels varied between jurisdictions. Current participation rates are generally low.
3. Fishery commercial logbooks (fishery dependent). Mandatory reporting of retained byproduct with specific rules relating to individual jurisdictions.
4. Threatened, Endangered or Protected Species (TEPS) interaction forms (fishery dependent). Mandatory forms used to report any interactions with TEPS.

Consistency of bycatch reporting in the SRLF compared to international best practice data collection standards was assessed using the US Tier Classification System in Table 3 (Appendix H, National Marine Fisheries Service, 2011).

Table 3. Tier criteria and classification scores for bycatch reporting in each jurisdiction of the SRLF. Scores given for each state were based on expert opinion of project members from each respective state using the criteria in Appendix H of the US National Marine Fisheries Bycatch Report (NMFS 2011).

Scoring criteria	Maximum possible points	SA	VIC	TAS
	Adequacy of Bycatch Data			
	Observer Data			
Longevity of observer program	5	5	4	4
Sampling frame	3	2	2	2
Sampling design				
- Sampling Vessels / Permits / Licenses	4	3	3	1
- Sampling Trips	4	3	3	3
- Sampling Hauls	4	3	3	3
Design implementation				
Spatial coverage	2	1	1	1
- Temporal coverage	2	1	1	1
- Vessel-selection bias	2	0	1	0
- Observer bias	2	2	2	2
Data-quality control	5	3	4	3
SECTION TOTAL	*33*	*23*	*24*	*20*
	Industry Bycatch Data			
SECTION TOTAL	*2*	*2*	*0*	*2*
	Supplemental Data			
Data available for extrapolation factors for unobserved components of the fishery	2	2	2	2
Data available for stratification	2	2	2	2
Data available for imputation	2	2	1	2
Data available for model covariates	2	2	2	2
Industry data verification	2	0	1	0
SECTION TOTAL	*10*	*8*	*8*	*8*
	Database / IT Considerations			
SECTION TOTAL	*3*	*3*	*0*	*3*
	Quality of the Bycatch Estimate			
	Analytical Approach			
Assumptions identified, tested and appropriate	10	3	2	3
Peer review / Publication				
- Observer program sampling design	4	2	2	0
- Analytical approach	4	0	0	2
Statistical bias of estimators	4	0	0	2
Measures of uncertainty	3	1	2	1
SECTION TOTAL	*25*	*6*	*6*	*8*
OVERALL SCORE	*73*	*42*	*38*	*41*
TIER		*2*	*2*	*2*

South Australia scored 42, Victoria 38 and Tasmania 41 out of a possible 73 points, placing all fisheries into Tier 2 in the classification scheme. This tier is defined as: Bycatch estimates were generally available. However, these estimates would have benefited from better data quality and/or analytical methods (such as improved sampling designs, increased coverage levels, and peer review of methods).

Scores for different aspects of each state's current bycatch data collection and management practices differed, but a number of common deficiencies were identified. These key areas for improvement are summarised in Box 1. In particular, the scoring in the 'observer data' section identified the lack of a well-designed randomised sampling design employed in the observer programs. Currently participation of vessels in the observer program is on a voluntary basis, with general low participation rates. For example, in Tasmania, the number of participating vessels decreased from 9 in 2010 to only 1 in the 2016 and 2017 seasons. In South Australia, the number of vessels participating in the observer program in 2016 was 9 out of 44 active vessels in the Northern Zone management area and 19 out of 163 active vessels in the Southern Zone management area. In Victoria during the 2016/17 fishing season (Nov – Sep), observers collected bycatch data from 12 vessels out of 42 active vessels in the Western Zone and from 5 vessels out of 21 active vessels in the Eastern Zone. The current design was therefore found to be lacking in terms of randomisation of sampling at various levels (e.g. seasons, vessels, hauls).

Box 1: Key areas for improvement in the SRLF bycatch monitoring program

1. Improvements in the sampling design of the observer program. Improved randomisation of vessels, trips and hauls to ensure a representative sample of the entire fishery is obtained each season.

2. Increased industry participation in the collection of bycatch and TEPS data. While programs exist for industry to collect bycatch data, participation rates are low and bycatch data is not consistently recorded.

3. Consistent data collection protocols and database management across all states. Current inconsistencies in the way data are collected, species named, weights recorded, TEPS interactions are recorded and database records are kept make cross-jurisdictional comparisons difficult.

4. Improved analytical approaches for bycatch data. A lack of historical focus on detailed analysis of bycatch data has meant a rigorous statistical approach has not been developed.

The "industry bycatch data" criterion in the TCS assesses whether industry bycatch data is available and is used in the bycatch estimation process. For the SRLF, the "2 pot" program in TAS and the voluntary pot sampling programs (3 pot) in NZ of SA and VIC all collect bycatch data from a small number of pots in each shot. Current participation rates in these programs are limited (e.g. only one fisher participates in the program in VIC), and the primary focus is on collecting size information for the target species rather than bycatch data.

The 'supplemental data' section of the TCS scores the availability and usefulness of additional bycatch data to draw wider inferences across the fishery. All states scored well in this regard, with programs such as research cruises which sample both fixed and random sites providing quality independent data to supplement observer and industry data.

This ‘database/IT considerations’ section of the TCS is focussed on scoring any constraints imposed by databases in the analytical approach. While no issues were noted at an individual state level, it was that there are inconsistencies in the way data is recorded across the three states.

Scores in the ‘analytical approach’ section were generally low and reflect the lack of historical focus on detailed bycatch estimation, monitoring and reporting in the SRLF.

1.2 Statistical exploration of the temporal and spatial consistency of bycatch data collection across the fishery and identification of key species, trends and patterns

South Australia

Analyses were done for each management area in SA: the NZ and SZ. In the NZRLF, fishery-independent observer data was available between the 2002 and 2017 fishing seasons. In the SZRLF, fishery-independent observer data were available between the 2000 and 2016 fishing seasons.

Table 4. Total number of pots in each zone of the SA RLF, percent of monitored pots with bycatch recorded and the number of monitored pots (in brackets).

Year	Total pots NZRLF	Total pots SZRLF	Percent of monitored pots with bycatch NZRLF (number of monitored pots)	Percent of monitored pots with bycatch SZRLF (number of monitored pots)
2001		1893		0% (2)
2002		122		6% (7)
2003	440	846	8% (34)	16% (132)
2004	1626	2310	14% (234)	10% (230)
2005	2167	1744	43% (922)	12% (212)
2006	2591	2568	40% (1037)	28% (714)
2007	516	885	22% (114)	20% (180)
2008	418	1511	42% (177)	15% (227)
2009	2681	2267	47% (1267)	20% (454)
2010	5096	2258	36% (1847)	20% (441)
2011	4479	3442	39% (1733)	19% (638)
2012	6892	2872	42% (2876)	15% (419)
2013	10992	2238	30% (3294)	21% (480)
2014	7557	2443	28% (2144)	16% (392)
2015	8243	2371	41% (3372)	23% (553)
2016	7171	2232	41% (2944)	22% (499)
2017	4107	1113	22% (921)	17% (186)

The total number of potlifts observed in each management area was variable between years (Table 4) and under sampling of bycatch was evident in early fishing seasons of the observer program (e.g. SZRLF - 2000 and 2001). The number of pots with bycatch recorded by observers for both the NZRLF and SZRLF pooled was also variable between months and years (Figure 2).

Data confidentiality for <5 licence holders precludes presentation of observer data at the spatial resolution at which they are provided (lat/long), however some general patterns were evident from preliminary mapping. In the SZRLF, observer coverage was spread relatively evenly among all marine fishing areas (MFAs) between 2003 and 2017. In the NZRLF, observer coverage varied spatially between years. In most years, observer coverage was limited to MFAs located on southern Eyre Peninsula, southern Yorke Peninsula and

Kangaroo Island. In 2005, 2012, 2013, 2014 and 2016, observer coverage was more evenly spread across the NZRLF, and included more MFAs located in the western part of the fishery (eastern Great Australian Bight).

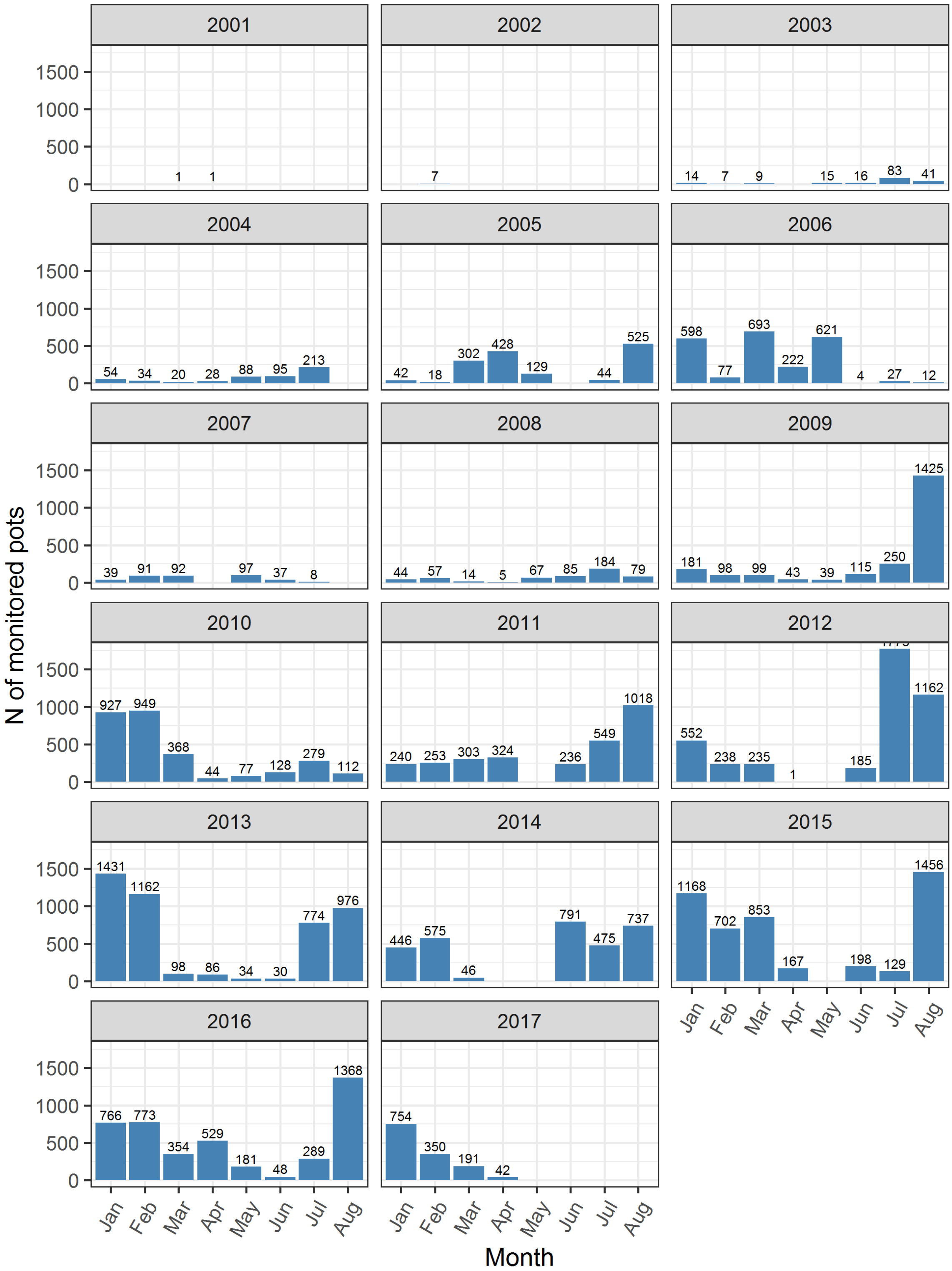

Figure 2. Number of pots with bycatch recorded each month in both the SZRLF and NZRLF in SA between 2000 and 2017 (calendar year).

Indices of relative importance for bycatch species were calculated for the NZRLF and SZRLF for all years pooled. In the NZRLF between 2003 and 2017, leatherjacket species and in particular Horseshoe Leatherjacket (*Meuschenia hippocrepis*), comprised the largest proportion of bycatch species by weight and number (Figure 2). Blue-throat Wrasse and Velvet Crabs (*Nectocarcinus integrifrons*) comprised the second and third largest percentage of bycatch observed by weight and by number, respectively. Other (unidentified) leatherjacket species and Ocean Jacket (*Nelusetta agraudi*) were also relatively important bycatch species.

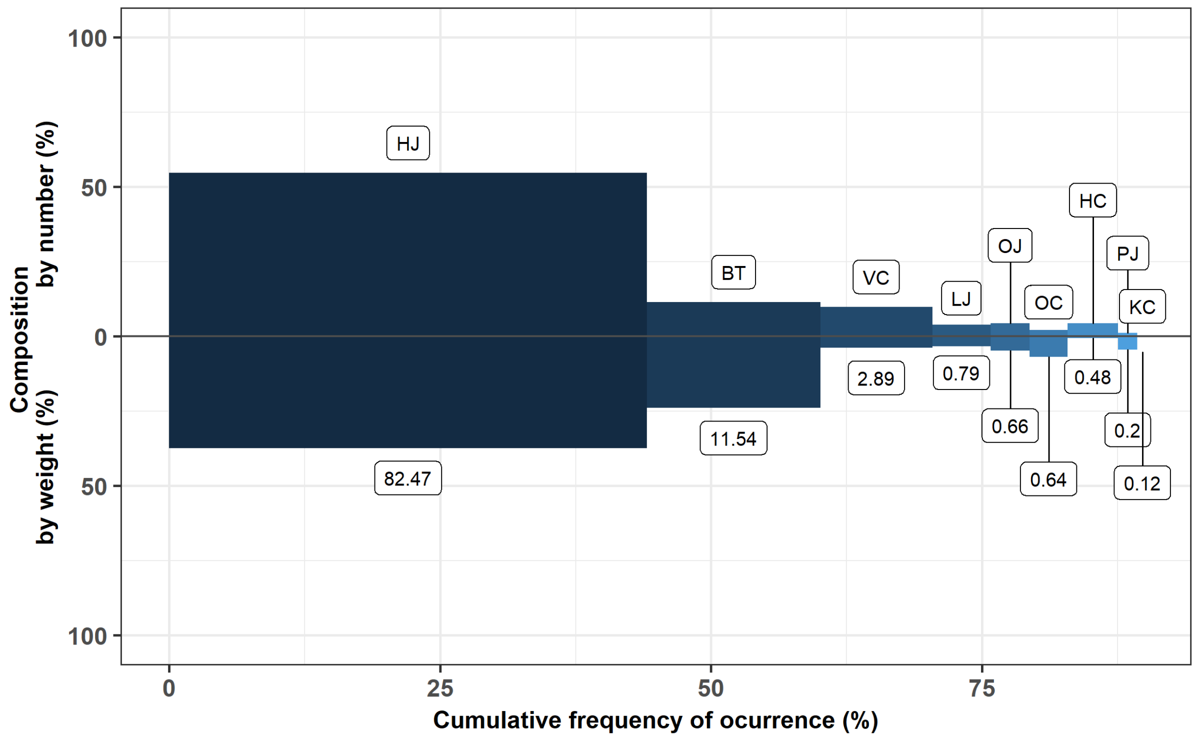

Figure 3. Index of relative importance for bycatch species reported in the NZRLF between 2003 and 2017. HJ = Horseshoe Leatherjacket, BT = Blue-throat Wrasse, VC = Velvet Crab, LJ = Leatherjacket (various species), OJ = Ocean Jacket, OC = Octopus, HC = Hermit Crab, P = Port Jackson Shark, KC = Giant Crab.

In the SZRLF between 2001 and 2017, Hermit Crabs (*Paguristes sp*.) (HC) comprised the largest percentage of bycatch observed by weight and by number (Figure 3). Octopus (OC) and leatherjacket species (LJ) comprised the second and third largest percentage of bycatch observed by weight and by number, respectively (Figure 4).

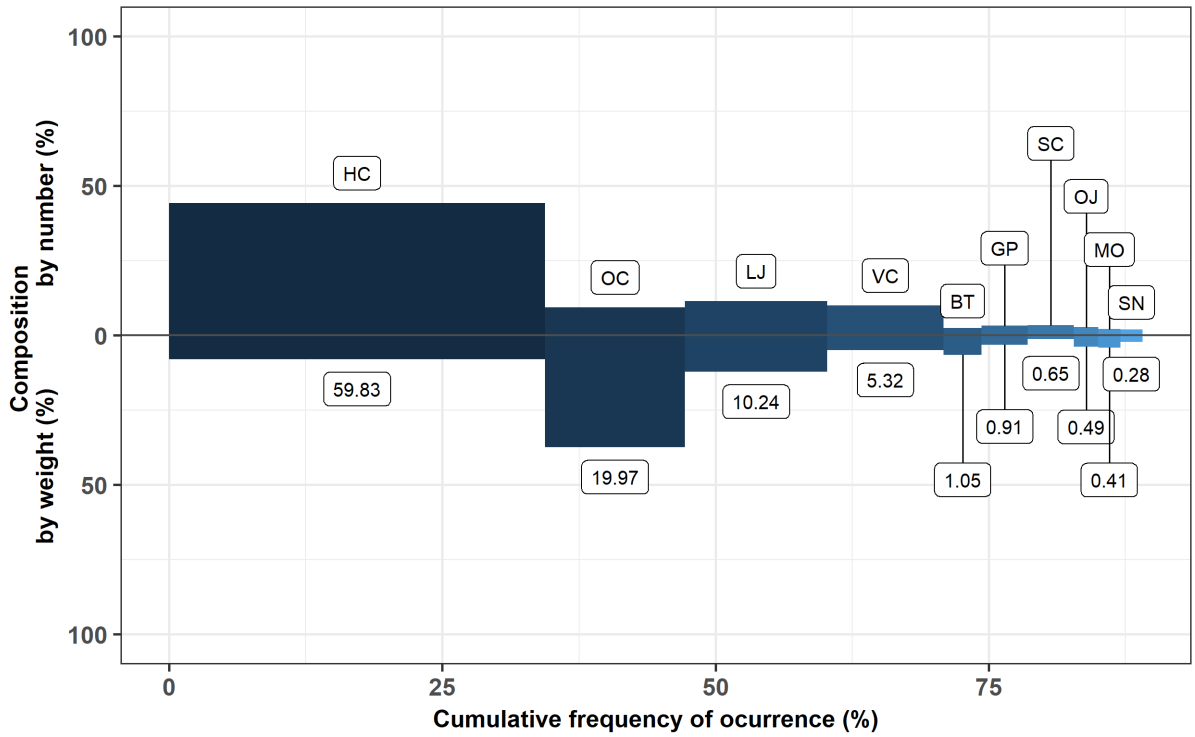

Figure 4. Index of relative importance for bycatch species reported in the SZRLF between 2001 and 2017. HC = Hermit Crab, OC = Octopus, LJ = Leatherjacket (various species), VC = Velvet Crab, BT = Blue throat Wrasse, GP = Gurnard Perch, SC = Slimy Cod, OJ = Ocean Jacket, MO = Morwong (various species), SN = Snapper.

In the NZRLF, the fishing effort (potlifts): bycatch ratio was 1: 0.64 ± 0.02 (n=912) when including undersized RL. This equates to a total of 0.64kg of bycatch per potlift. When excluding undersized RL, the effort:bycatch ratio was 1: 0.47 ± 0.02 (n=912), equating to 0.46 kg of bycatch per potlift. Expanding this to the total commercial effort across the NZRLF, a mean total of 304.7 ± 25.3 tons of bycatch is caught in a season when including undersized RL and 223.4 ± 21.6 tons when excluding undersized RL. The reported commercial catch in the NZRLF for the 2017/18 season was 310 t, and therefore the estimated bycatch represents 49.6% of the total catch when including undersized RL, and 36.3% excluding undersized RL for that season.

In the SZRLF, the fishing effort (potlifts): bycatch ratio was 1: 0.65 ± 0.02 (n=530) when including undersized RL. This equates to a total of 0.65 kg of bycatch per potlift. When excluding undersized RL, the effort:bycatch ratio was 1: 0.15 ± 0.01 (n=912), equating to 0.15 kg of bycatch per potlift. Expanding this to the total commercial effort across the SZRLF, a mean total of 828.8 ± 60.6 tons of bycatch is caught in a season when including undersized

RL and 196.8 ± 29.6 tons when excluding undersized RL. The reported commercial catch in the SZRLF for the 2017/18 season was 1245.7 t, and therefore the estimated bycatch represents 40.0% of catch including undersized RL, and 9.5% excluding undersized RL for that season.

Victoria

Analyses were done for each management area in VIC: the Western Zone (WZ) and Eastern Zone (EZ). Fisheries independent observer data was available for years between 2005 and 2017, with the exception of 2013 in the EZ, where no sampling occurred (Table 5). The number of monitored pots varied throughout the year, with more pots monitored in earlier years in the time-series of data (Figure 5).

Maps showing spatial and temporal coverage of the VIC on-board observer program were withheld from this report due to confidentiality (as per SA - the VFA is not permitted to make public fisheries data collected from less than five fishers). Generally, the observer coverage is fairly well represented in the Portland, Warrnambool and Apollo Bay regions in the WZ and in the Queenscliff region in the EZ. Less well represented are the San Remo and Lakes Entrance Regions, further to the east where fewer operators and lower catches make it more difficult to collect data.

Table 5. Total number of pots sampled and total number of pots containing bycatch as observed by on-board observers in the WZ and EZ of the Victorian rock lobster fishery between 2005 and 2017.

Years	Western Zone		Eastern Zone	
	Total pots sampled	Total pots with bycatch	Total pots sampled	Total pots with bycatch
2005	10450	1066	4065	556
2006	14742	1275	3639	381
2007	13234	1163	2329	192
2008	13502	1110	535	63
2009	10636	959	1719	188
2010	7615	472	43	3
2011	7163	548	904	78
2012	2021	170	500	60
2013	2884	192		
2014	2327	121	663	89
2015	3043	130	1137	235
2016	2037	221	701	157
2017	2654	215	1306	206

Figure 5. Number of observers monitored pots each month in both the WZ and EZ of the VRLF between 2005 and 2017 (calendar year).

Important bycatch species, as measured by the IRI, varied between the two management zones in VIC (Figures 6 and 7). In the WZ, Hermit Crabs (*Paguristes sp*.), Velvet Crabs (*Nectocarcinus integrifrons*) and Octopus (various species) were the three most important species. In the EZ, leatherjackets (various species), Draughtboard Sharks (*Cephaloscyllium laticeps*), Hermit Crabs (*Paguristes sp*.) and Port Jackson Sharks (*Heterodontus portusjacksoni*) were the most important bycatch species.

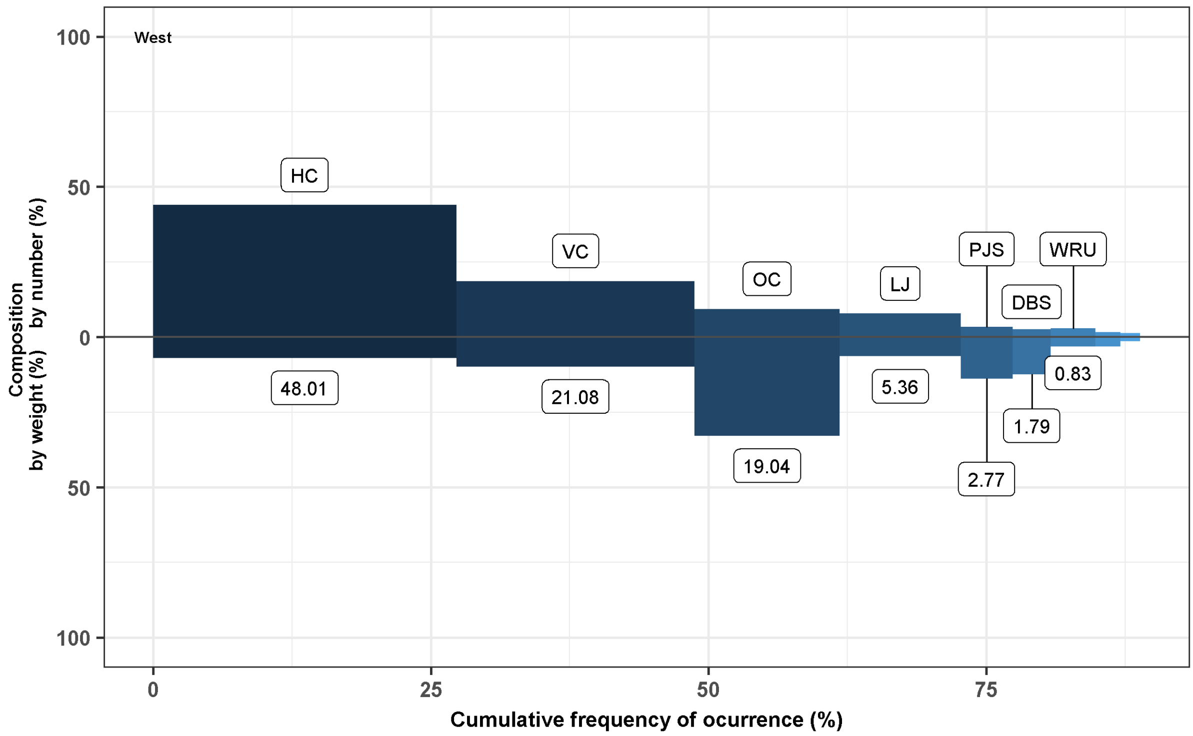

Figure 6. Index of relative importance for bycatch species reported in the WZ of the VRLF between 2005 and 2017. HC = Hermit Crab, VC = Velvet Crab, OC = Octopus, LJ = Leatherjacket, PJS = Port Jackson Shark, DBS = Draughtboard Shark, WRU = Wrasse (unidentified).

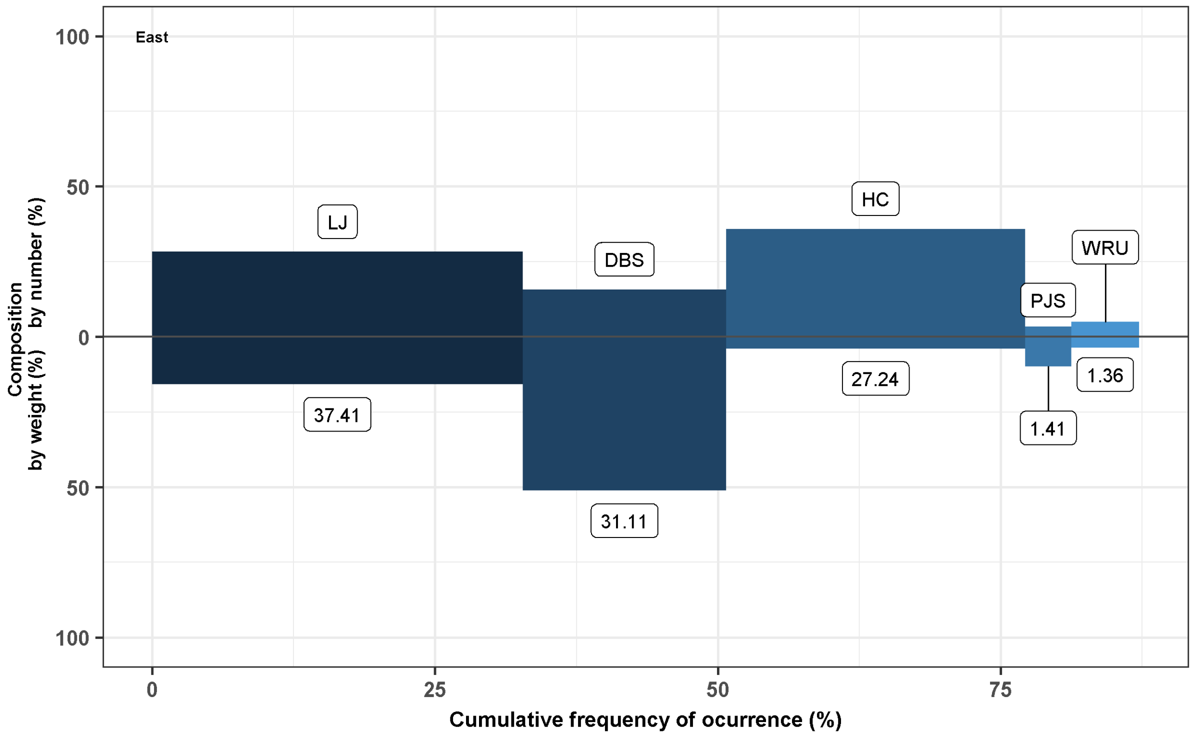

Figure 7. Index of relative importance for bycatch species reported in the EZ of the VRLF between 2005 and 2017. LJ = Leatherjacket (various species), DBS = Draughtboard Shark, HC = Hermit Crab, PJS = Port Jackson Shark, WRU = Wrasse (unidentified).

In the EZ, the fishing effort (potlifts): bycatch ratio was 1: 0.37 ± 0.01 (n=383) when including undersized RL. This equates to a total of 0.37 kg of bycatch per potlift. When excluding undersized RL, the effort:bycatch ratio was 1: 0.27 ± 0.01 (n=383), equating to 0.27 kg of bycatch per potlift. Expanding this to the total commercial effort across the EZ, a mean total of 47.9 ± 3.8 tons of bycatch is caught in a season when including undersized RL and 35.8 ± 3.3 tons when excluding undersized RL. The reported commercial catch in the EZ for the 2015/16 season was 59 t, and therefore the estimated bycatch represents 44.8% of the total catch when including undersized RL, and 33.5% when excluding undersized RL for that season.

In the WZ, the fishing effort (potlifts): bycatch ratio was 1: 0.55 ± 0.04 (n=1445) when including undersized RL. This equates to a total of 0.55 kg of bycatch per potlift. When excluding undersized RL, the effort:bycatch ratio was 1: 0.12 ± 0.01 (n=1445), equating to 0.12 kg of bycatch per potlift. Expanding this to the total commercial effort across the WZ, a mean total of 323.4 ± 28.0 tons of bycatch is caught in a season when including undersized RL and 75.3 ± 13.5 tons when excluding undersized RL. The reported commercial catch in the WZ for the 2015/16 season was 230 t, and therefore the estimated bycatch represents 58.4% of the total when catch including undersized RL, and 13.6% when excluding undersized RL for that season.

Tasmania

Bycatch data was available from the TAS fishery between 1992 and 2016. Collection of bycatch data in TAS varied in time and space, with some years having excellent coverage of all major fishery zones (NW, NE, SW, SE; e.g. 2004-2006 and 2008-2012), but other years having poor spatial coverage, particularly in earlier years of the observer program Figures 8 and 9). The number of monitored pots across months within years showed relatively consistent sampling across the whole fishing season (Figures 10 and 11).

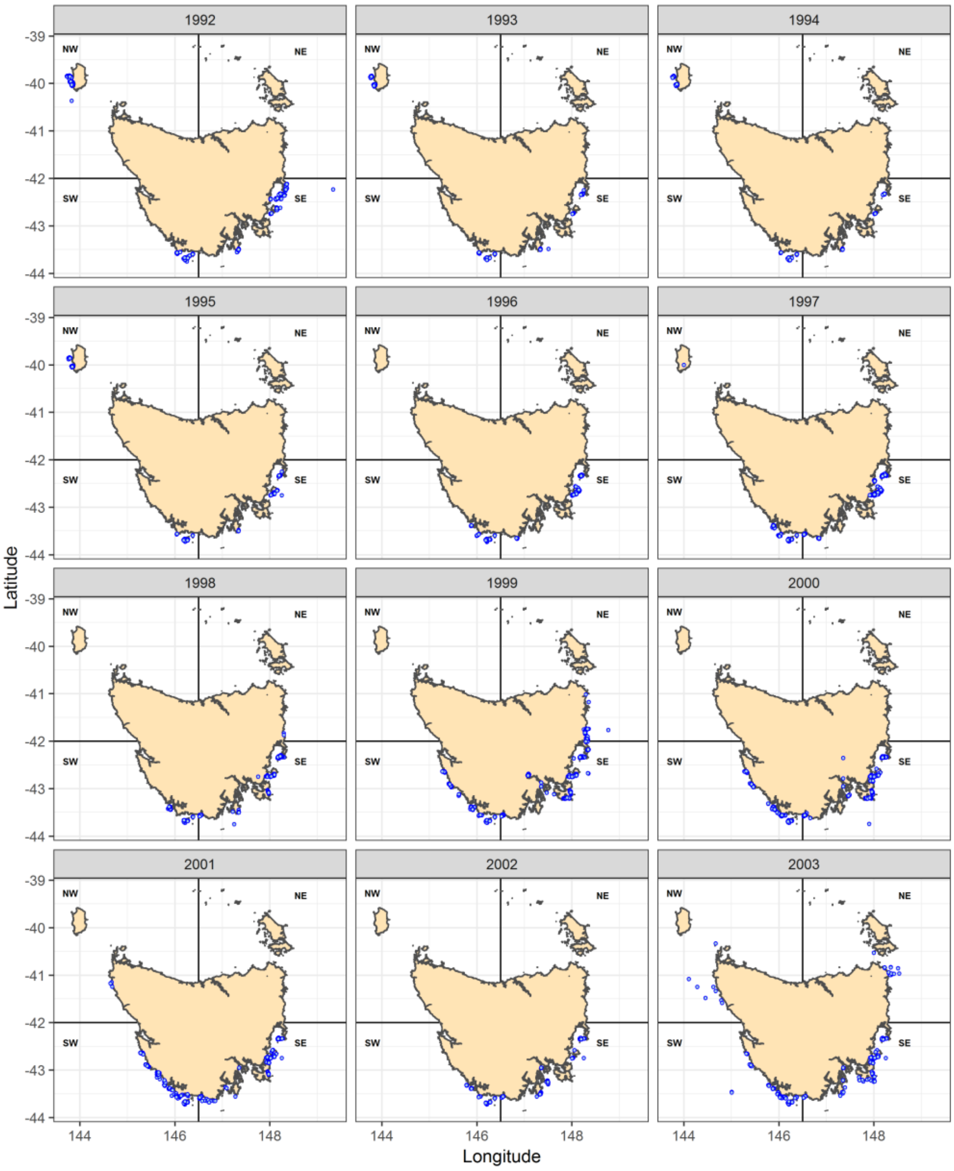

Figure 8. Spatial coverage of observer collected bycatch data in TAS 1992 – 2003.

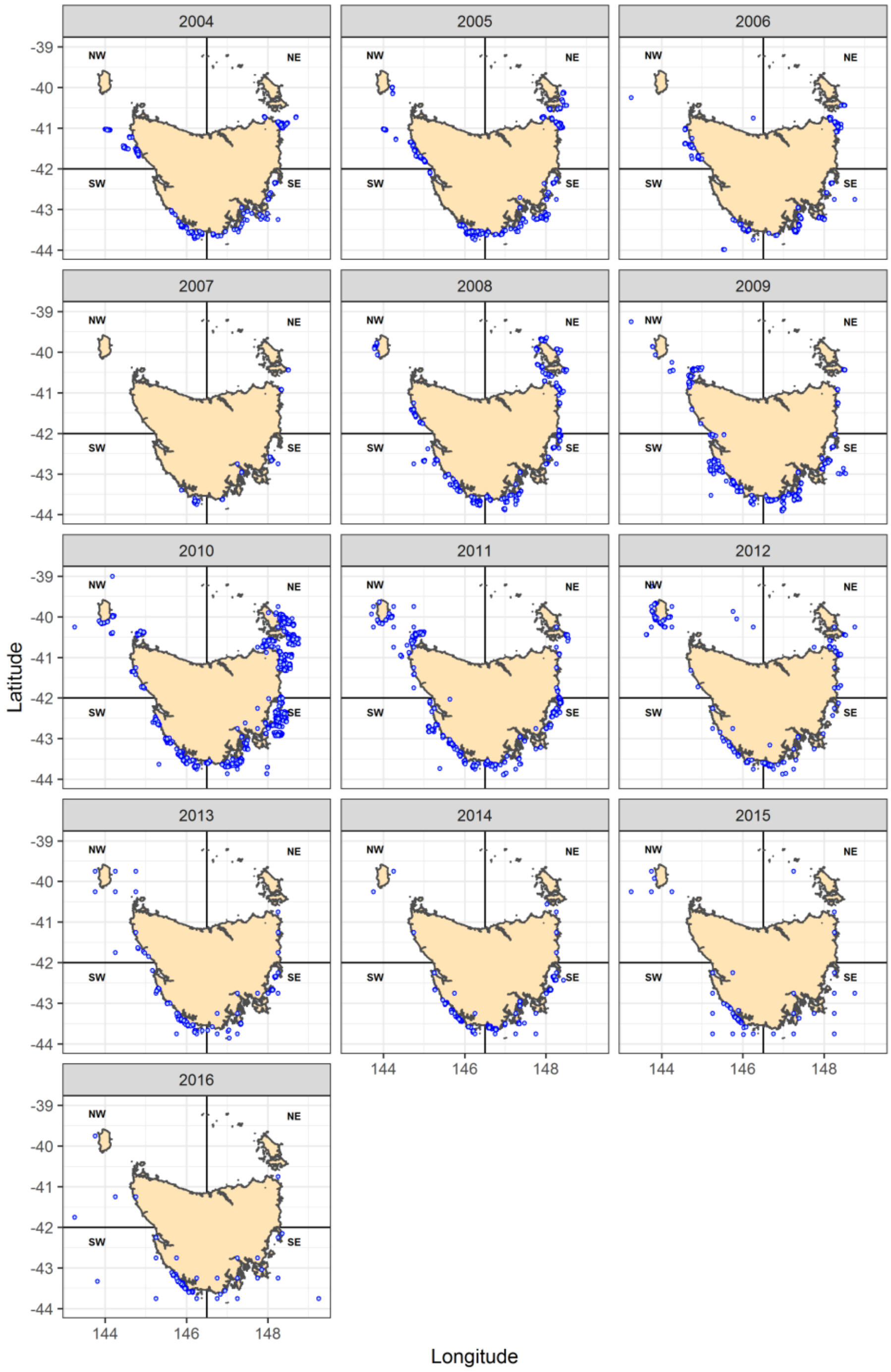

Figure 9. Spatial coverage of observer collected bycatch data in TAS 2004 – 2016.

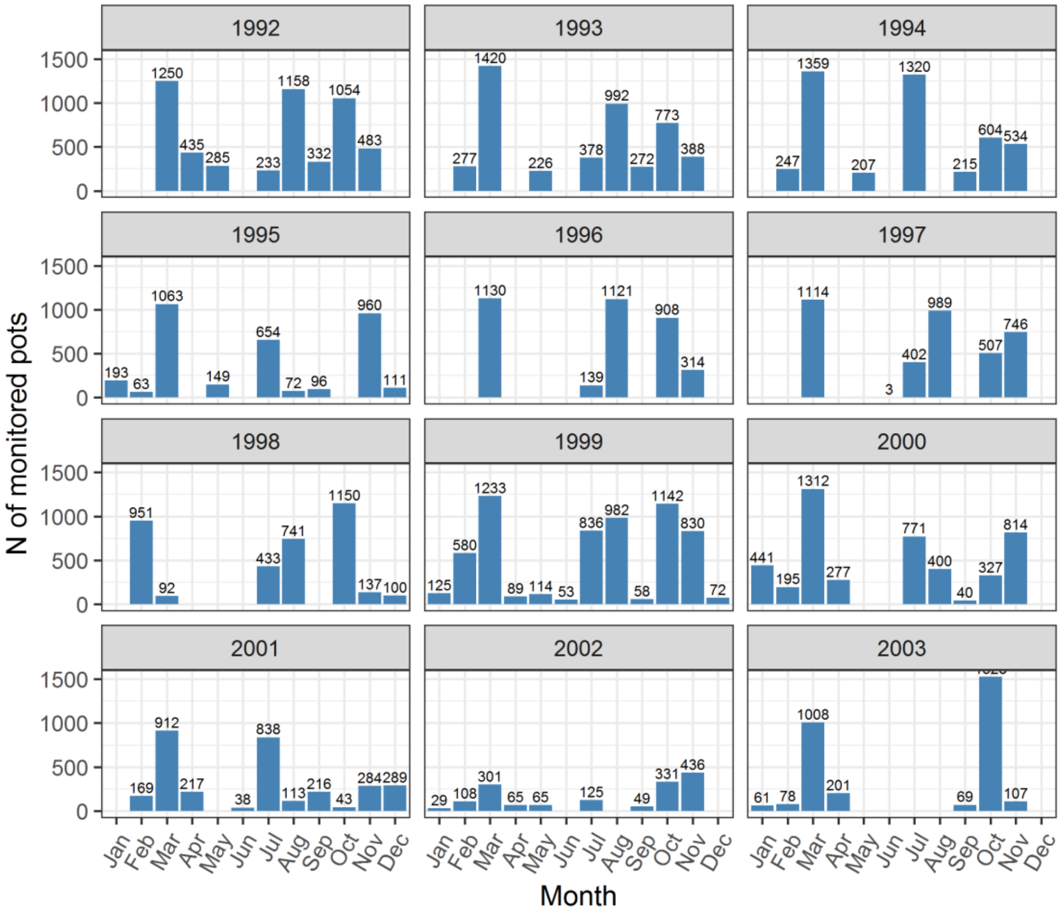

Figure 10. Number of monitored pots with bycatch data by month in TAS from 1992 – 2003.

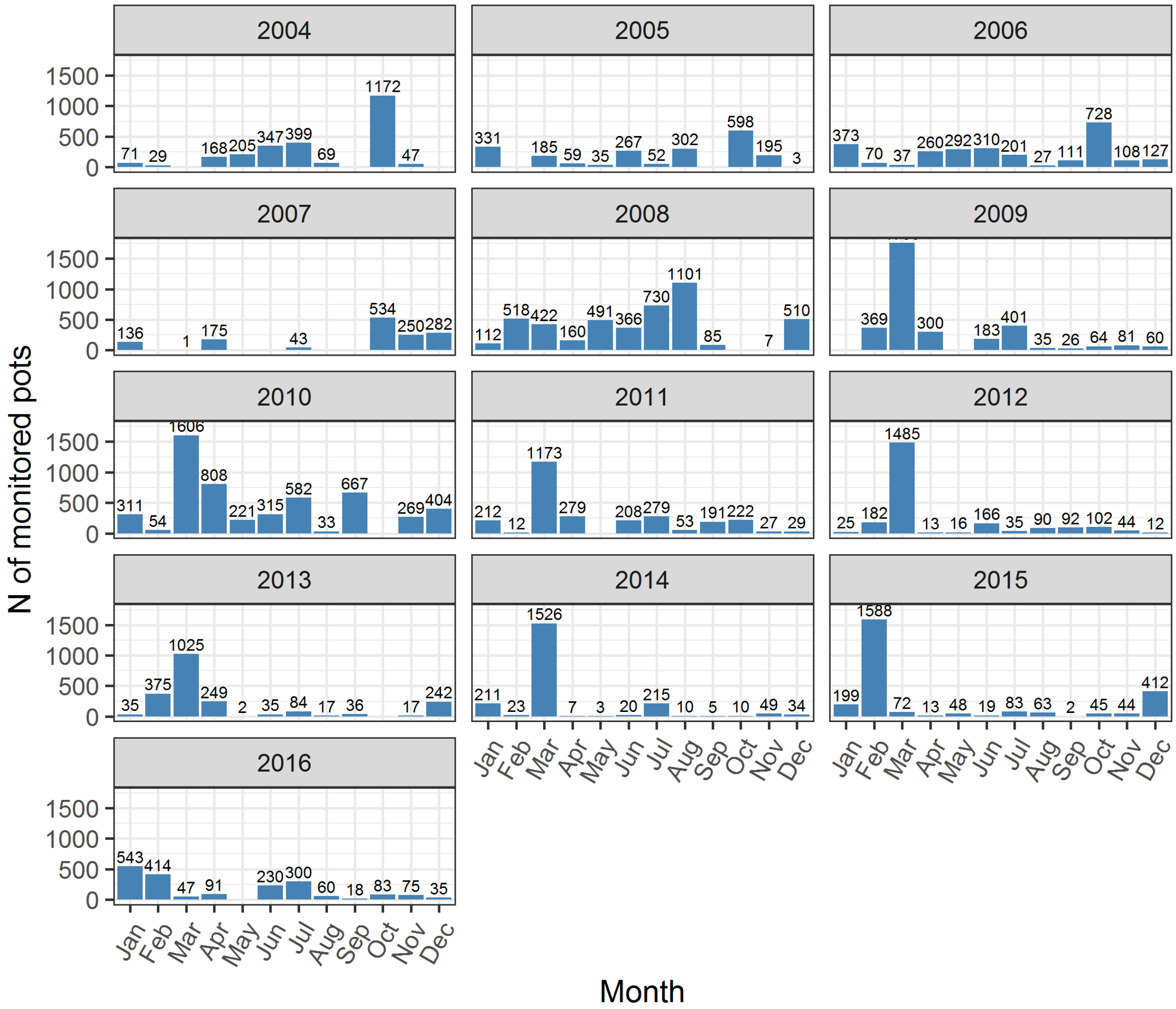

Figure 11. Number of monitored pots with bycatch data by month in TAS from 2004 – 2016.

Important bycatch species, as measured by the IRI and considering the whole of the TAS fishery, were dominated by Hermit Crabs (*Paguristes sp*.) and Draughtboard Sharks (*Cephaloscyllium laticeps*), making up 63.55 and 34.25% cumulative frequency of occurrence, respectively (Figure 9). Octopus, leatherjacket (various species) and Velvet Crab were also a relatively important component of the bycatch (Figure 12).

In TAS, the fishing effort (potlifts):bycatch ratio was 1: 2.71 ± 0.13 (n=1300) when including undersized RL. This equates to a total of 2.71 kg of bycatch per potlift. When excluding undersized RL, the effort:bycatch ratio was 1: 0.50 ± 0.02 (n=1300), equating to 0.50 kg of bycatch per potlift. Expanding this to the total commercial effort, a mean total of 3638.0 ± 45.5 tons of bycatch is caught in a season when including undersized RL and 668.5 ± 19.5 tons when excluding undersized RL. The reported commercial catch in the TRLF for the 2018/19 season was 1033.74 t, and therefore the estimated bycatch represents 77.9% of the total catch when including undersized RL, and 14.3% when excluding undersized RL for that season.

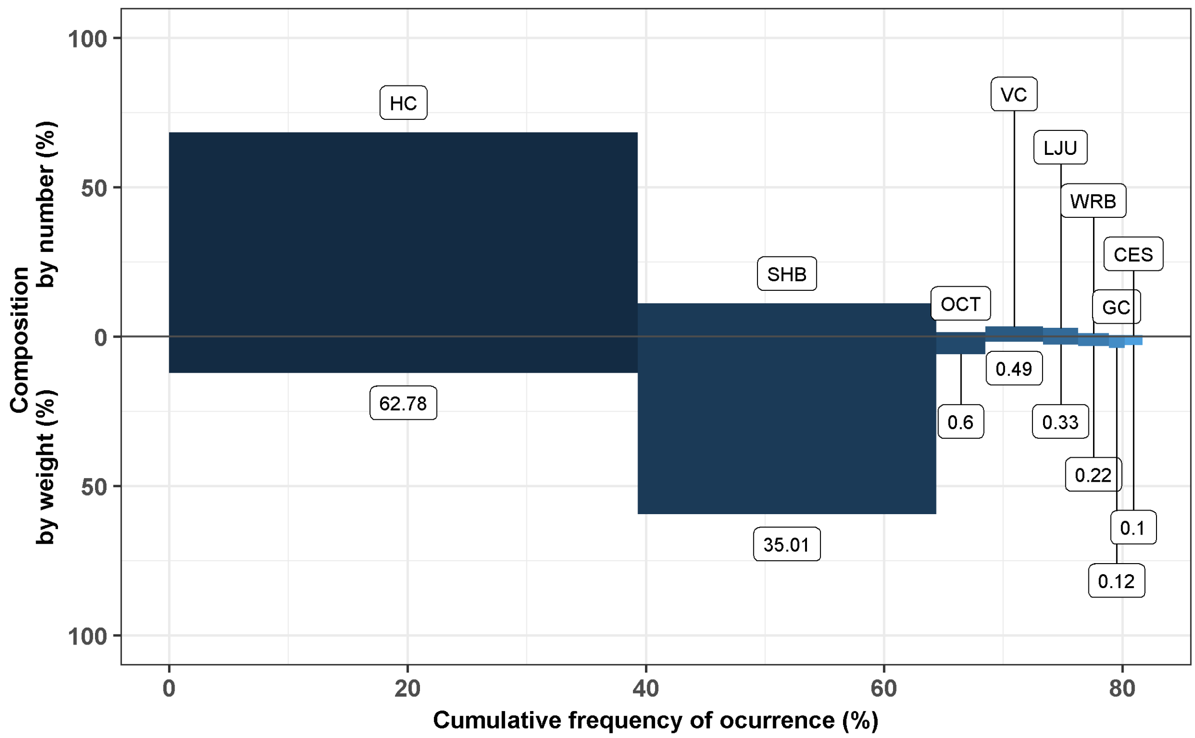

Figure 12. Index of relative importance for bycatch species reported in the Tasmanian Rock Lobster fishery; Hermit Crab (HC), Draughtboard Shark (SHB), Octopus (OCT), Velvet Crab (VC), Leatherjacket (unidentified) (LJU), Blue-throat Wrasse (WRB), Giant crab (GC) and Conger Eel (CES).

2. Results of the risk assessment for bycatch, byproduct and TEP species across the Southern Rock Lobster fishery

Byproduct species

There were 75 byproduct species assessed: 53 teleosts, 10 chondrichthyans, 7 molluscs and 5 crustaceans. Of these, no species was ranked as high risk, 18 were given a medium risk rating and 57 a low risk rating (Figure 13 and Appendix B). There was an average of 1.21 missing attributes out of 11 for all byproduct species. Missing attributes were all related to life history parameters (see the productivity attributes in Table 1), and therefore resulted in higher risk on the productivity axis for those species with a large number (> 3) of missing attributes. The full list of byproduct species and their risk scores is given in Appendix B.

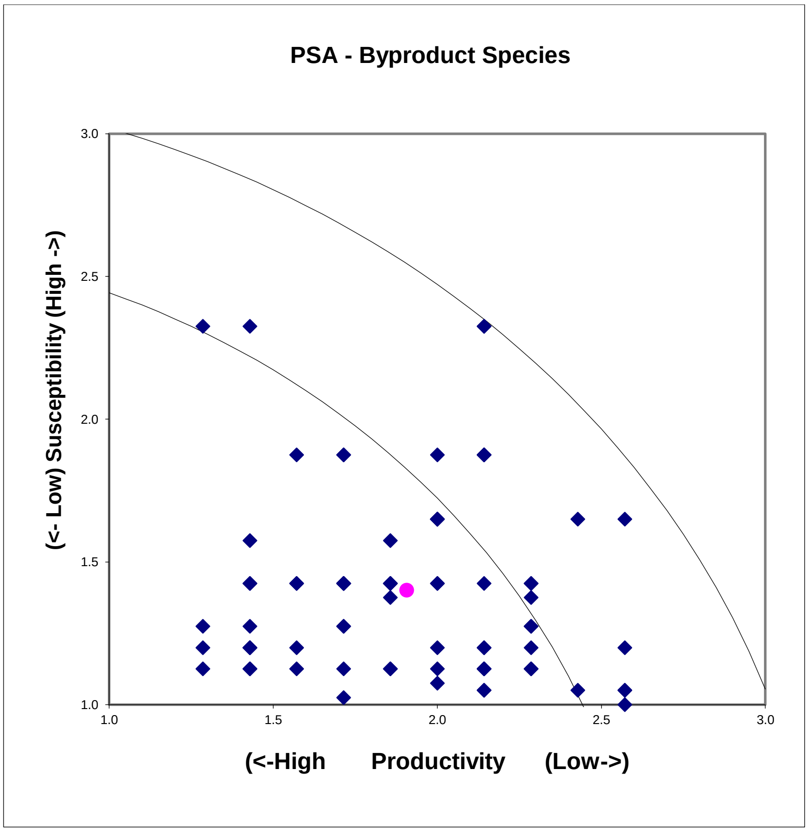

Figure 13. Results of PSA risk assessment for all 75 byproduct species across the SRLF. Blue points represent individual species and their score on the productivity and susceptibility axes, but note that some points may represent several species with the same score. The pink point is the average for all byproduct species. Bands denote overall risk from low (left), to medium (middle) to high (right).

Bycatch species

There were 42 bycatch species assessed: 4 chondrichthyan species, 25 teleost species, 5 crustaceans, 3 echinoderms, 3 molluscs, and 2 marine birds. No species were ranked as high risk, 4 species were ranked as medium risk and 38 species were ranked as low risk (Figure 14 and Appendix B). There was a significant amount of missing species attribute information for discard species, with an average of 2.29 missing attributes out of 11, with all missing attributes being related to productivity (Table 1). The full list of bycatch species and their risk scores is given in Appendix B.

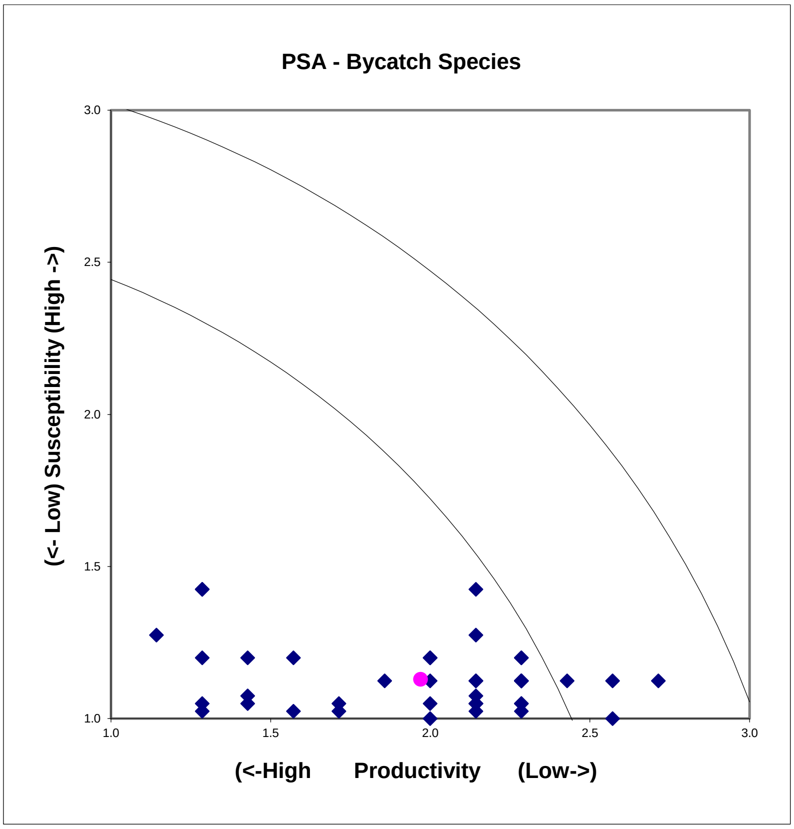

Figure 14. Results of PSA risk assessment for all 42 bycatch species across the SRLF. Blue points represent individual species and their score on the productivity and susceptibility axes, but note that some points may represent several species with the same score. The pink point is the average for all bycatch species. Bands denote overall risk from low (left), to medium (middle) to high (right).

TEPS

There were 134 TEPS assessed: 36 marine mammals, 60 marine birds, 3 marine reptiles, 3 chondrichthyans and 32 teleosts. No species were ranked as high risk, 72 species were ranked as medium risk and 62 species were ranked as low risk (Figure 15 and Appendix B). There was an average of 0.78 missing attributes. The full list of TEPS and their risk scores is given in Appendix B.

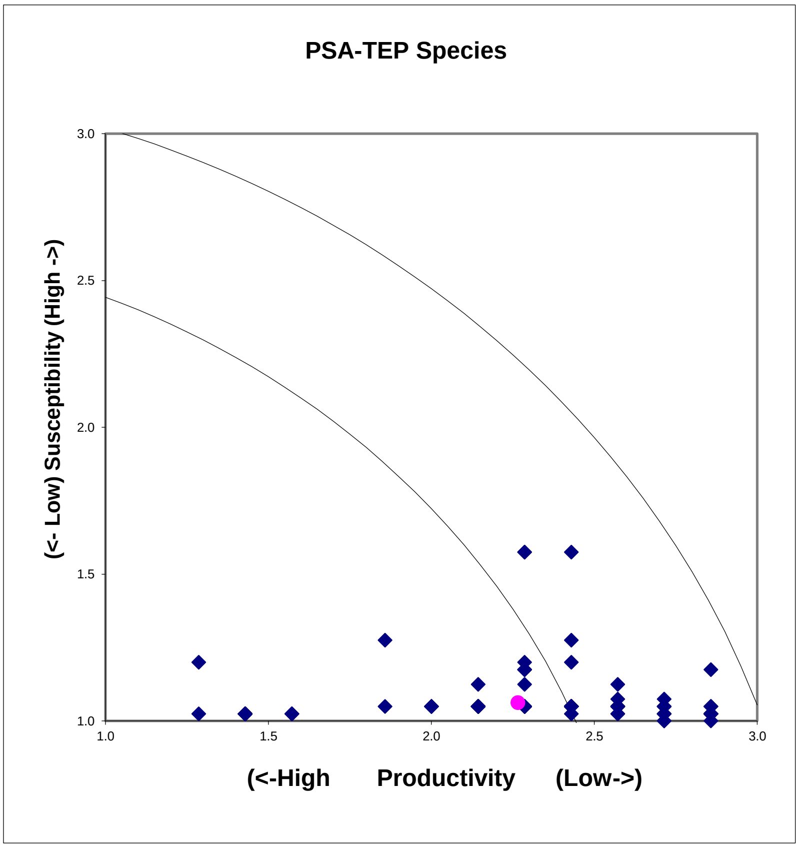

Figure 15. Results of PSA risk assessment for all 134 TEPS across the SRLF. Blue points represent individual species and their score on the productivity and susceptibility axes, but note that some points may represent several species with the same score. The pink point is the average for all TEPS. Bands denote overall risk from low (left), to medium (middle) to high (right).

Due to observer programs running across all jurisdictions, information regarding encounter rates and types of interaction with TEPS species were available from both historical database records and through direct input from observers during the workshops held in each jurisdiction. Species that were noted to have a high availability (observed on greater than 2 out of 3 trips) were seals (all species, Australian fur seals, Long Nosed Fur Seals and Australian Sea Lions and not distinguished) and Shy Albatross which were noted by a TAS observer to be present during most trips. Dolphins (species not distinguished) and Australasian gannets (*Morus serrator*) were noted to be present between one-third and two-thirds of trips resulting in medium availability risks.

Considering when a TEPS were observed, observers confirmed that encounters with gear were very rare, resulting in low risks scores in terms of encounterability for all TEPS. Also, selectivity of the gear for all TEPS was considered to be low. Potential issues raised during the workshops, and information from mandatory fishery dependent TEPS logbooks identified the main potential risks to TEPS to be:

- Potential for marine mammal entanglements in pot lines. The major risk identified was for cetacean species due to their large size.
- Potential for marine reptile (primarily Leatherback Turtles) entanglements in pot lines.
- Potential for juvenile seals and sea lions to become trapped in pots and drown (where seal exclusion devices on pots are not mandated).
- Potential for entrapment and drowning of diving birds in pots.
- Potential for birds to collide with fishing vessels at night due to vessel light spill.

Where any of these encounters did occur, there was a potential for mortality, with a high likelihood of mortality for any individual trapped in a pot. For entanglements, there was a potential to free the entangled animal, with reports of this occurring on several occasions. Post-capture mortality scores were therefore left as high for the majority of TEPS species in order to be precautionary.

From an ecological risk assessment perspective, no TEPS was considered to be high risk primarily due to the very low encounter rates of TEPS with the gear. It was noted that improved record keeping (both fishery dependent and fishery independent) relating to sightings of TEPS would help in determining whether the potential for interactions may be increasing due to increasing population sizes of some species. Also, improved reporting in this respect would aid in determining spatial and temporal patterns and the potential for interactions with gear.

Priority species for further analysis

Based on the PSA risk scores, 10 byproduct species were prioritised as requiring a more detailed quantitative analysis of their status (Table 6). Susceptibility scores for the species ranked as medium risk in this table were all greater than average (1.5 out of 3). Reasoning for the higher than average susceptibility for these species is given below. Medium risk TEPS species were not subjected to further quantitative analysis as there were very few encounters resulting in PCM, and medium risk scores were primarily related to life history (i.e. productivity) attributes rather than impacts from fishing activities (i.e. susceptibility).

Table 6. Short-list of 10 species identified for further research due to potential risk scores from the PSA. The number of missing attributes is shown because a higher number of missing attributes result in higher productivity risk scores and thus higher overall risk.

Common name	Scientific name	Overall PSA ranking	No. missing attributes
Draughtboard Shark	*Cephaloscyllium laticeps*	Medium	2
Six Spine Leatherjacket	*Meuschenia freycineti*	Medium	3
Degen's Leatherjacket	*Thamnaconus degeni*	Medium	3
Horseshoe Leatherjacket	*Meuschenia hippocrepis*	Medium	3
Mosaic Leatherjacket	*Eubalichthys mosaicus*	Medium	3
Gunn's Leatherjacket	*Eubalichthys gunnii*	Medium	3
Toothbrush Leatherjacket	*Acanthaluteres vittiger*	Medium	1
Ocean Perch	*Helicolenus percoides*	Medium	1
Blue-throat Wrasse	*Notolabrus tetricus*	Medium	0
Southern Conger Eel	*Conger verreauxi*	Medium	2

Draughtboard Shark had a high productivity risk score, which is typical for chondrichthyan species due to their slow growth and low reproductive output. The Draughtboard Shark also had 2 missing life history attributes related to age at maturity and maximum age. This species had a medium score on the susceptibility scale (1.65 out of 3) due to relatively high overlap in range and habitat/depth preference with the fishery and being strongly attracted to bait. Draughtboard Sharks are a dominant bycatch species in TAS and eastern VIC (see objective 1 results). While there was historical reporting of this species being retained as byproduct, overall retained captures are likely to be a small proportion of bycatch and the species is likely to have high survival rates when released (e.g. see Awruch et al. 2012).

All the leatherjacket species in Table 6 had a high overlap in distribution with the fishery effort and were associated with shallower rocky reefs, resulting in high scores for availability and encounterability. Leatherjacket species were found to be important bycatch/byproduct species across the entire fishery (see Objective 1 of this project) indicating at least some selectivity of the gear resulting in medium risk scores. Also, workshops conducted in each jurisdiction identified that leatherjacket species have high susceptibility to barotrauma and are often kept for bait or sale resulting in a high PCM risk. Many of the leatherjacket species in Table 6 also have a lack in information for 3 life history parameters, resulting in higher than expected productivity risk scores. Improved life history information for these species may result in lower overall risk scores in future PSA analysis.

Ocean Perch and Blue-throat Wrasse (Table 6) had a high overlap in distribution with the fishery effort and were associated with shallower rocky reefs, resulting in high scores for availability and encounterability. These species were all noted to be particularly susceptible to barotrauma and may be kept for use as bait thereby resulting in high PCM scores. Perch and wrasse species were found to be relatively important bycatch/byproduct species across the entire fishery (see Objective 1 of this project) indicating at least some selectivity of the gear and resulting in medium risk scores.

Southern Conger Eel had information missing relating to 2 life history parameters. Whilst this results in a precautionary high-risk productivity score, this species is known to be slower growing and likely to have relatively low abundance compared to other reef fish. It is likely to be attracted to bait in pots and was found to be an important bycatch species (see Objective 1) resulting in a medium selectivity risk score. It has also been historically reported as a byproduct species resulting in a precautionary medium PCM risk score.

3.1 Scaling up of biomass estimates to the commercial fishery

For key identified scalefish species, we compare estimated bycatch amounts relative to reported catches of these species when targeted within a fishery over the same period where available. We do this in order to examine estimated bycatch amounts relative to a targeted fishery where quotas may have been set. Results are presented by species, with results for each jurisdiction presented within the species or group.

Draughtboard Shark

Estimated total bycatch amounts of Draughtboard Shark were high in TAS and relatively low in VIC (Figures 16 and 17). Estimated total bycatch of Draughtboard Sharks in SA was low, with bycatch only being recorded for two seasons (2009 and 2010) in the SZ, and insufficient data available to allow prediction of total bycatch of Draughtboard Shark in SA using the model-based approach.

Total estimated bycatch of Draughtboard Shark in TAS varied between approximately 150 and 250 tonnes annually (Figure 16), with a mean biomass of 191.6 ± 75.3 tonnes over the time period examined. While this is a considerable amount of bycatch given the annual catch of the target species in Tasmania, the fishing mortality from the SRLF for this species is expected to be minimal as this species is not a preferred byproduct species for bait or consumption, is not susceptible to barotrauma and is known to be particularly resilient to PCM. A small amount of Draughtboard Shark has been historically reported as byproduct from the SRLF in TAS (0.5 to 6 tonnes in the years 2007 to 2012; Hartmann, Gardner & Hobday 2013), however this is a very small proportion of the projected total bycatch estimated annually. We found that escape gaps had a minimal effect on bycatch of Draughtboard Shark, with a reduction of only 13.6% in pots with escape gaps compared to those without in TAS.

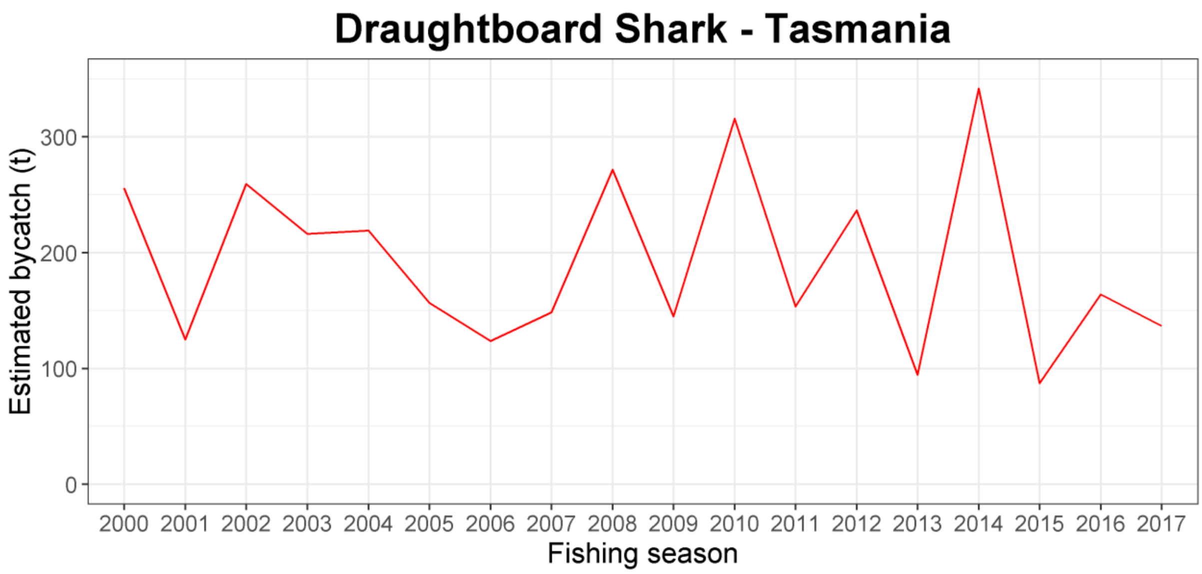

Figure 16. Predicted estimates of Draughtboard Shark from the commercial SRLF in TAS between 2000 and 2017. Predictions were made based on model parameter estimates from data collected from observer and research programs over the same period.

Predicted estimates of Draughtboard Shark bycatch in VIC are typically almost an order of magnitude lower than TAS (Figure 17), with a mean biomass of 22.0 ± 13.2 tonnes over the time period examined. Historical reporting of byproduct indicates that this species is also not preferentially kept for bait or consumption in the VIC RLF, with an annual mean of 0.011 ± 0.008 tonnes in the WZ and 0.382 ± 0.457 in the EZ between 2005 and 2017.

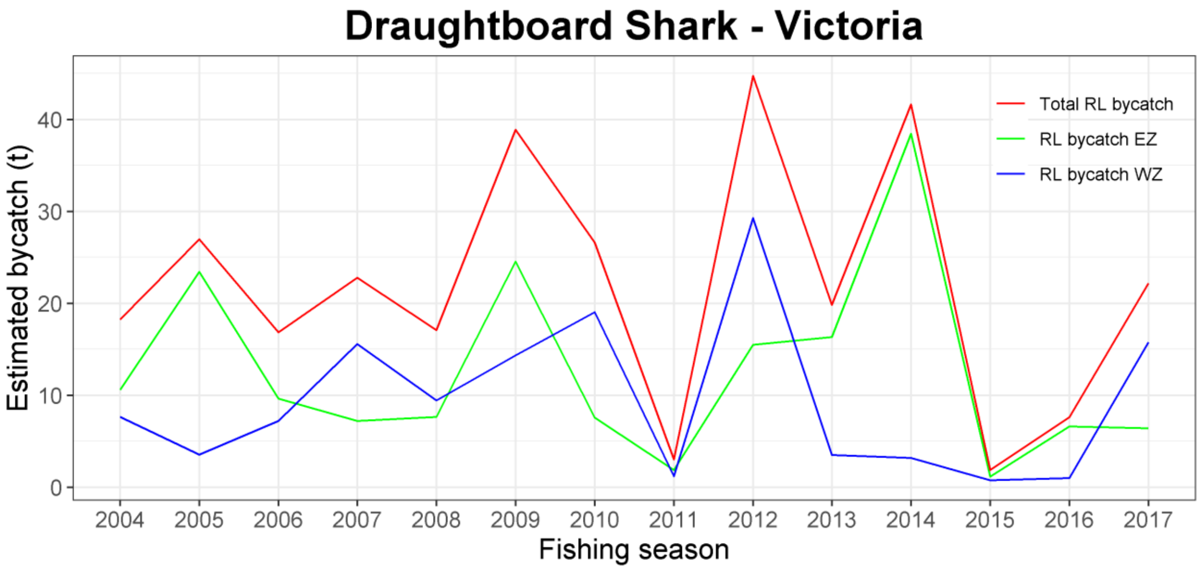

Figure 17. Predicted total bycatch of Draughtboard Shark from the commercial SRLF in VIC between 2004 and 2017. Predictions were made based on model parameter estimates from data collected from the observer program operating aboard commercial vessels over the same period. Predictions were made for each management area (EZ = Eastern Zone; WZ = Western Zone), with the total estimated bycatch displayed being the sum of each zone for the given season.

Leatherjackets

Scaled-up estimates of total leatherjacket biomass caught as bycatch in each jurisdiction indicate that the total amounts of bycatch may be comparable to that reported in scalefish fisheries in TAS, VIC (Figures 19 and 20 respectively) and SA (Figure 18) (Steer et al. 2018). For SA the mean seasonal biomass (excluding missing data) of leatherjacket over the time period examined was 35.2 ± 11.7 tonnes for the NZ, 19.2 ± 8.4 tonnes for the SZ, and 54.3 ± 12.6 tonnes over the entire jurisdiction. For TAS, the mean biomass over the time period examined was 5.0 ± 4.1 tonnes. For VIC, the mean biomass over the time period examined was 4.4 ± 3.0 tonnes for the EZ, 4.6 ± 4.6 tonnes for the WZ, and 9.0 ± 7.0 tonnes over the entire jurisdiction.

Analysis of TAS observer data (predominantly open escape gaps) and research cruise data (predominantly closed escape gaps) highlighted the importance of escape gaps as a bycatch mitigation measure for leatherjacket species. Model parameter estimates for the effect of escape gaps across the entire time-series of TAS data indicated that average bycatch of leatherjackets was reduced by 93.3% in pots with escape gaps opened.

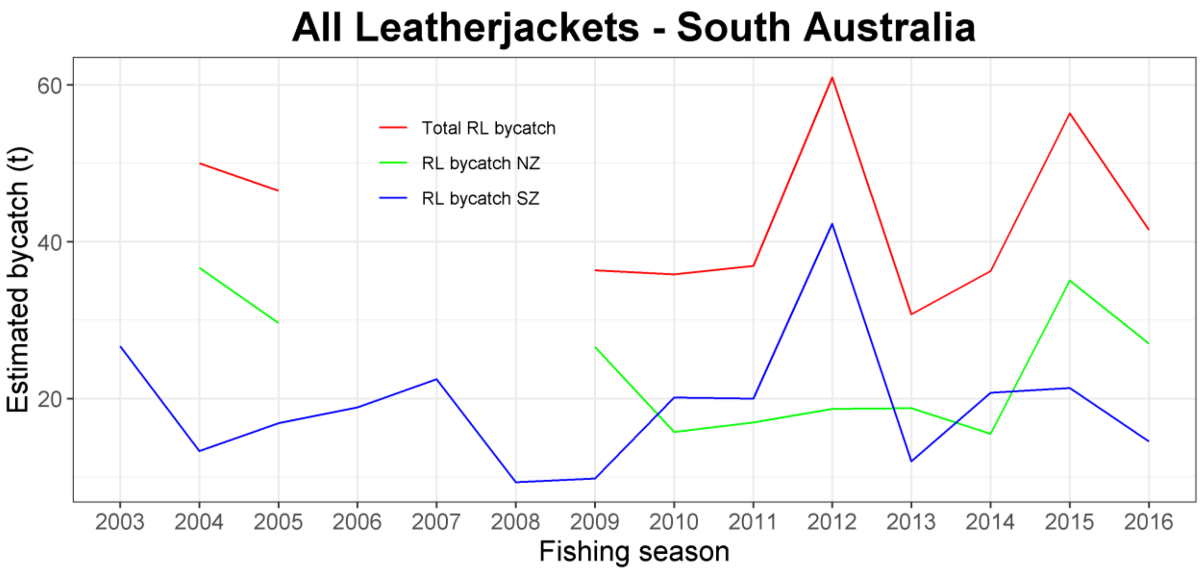

Figure 18. Predicted total bycatch of leatherjackets (all species combined but excluding Ocean Jackets) from the commercial SRLF in SA between 2003 and 2016. Predictions were made based on model parameter estimates from data collected from the observer program operating aboard commercial vessels over the same period. Predictions were made for each management area (SZ = Southern Zone; NZ = Northern Zone), with the total estimated bycatch displayed being the sum of each zone for the given season. An adjustment was made to the total estimated bycatch for the NZ as closed escape gaps were used by the observers whereas the commercial fishery uses escape gaps. This factor was estimated from work conducted by Linnane *et al.* (2011) for leatherjacket species where both open and closed escape data existed prior and post introduction of mandatory escape gaps in the NZ. Data confidentiality requirements in SA preclude presentation of data from some seasons when collected from fewer than five fishers.

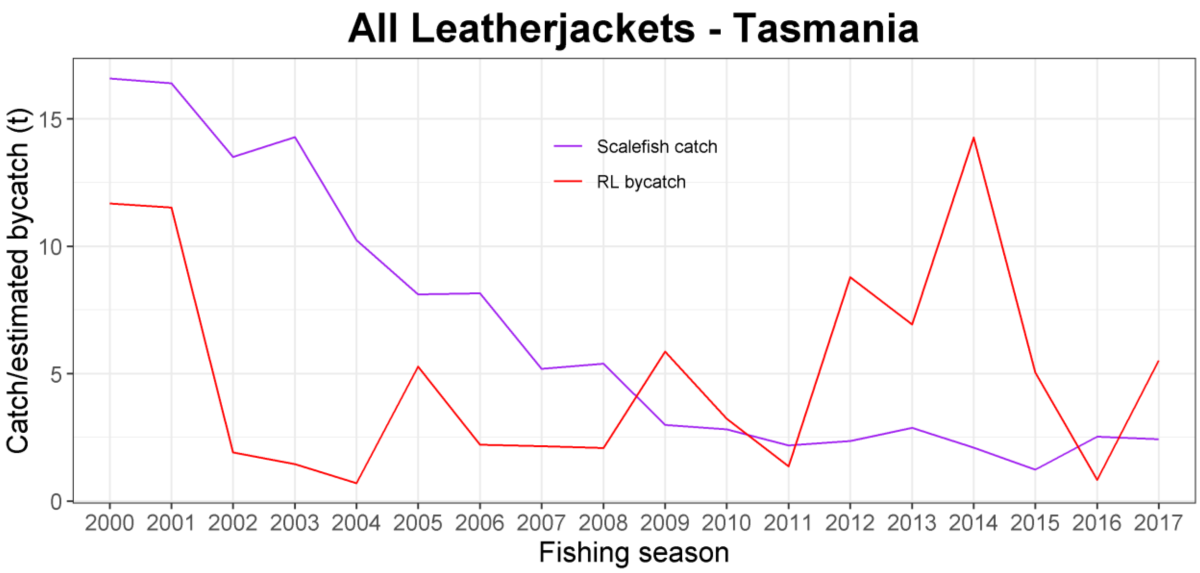

Figure 19. Predicted total bycatch of leatherjackets (all species combined) from the commercial SRLF in TAS between 2000 and 2017. Predictions were made based on model parameter estimates from data collected from observer and research programs over the same period. The total leatherjacket catch reported from the commercial scalefish fishery over the same period is shown for comparative purposes.

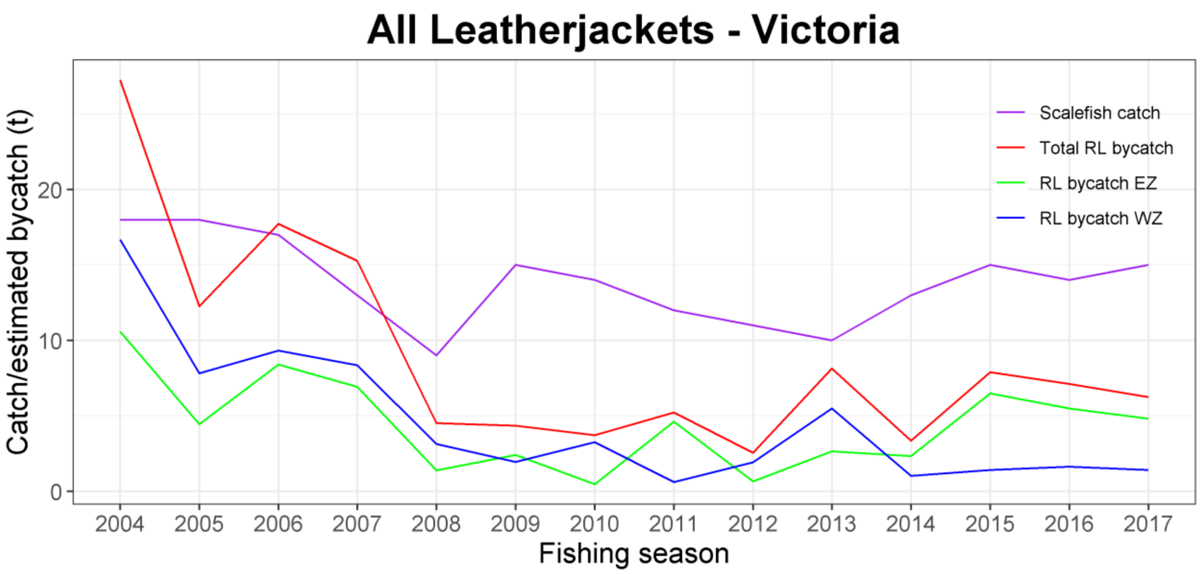

Figure 20. Predicted total bycatch of leatherjackets (all species combined) from the commercial SRLF in VIC between 2004 and 2017. Predictions were made based on model parameter estimates using data collected from the observer program aboard commercial vessels over the same period. Predictions were made for each management area (EZ = Eastern Zone; WZ = Western Zone), with the total estimated bycatch displayed being the sum of each zone for the given season. The total leatherjacket catch reported from the commercial scalefish fishery over the same period is shown for comparative purposes (DEPI 2008; DEPI 2014; DEPI 2015).

Of the six short-listed leatherjacket species identified in the PSA risk analysis as priorities for further assessment, only 2 species from two jurisdictions had sufficient data to estimate total bycatch: Degens Leatherjacket in TAS (Figure 21) and Horseshoe Leatherjacket in SA (Figure 22). For Degens Leatherjacket, the mean total biomass over the time period examined was 0.3 ± 0.5 tonnes in TAS. In SA, the mean biomass (excluding missing data) of Horseshoe Leatherjacket over the time period examined was 33.4 ± 10.0 tonnes in the NZ, 1.3 ± 0.9 tonnes in the SZ, and 25.1 ± 17.4 tonnes over the entire jurisdiction of SA. While these data provide a useful baseline, the proportion of unidentified leatherjackets in each of these jurisdictions that are either Degen's Leatherjacket or Horseshoe Leatherjacket remains unknown, and therefore the total bycatch estimates presented for each species are likely underestimated.

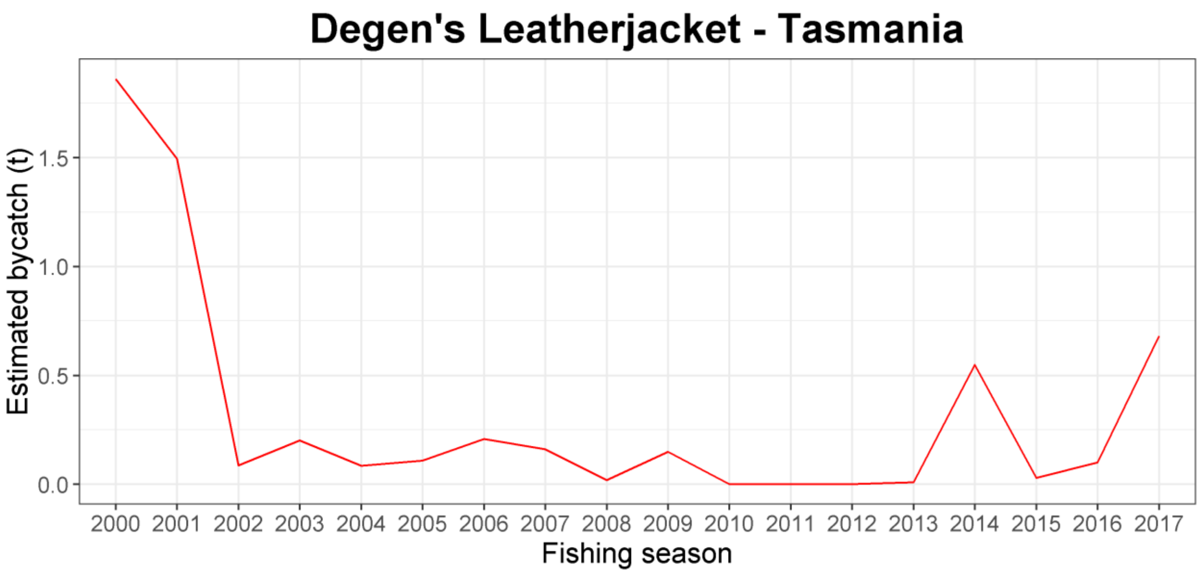

Figure 21. Predicted total bycatch of Degen's Leatherjacket from the commercial SRLF in TAS between 2000 and 2017. Predictions were made based on model parameter estimates from data collected from observer and research programs over the same period.

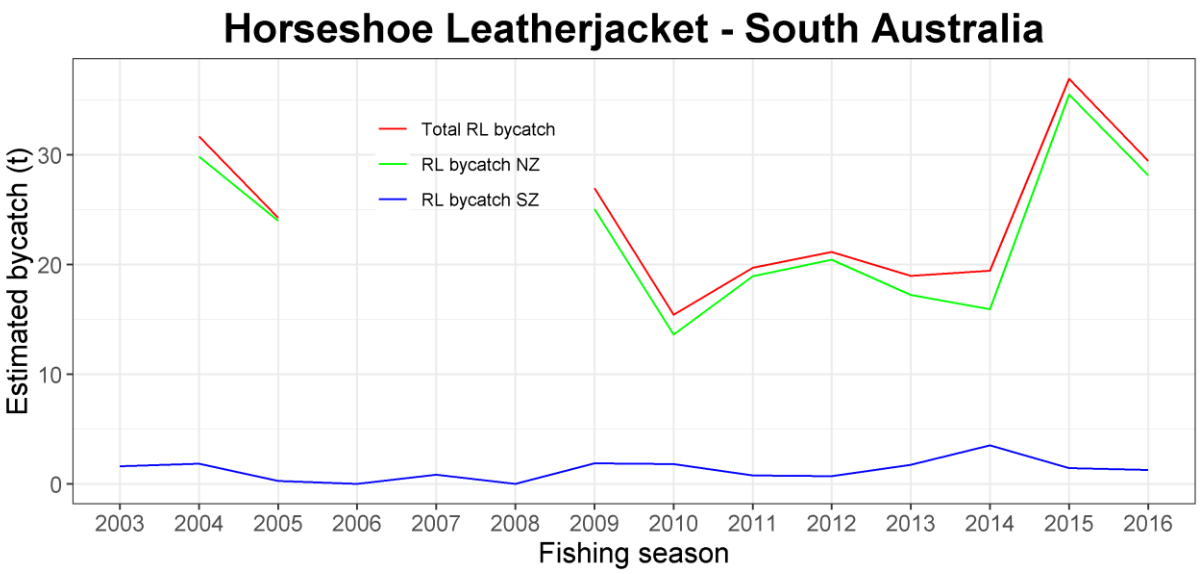

Figure 22. Predicted total bycatch of Horseshoe Leatherjacket from the commercial SRLF in SA between 2003 and 2016. Predictions were made based on model parameter estimates from data collected from the observer program operating aboard commercial vessels over the same period. Predictions were made for each management area (SZ = Southern Zone; NZ = Northern Zone), with the total estimated bycatch displayed being the sum of each zone for the given season. An adjustment was made to the total estimated bycatch for the NZ as closed escape gaps were used by the observers whereas the commercial fishery uses escape gaps. This factor was estimated from work conducted by Linnane *et al.* (2011) for Leatherjacket species where both open and closed escape data existed prior and post introduction of mandatory escape gaps in the NZ. Data confidentiality requirements in SA preclude presentation of data from some seasons when collected from fewer than five fishers.

Ocean Perch

Bycatch data for Ocean Perch were only sufficient for modelling purposes for TAS (Figure 23). This species may also be a component of the bycatch in other states where there were also “unidentified perch”. Estimated total bycatch of this species is potentially significant, typically exceeding the amount reported in the scalefish fishery, for which there only seems to be a small targeted fishery. However, estimates are highly variable dropping to almost zero in some years and approximately 30 tonnes in others. Over the time period examined the mean biomass was 10.4 ± 10.7 tonnes.

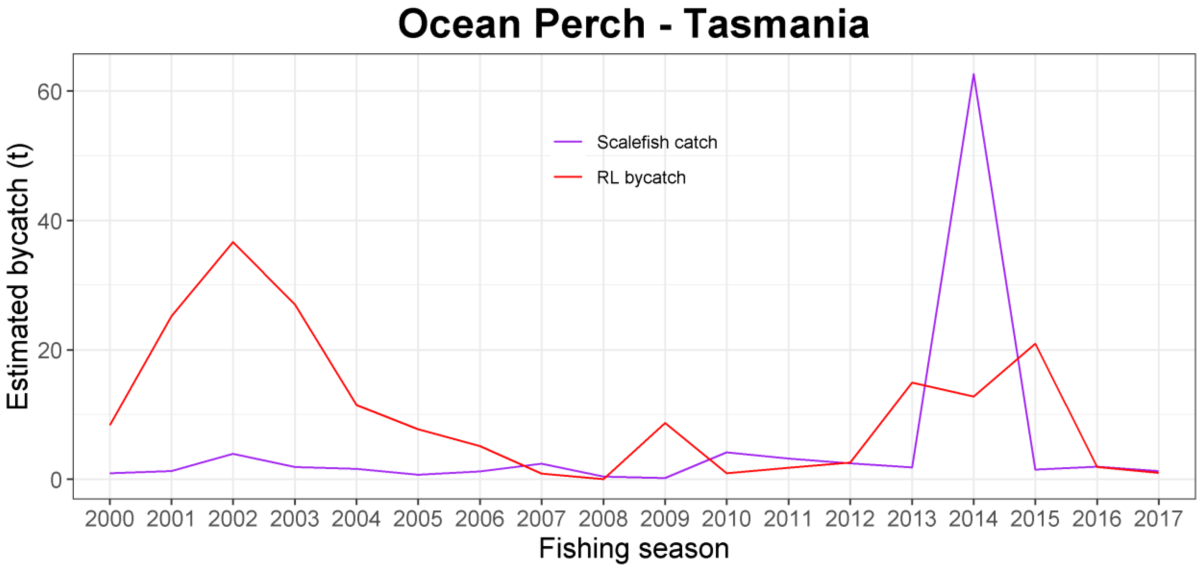

Figure 23. Predicted total bycatch of Ocean Perch from the commercial SRLF in TAS between 2000 and 2017. Predictions were made based on model parameter estimates from data collected from observer and research programs over the same period. Reported catch from the commercial scalefish fishery for this species over the same period is given for comparative purposes.

Blue-throat Wrasse

Blue-throat Wrasse were found to be a significant bycatch species across all jurisdictions, with total estimated bycatch in SA and TAS exceeding the reported catch from the scalefish fishery in some seasons (Figures 24 and 25, respectively), and being a smaller proportion of the scalefish catch in VIC (Figure 26). In SA, the mean biomass (excluding missing data) of Blue-throat Wrasse over the time period examined was 10.8 ± 5.3 tonnes for the NZ, 11.3 ± 3.8 tonnes for the SZ, and 19.0 ± 4.4 tonnes over the entire jurisdiction. For TAS, the mean biomass over the time period examined was 3.6 ± 3.5 tonnes. In Victoria, the mean biomass over the time period examined was 1.0 ± 0.9 tonnes for the EZ, 1.7 ± 1.4 tonnes for the WZ, and 2.6 ± 2.3 tonnes over the entire jurisdiction.

Analysis of TAS observer data (predominantly open escape gaps) and research cruise data (predominantly closed escape gaps) also highlighted the importance of escape gaps as a bycatch mitigation measure for Blue-throat Wrasse. Model parameter estimates for the effect of escape gaps across the entire time-series of TAS data indicated that average bycatch of Blue-throat Wrasse was reduced by 96.7% in pots with escape gaps open. This was a larger effect than reported in the NZ in SA in the year following the introduction of mandatory escape gaps (approximately 50%; Linnane *et al.* 2011).

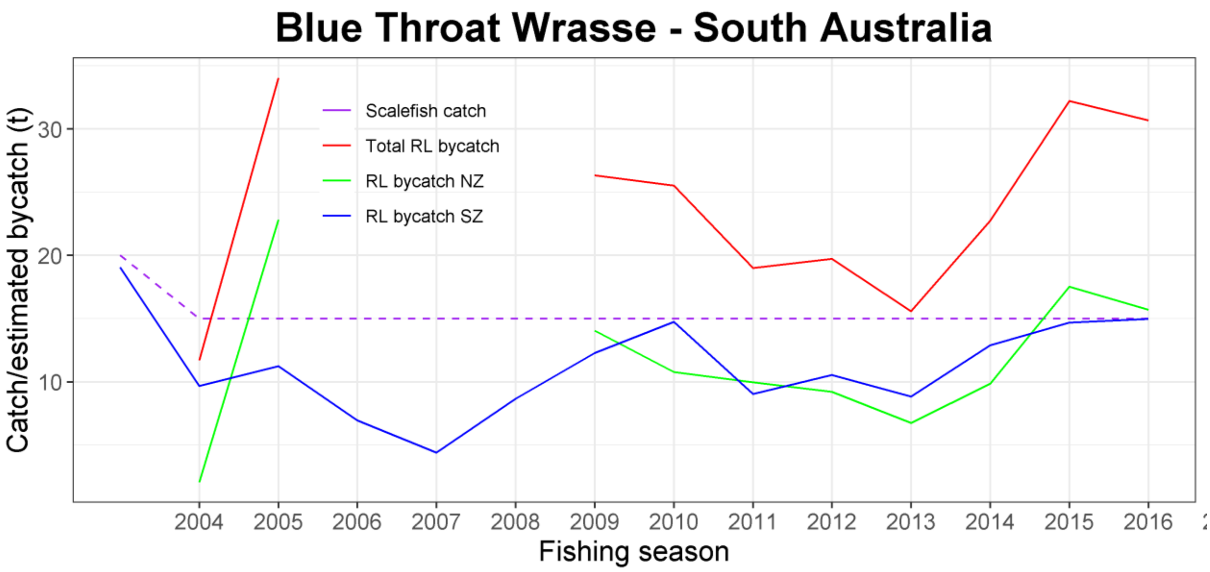

Figure 24. Predicted total bycatch of Blue-throat Wrasse from the commercial SRLF in SA between 2003 and 2016. Predictions were made based on model parameter estimates from data collected from the observer program operating aboard commercial vessels over the same period. Predictions were made for each management area (SZ = Southern Zone; NZ = Northern Zone), with the total estimated bycatch displayed being the sum of each zone for the given season. An adjustment was made to the total estimated bycatch for the NZ as closed escape gaps were used by the observers whereas the commercial fishery uses escape gaps. This factor was estimated from TAS models for Blue-throat Wrasse where both open and closed escape data existed. Data confidentiality requirements in SA preclude presentation of data from some seasons when collected from fewer than five fishers. The purple dashed line shows the approximate level of Blue-throat Wrasse catch from the commercial scalefish fishery over the same period, which represents an average estimate of catch from the stock assessment report (Steer et al. 2018).

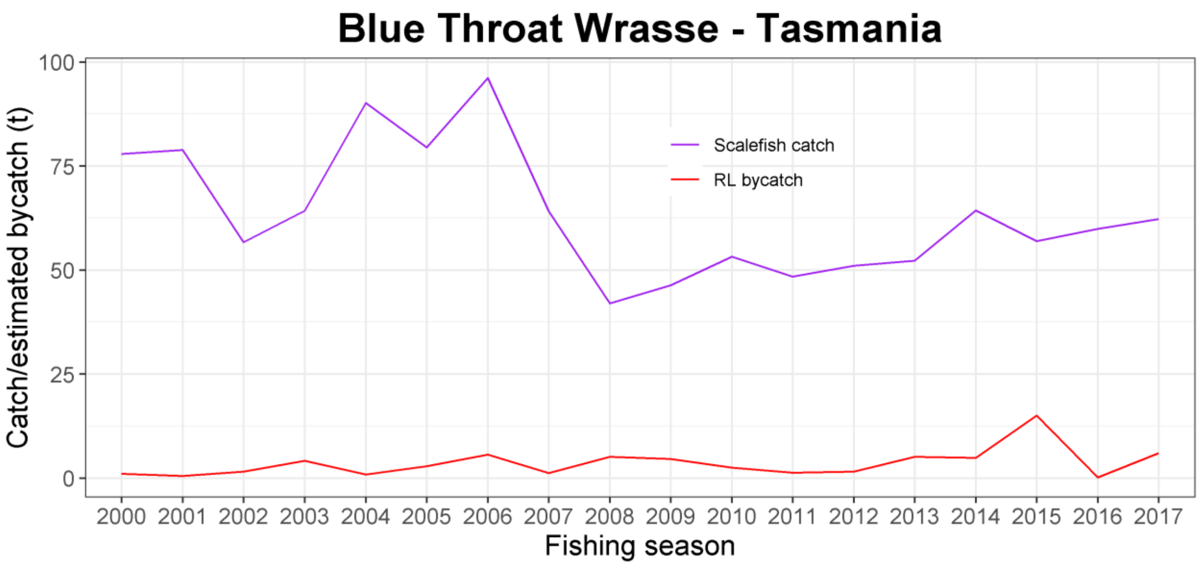

Figure 25. Predicted total bycatch of Blue-throat Wrasse from the commercial SRLF in TAS between 2000 and 2017. Predictions were made based on model parameter estimates from data collected from observer and research programs over the same period. The reported commercial scalefish catch over the same period is shown for comparative purposes.

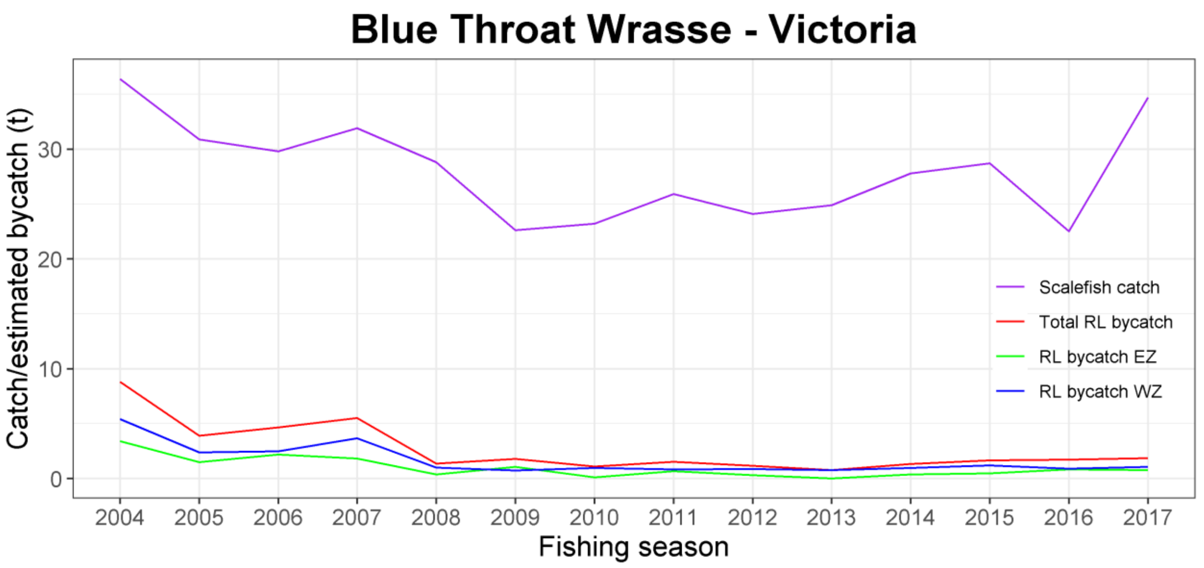

Figure 26. Predicted total bycatch of Blue-throat Wrasse from the commercial SRLF in VIC between 2004 and 2017. Predictions were made based on model parameter estimates from data collected from observer and research programs over the same period. Predictions were made for each management area (EZ = Eastern Zone; WZ = Western Zone), with the total estimated bycatch displayed being the sum of each zone for the given season. The reported commercial scalefish catch estimated for Blue-throat Wrasse over the same period is shown for comparative purposes (DEPI 2008; DEPI 2014; DEPI 2015).

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Conger Eel

Total estimates of Conger Eel bycatch were relatively lower in SA (Figure 27) and VIC (Figure 29), compared to TAS (Figure 28). For SA the mean biomass (excluding missing data) of Conger Eel over the time period examined was 0.4 ± 0.4 tonnes for the NZ, 2.9 ± 2.5 tonnes for the SZ, and 3.1 ± 2.3 tonnes over the entire jurisdiction. For TAS, the mean biomass over the time period examined was 12.1 ± 6.9 tonnes. For VIC, the mean biomass over the time period examined was 0.03 ± 0.1 tonnes for the EZ, 1.6 ± 1.3 tonnes for the WZ, and 1.6 ± 1.3 tonnes over the entire jurisdiction.

A moderate amount of Conger Eel has been historically reported as byproduct for both bait and consumption in TAS (on average approximately 2.5 tonnes annually between 2007 and 2012; Hartmann, Gardner & Hobday 2013).

Estimates of the effect of escape gaps on the bycatch of Conger Eel in TAS suggest that bycatch is reduced by approximately 90.4% when escape gaps are used. We used this figure in reducing the total bycatch estimates made for the NZ in SA, but note that further exploration of the effect of escape gaps on amounts of bycatch for this and other species is warranted.

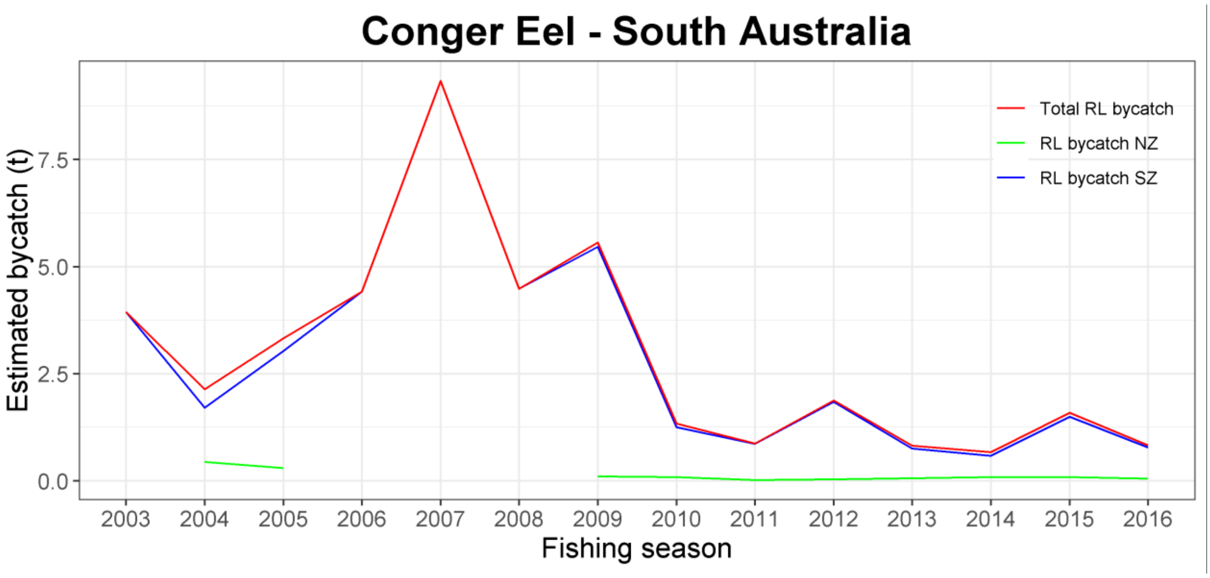

Figure 27. Predicted total bycatch of conger eel from the commercial SRLF in SA between 2003 and 2016. Predictions were made based on model parameter estimates from data collected from the observer program operating aboard commercial vessels over the same period. Predictions were made for each management area (SZ = Southern Zone; NZ = Northern Zone), with the total estimated bycatch displayed being the sum of each zone for the given season. An adjustment was made to the total estimated bycatch for the NZ as closed escape gaps were used by the observers whereas the commercial fishery uses escape gaps. This factor was estimated from TAS models for Conger Eel where both open and closed escape data existed. Data confidentiality requirements in SA preclude presentation of data from some seasons when collected from fewer than five fishers.

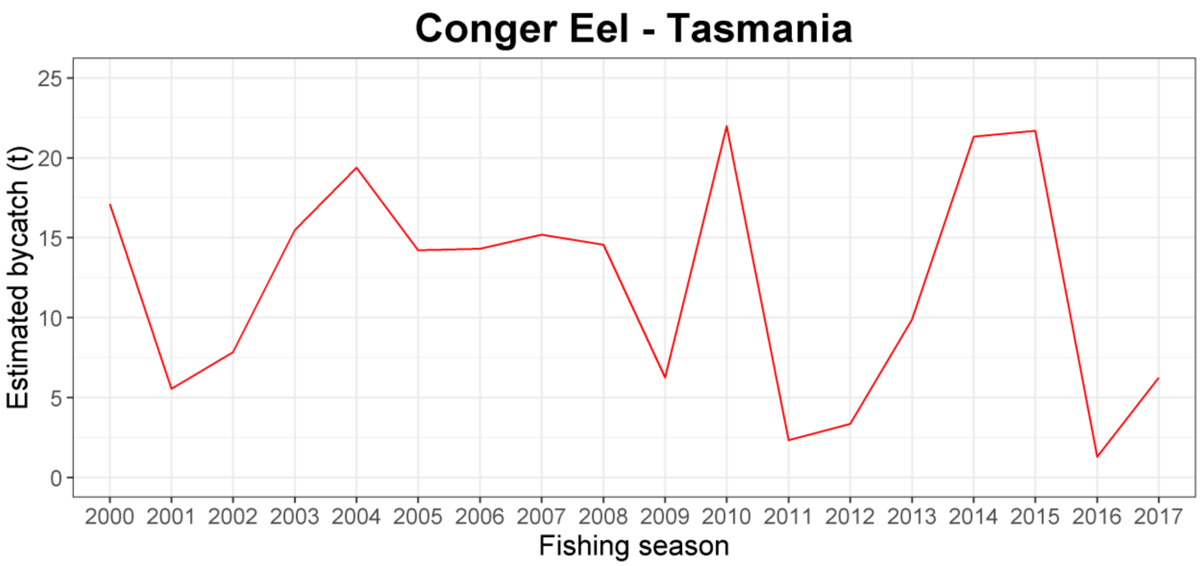

Figure 28. Predicted total bycatch of Conger Eel from the commercial SRLF in TAS between 2000 and 2017. Predictions were made based on model parameter estimates from data collected from observer and research programs over the same period.

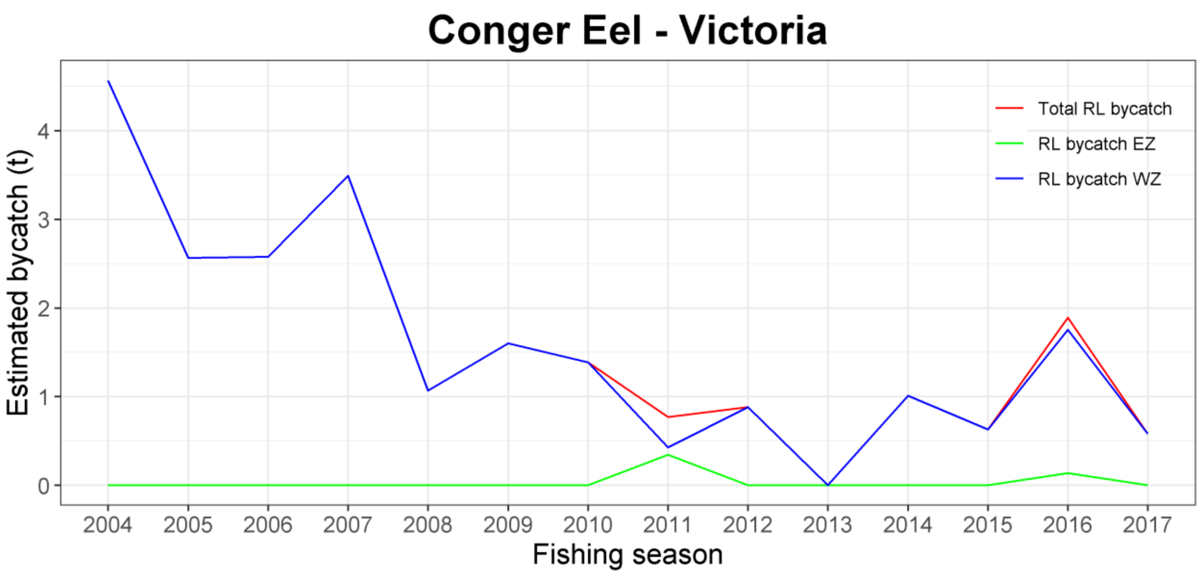

Figure 29. Predicted total bycatch of Conger Eel from the commercial SRLF in VIC between 2004 and 2017. Predictions were made based on model parameter estimates from data collected from observer and research programs over the same period.

3.2 Data poor stock assessments for medium risk species

Data poor stock assessments

The Catch-MSY approach was applied to Blue-throat Wrasse and leatherjackets (grouped) in both VIC and TAS. We present the results of this analysis below.

Victoria

Blue-throat Wrasse

Total catch of Blue-throat Wrasse was available for the commercial scalefish fishery from 1990-2017, and from our estimates of bycatch in the RL fishery from 2004-2017 (Figure 30). Estimation of r, K and MSY using the Catch-MSY approach based on this history of total catch are displayed in Table 7. Using the model outputs, the trend of total biomass of Blue-throat Wrasse between 2004 and 2017 was estimated using the Catch-MSY approach, along with future projections based on differing levels of bycatch (Figure 31). The increased levels of bycatch are estimated relative to the scalefish catch averaged over the last 5 years of data. The biomass these changes equate to, the associated catch assuming a 75% PCM level and the probability of reaching limit and reference points are summarised in Table 8. Estimated levels of Blue-throat Wrasse bycatch typically fall in the 0-10% range and therefore the probability of reaching $0.2B_0$ (i.e. 20% or less of the estimated virgin biomass) would typically be < 15%, and the probability of the stock being equal to or exceeding 0.4B0 (i.e. 40% of the estimated virgin biomass) would typically be > 60%.

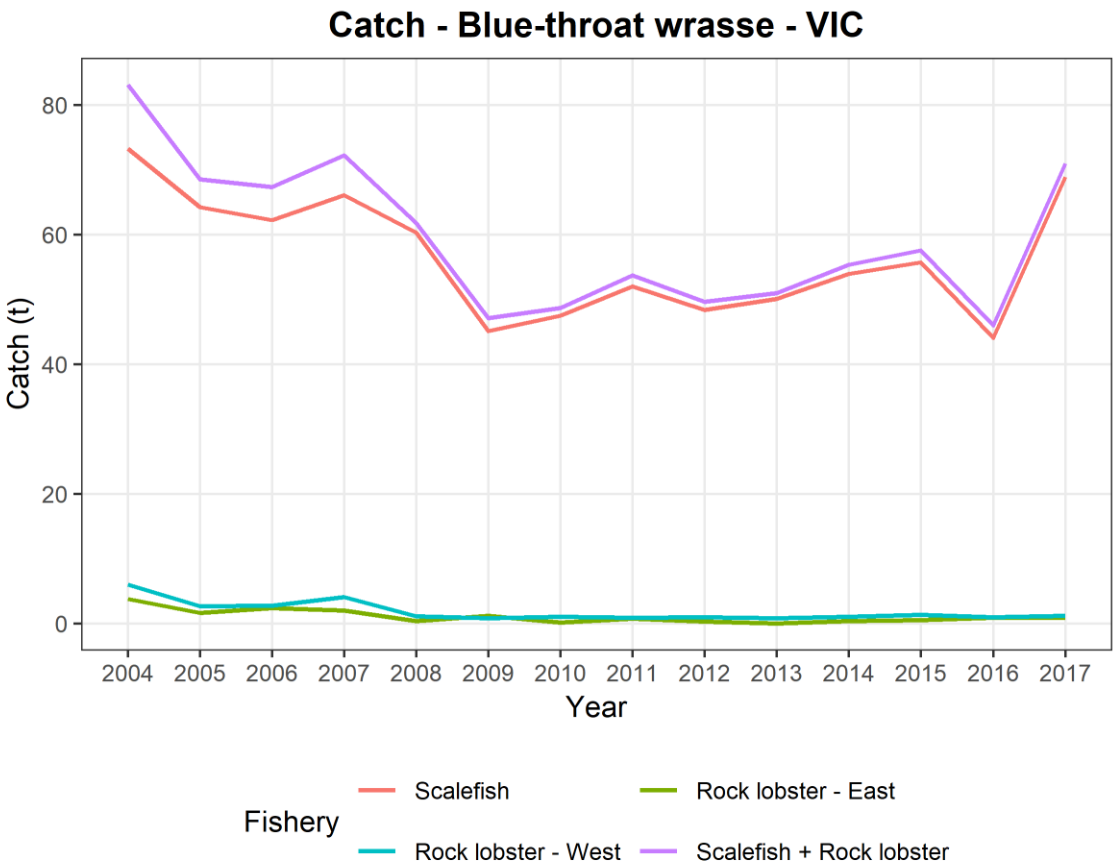

Figure 30. Time series of catch of Blue-throat Wrasse in Victoria from available fishery sources used in the Catch-MSY model.

Table 7. Summary output of key parameters from Catch-MSY analysis for Blue-throat Wrasse in VIC, showing median (50%) estimates for r, K, MSY and Current Depletion, with 95% intervals.

Parameter	2.50%	50%	97.50%
r	0.152	0.312	0.64
K	449.63	759.3	1282.2
MSY	41.101	59.204	85.281
CurrDepl	0.207	0.503	0.798

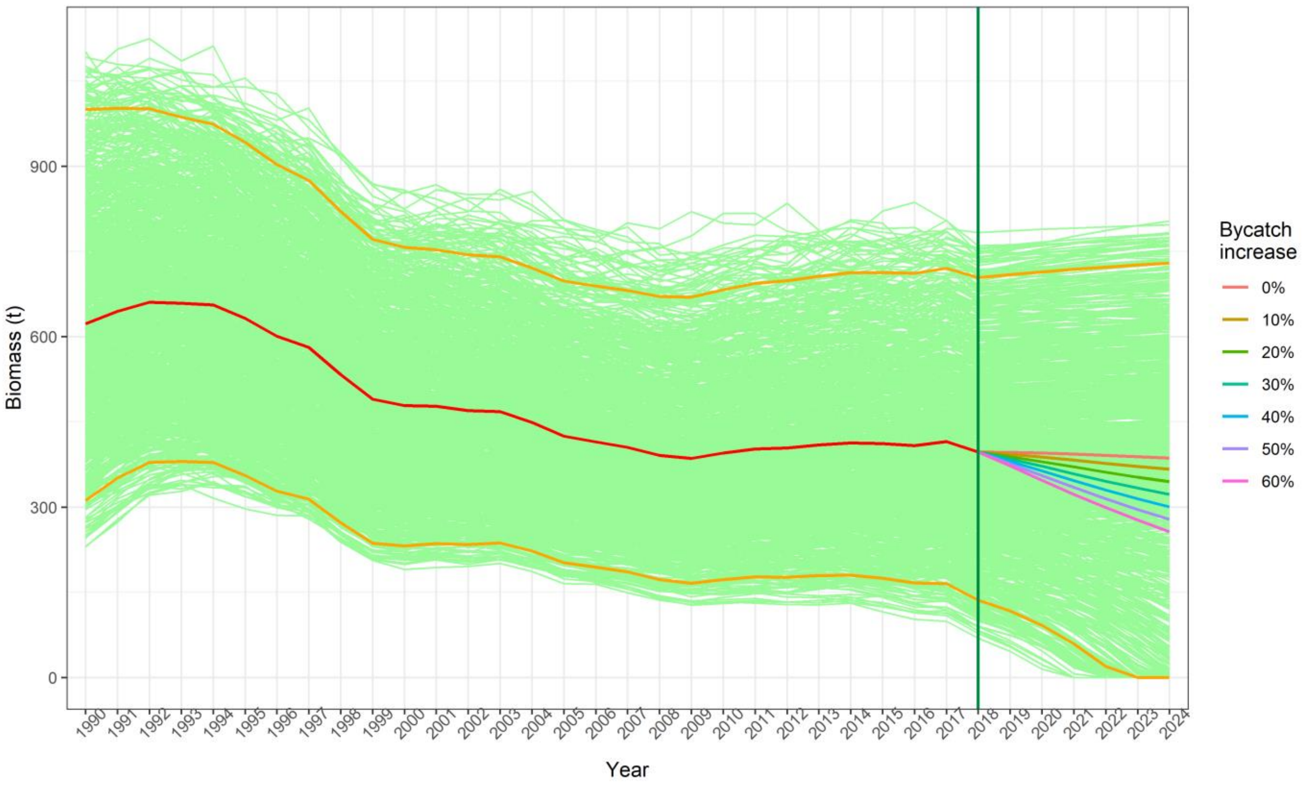

Figure 31. The estimated trend of total biomass based on catch for Blue-throat Wrasse in Victoria between 1990 and 2017. Data was available from the scalefish fishery for the entire time-series, and for estimated RL bycatch from 2004-2017. Green lines show simulated trajectories from 'accepted' values of r and K from the Catch-MSY model. The red line shows the median trajectory, with the orange lines showing the 95% confidence intervals. Biomass projections after 2017 were based assuming 75% of post-capture mortality and no change (0%) to a 60% increase of biomass of the mean of the last 5 years of the scalefish fishery.

Table 8. Projected potential increased levels of bycatch (percentage of the mean of the last 5 years of scalefish fishery catch) of Blue-throat Wrasse in VIC, associated catch level, post-capture mortality (assuming 75% level) and the estimation of the probability of reaching the standard reference points of lower than 20% and greater than or equal to 40% of the estimated virgin biomass (B_0) by 2024 as limit and target reference point respectively.

Increase (%)	Catch (t)	Post capture mortality (t)	$0.2B_0$	$0.4B_0$
0%	1.63	1.22	11.92	64.64
10%	5.45	4.09	13.15	62.43
20%	10.91	8.18	15.79	57.93
30%	16.36	12.27	18.65	53.32
40%	21.81	16.36	21.76	48.62
50%	27.27	20.45	25.11	43.85
60%	32.72	24.54	28.71	39.03

Leatherjackets

Total catch of leatherjackets was available for the commercial scalefish fishery, and from our estimates of bycatch in the RL fishery from 2004-2017 (Figure 32). Estimation of r, K and MSY using the Catch-MSY approach based on this history of total catch are displayed in Table 9. Using the model outputs, the trend of total biomass of leatherjackets between 2004 and 2017 was estimated using the Catch-MSY approach, along with the future projections based on differing levels of bycatch (Figure 33). These increased levels of bycatch are estimated relative to the scalefish catch averaged over the last 5 years of data. The biomass these changes equate to, the associated catch assuming a 75% PCM level and the probability of reaching limit and reference points are summarised in Table 10. Estimated levels of leatherjacket bycatch may sometimes exceed the 60% range of approximately 6 t total, and therefore the probability of reaching $0.2B_0$ (i.e. 20% or less of the estimated virgin biomass) would sometimes be > 50%, and the probability of the stock being equal to or exceeding 0.4B0 (i.e. 40% of the estimated virgin biomass) if bycatch was > 6t would be approximately 25%.

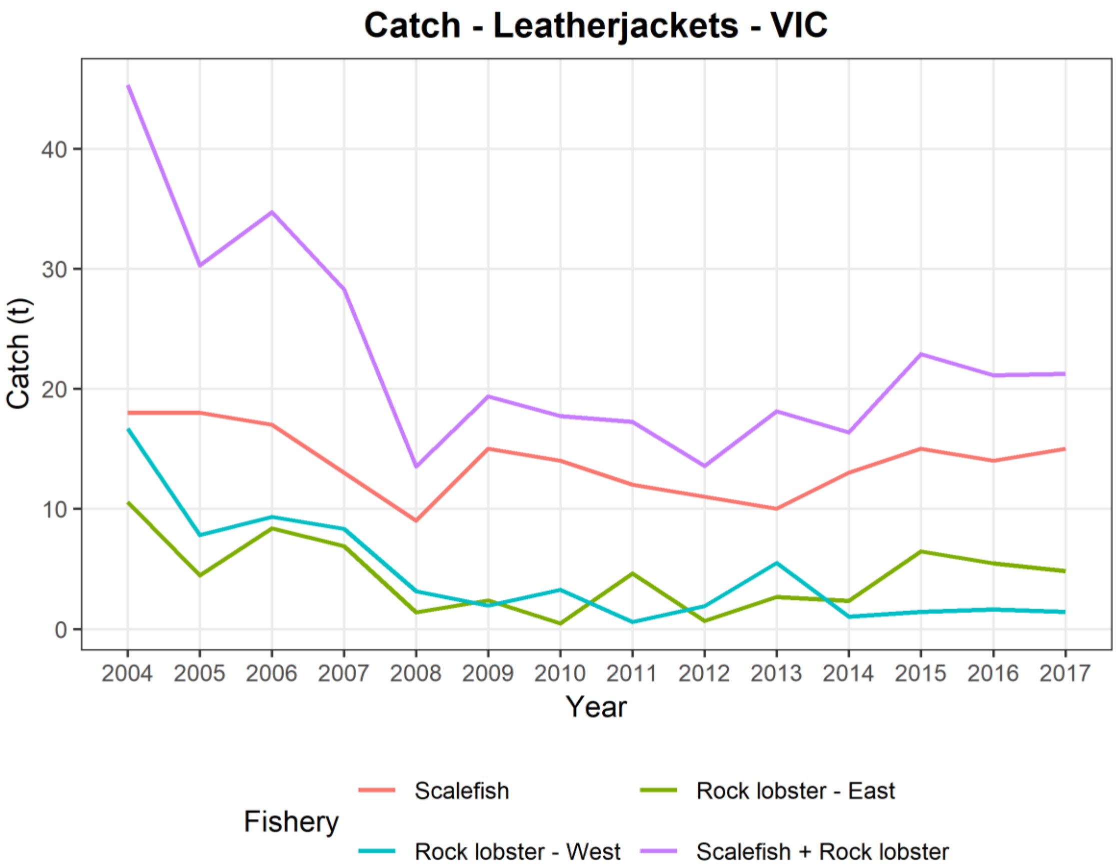

Figure 32. Time series of catch of leatherjackets in VIC from available fishery sources used in the Catch-MSY model.

Table 9. Summary output of key parameters from Catch-MSY analysis for leatherjackets in VIC, showing median (50%) estimates for r, K, MSY and Current Depletion, with 95% intervals.

Parameter	2.50%	50%	97.50%
r	0.175	0.344	0.674
K	143.83	224	348.88
MSY	12.009	19.25	30.858
CurrDepl	0.122	0.409	0.696

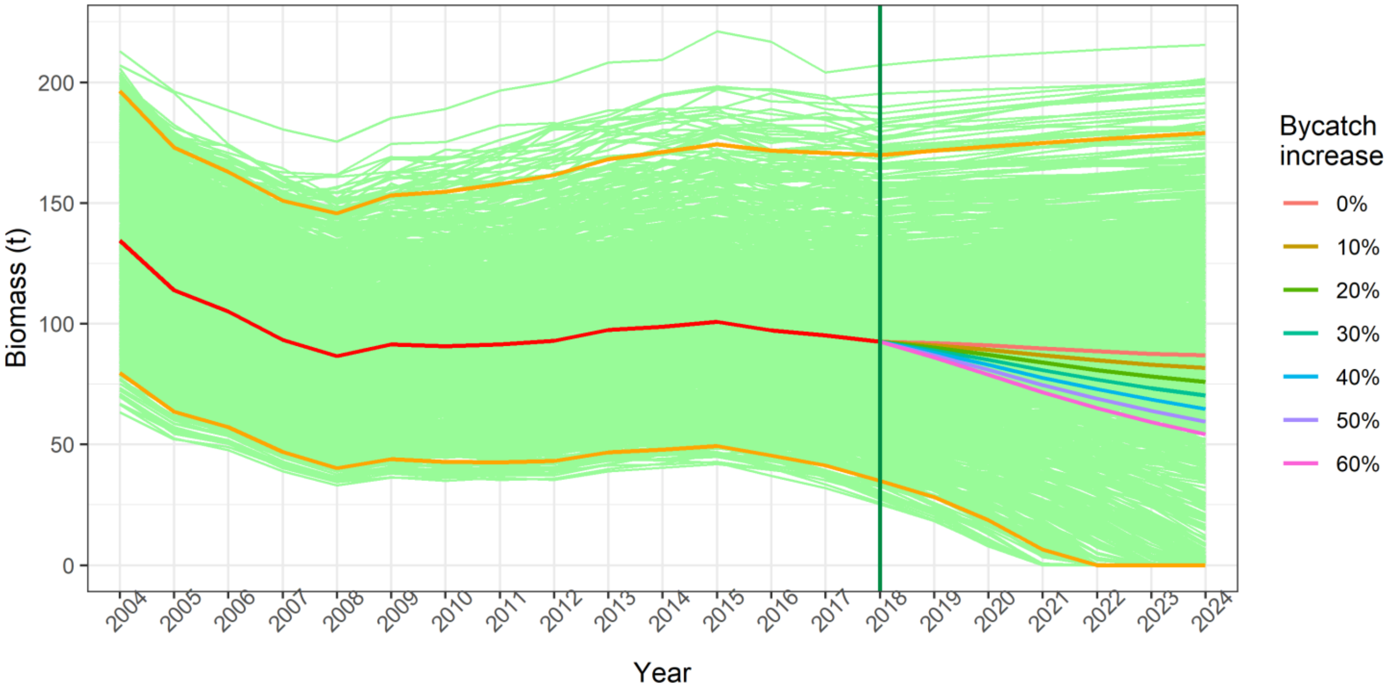

Figure 33. The estimated trend of total biomass based on catch for leatherjackets in VIC between 2004 and 2017. Green lines show simulated trajectories from ‘accepted’ values of r and K from the Catch-MSY model. The red line shows the median trajectory, with the orange lines showing the 95% confidence intervals. Biomass projections after 2017 were based assuming 75% of post-capture mortality and no change (0%) to a 60% increase of biomass of the mean of the last 5 years of the scalefish fishery.

Table 10. Projected potential increased levels of bycatch (percentage of the mean of the last 5 years of scalefish fishery catch) of leatherjackets in VIC, associated catch level, post-capture mortality (assuming 75% level) and the estimation of the probability of reaching the standard reference points of lower than 20% and greater than or equal to 40% of the estimated virgin biomass (B_0) by 2024 as limit and target reference point respectively.

Increase (%)	Catch (t)	Post capture mortality (t)	$0.2B_0$	$0.4B_0$
0%	0.20	0.15	22.18	48.06
10%	1.35	1.01	23.71	45.95
20%	2.68	2.01	26.85	41.74
30%	4.03	3.02	30.11	37.55
40%	5.36	4.02	33.51	33.40
50%	6.71	5.03	37.04	29.29
60%	8.04	6.03	40.72	25.26

Tasmania

Blue-throat Wrasse

Total catch of Blue-throat Wrasse was available for the commercial scalefish and from our estimates of bycatch in the RL fishery from 2004-2017 (Figure 34). Estimation of r, K and MSY using the Catch-MSY approach based on this history of total catch are displayed in Table 11. Using the model outputs, the trend of total biomass of Blue-throat Wrasse between 2000 and 2017 was estimated using the Catch-MSY approach, along with the future projections based on differing levels of bycatch (Figure 35). These increased levels of bycatch are estimated relative to the scalefish catch averaged over the last 5 years of data. The biomass these changes equate to, the associated catch assuming a 75% PCM level and the probability of reaching limit and reference points is summarised in Table 12. Estimated levels of RL bycatch typically fall in the 0-20% range and therefore the probability of reaching $0.2B_0$ (i.e. 20% or less of the estimated virgin biomass) would typically be in the range of approximately 18-25%, and the probability of the stock being equal to or exceeding 0.4B0 (i.e. 40% of the estimated virgin biomass) would typically be in the range of 43-52%.

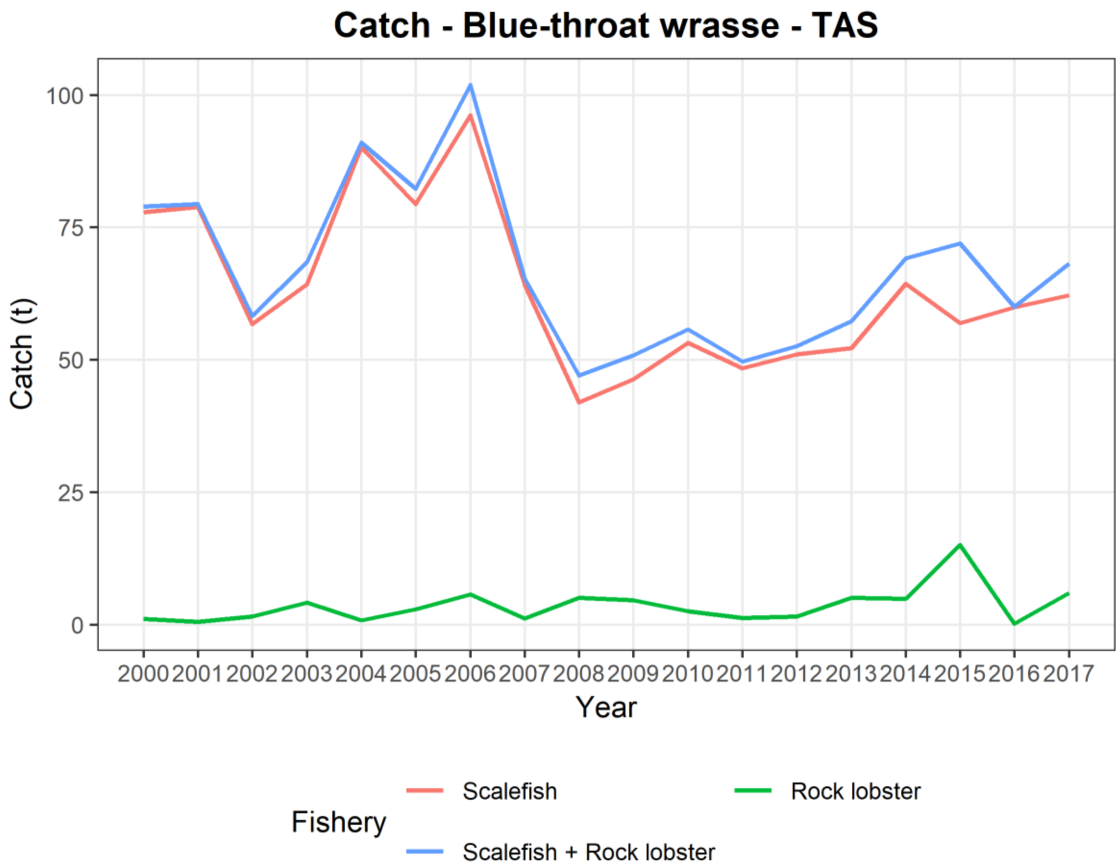

Figure 34. Time series of catch of Blue-throat Wrasse in TAS from available fishery sources used in the Catch-MSY model.

Table 11. Summary output of key parameters from Catch-MSY analysis for Blue-throat Wrasse in TAS, showing median (50%) estimates for r, K, MSY and Current Depletion, with 95% intervals.

Parameter	2.50%	50%	97.50%
r	0.171	0.329	0.635
K	533.18	874.32	1433.7
MSY	49.678	71.971	104.27
CurrDepl	0.134	0.423	0.712

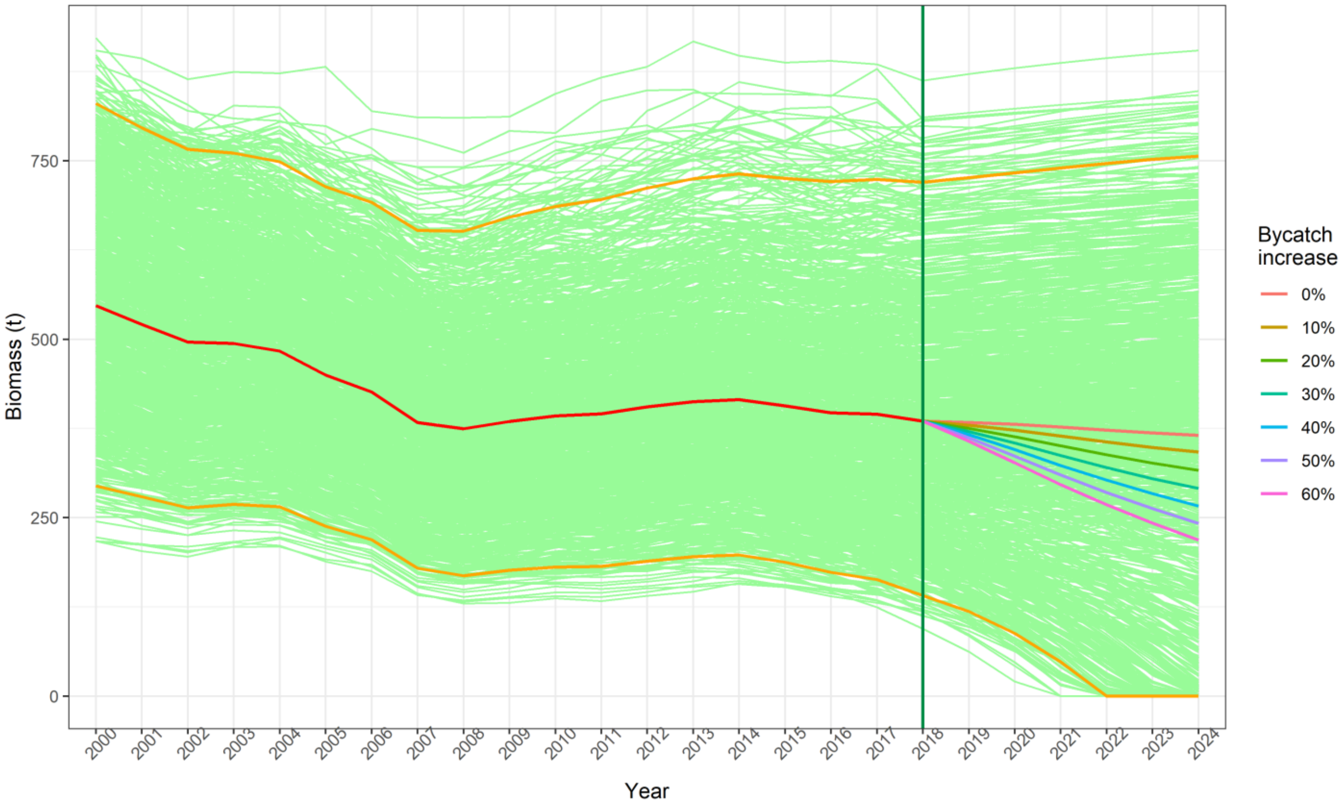

Figure 35. The estimated trend of total biomass based on catch for Blue-throat Wrasse in TAS between 2000 and 2017. Green lines show simulated trajectories from 'accepted' values of r and K from the Catch-MSY model. The red line shows the median trajectory, with the orange lines showing the 95% confidence intervals. Biomass projections after 2017 were based assuming 75% of PCM and no change (0%) to a 60% increase of biomass of the mean of the last 5 years of the scalefish fishery.

Table 12. Projected potential increased levels of bycatch (percentage of the mean of the last 5 years of scalefish fishery catch) of Blue-throat Wrasse in TAS, associated catch level, post-capture mortality (assuming 75% level) and the estimation of the probability of reaching the standard reference points of lower than 20% and greater than or equal to 40% of the estimated virgin biomass (B_0) by 2024 as limit and target reference point respectively.

Increase (%)	Catch (t)	Post capture mortality (t)	$0.2B_0$	$0.4B_0$
0%	4.67	3.50	18.64	52.88
10%	5.91	4.43	21.69	48.50
20%	11.83	8.87	25.22	43.66
30%	17.73	13.30	28.93	38.87
40%	23.64	17.73	32.83	34.12
50%	29.56	22.17	36.90	29.46
60%	35.47	26.60	41.15	24.92

Leatherjackets

Total catch of leatherjackets was available for the commercial scalefish fishery, and from our estimates of bycatch in the RL fishery from 2000-2017 (Figure 36). Estimation of r, K and MSY using the Catch-MSY approach based on this history of total catch are displayed in Table 13. Using the model outputs, the trend of total biomass of leatherjackets between 2000 and 2017 was estimated using the Catch-MSY approach, along with the future projections based on differing levels of bycatch (Figure 37). These increased levels of bycatch are changes relative to the scalefish catch averaged over the last 5 years of data. The biomass these changes equate to, the associated catch assuming a 75% PCM level and the probability of reaching limit and reference points is summarised in Table 14. Estimated levels of RL bycatch may sometimes exceed the 60% range of approximately 1 t total, and therefore the probability of reaching $0.2B_0$ (i.e. 20% or less of the estimated virgin biomass) would sometimes be > 45%, and the probability of the stock being equal to or exceeding 0.4B0 (i.e. 40% of the estimated virgin biomass) if bycatch was > 1 t would be approximately < 20%.

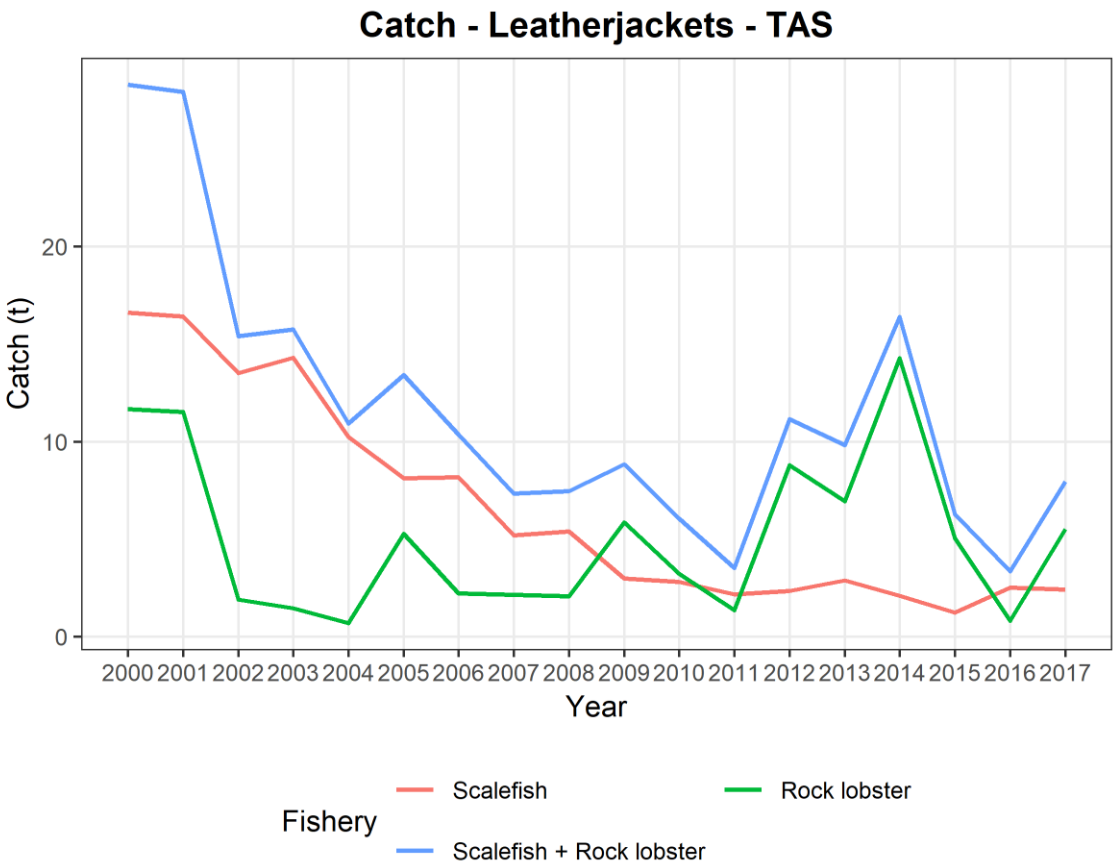

Figure 36. Time series of catch of leatherjackets in TAS from available fishery sources used in the Catch-MSY model.

Table 13. Summary output of key parameters from Catch-MSY analysis for leatherjackets in TAS, showing median (50%) estimates for r, K, MSY and Current Depletion, with 95% intervals.

Parameter	2.50%	50%	97.50%
r	0.153	0.296	0.574
K	101.07	144.2	205.72
MSY	6.545	10.685	17.442
CurrDepl	0.04	0.291	0.542

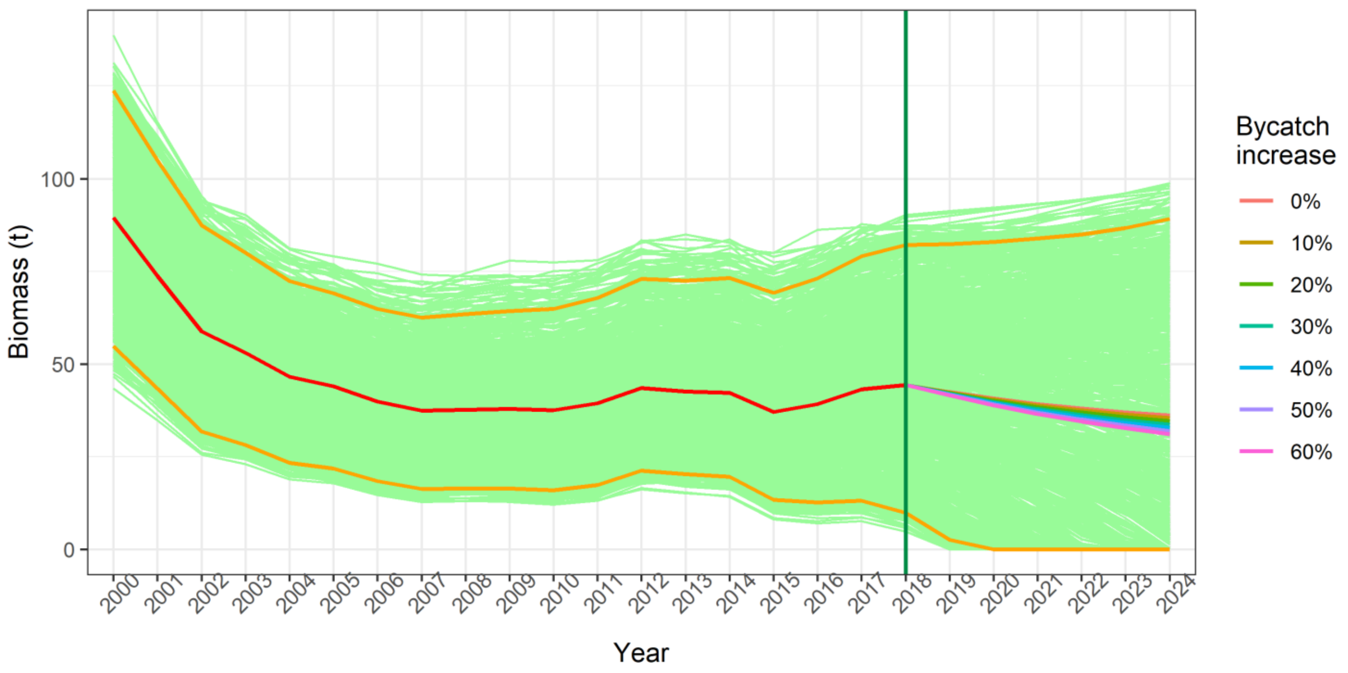

Figure 37. The estimated trend of total biomass based on catch for leatherjackets in TAS between 2000 and 2017. Green lines show simulated trajectories from ‘accepted’ values of r and K from the Catch-MSY model. The red line shows the median trajectory, with the orange lines showing the 95% confidence intervals. Biomass projections after 2017 were based assuming 75% of post-capture mortality and no change (0%) to a 60% increase of biomass of the mean of the last 5 years of the scalefish fishery.

Table 14. Projected potential increased levels of bycatch (percentage of the mean of the last 5 years of scalefish fishery catch) of leatherjackets in TAS, associated catch level, post-capture mortality (assuming 75% level) and the estimation of the probability of reaching the standard reference points of lower than 20% and greater than or equal to 40% of the estimated virgin biomass (B_0) by 2024 as limit and target reference point respectively.

Increase (%)	Catch (t)	Post capture mortality (t)	$0.2B_0$	$0.4B_0$
0%	0.17	0.13	40.24	24.49
10%	0.23	0.17	40.94	23.73
20%	0.45	0.34	41.73	22.88
30%	0.67	0.5	42.52	22.04
40%	0.89	0.67	43.32	21.21
50%	1.12	0.84	44.14	20.38
60%	1.35	1.01	44.96	19.55

Alternative approaches to determining reference points

We present below the results of two alternative approaches to setting reference points for monitoring of bycatch: inter-annual change in CPUE and detecting trends in catch. Results are presented by state for all the key species where sufficient data was available. For Degen's Leatherjacket in TAS, species level data was mainly collected on research cruises rather than observer trips, and therefore we omit presenting analysis for this species.

Inter-annual change in CPUE

South Australia

Sufficient data was available to examine inter-annual change in CPUE in SA for: leatherjackets (Figure 38), Conger Eel (Figure 39) and Blue-throat Wrasse (Figure 40). Proportion change was calculated using the maximum CPUE in the time-series.

Inter-annual change in CPUE displayed a high degree of variability for all species in both management zones in SA, but generally stayed within a 50% difference between years. The exception was leatherjackets, which breached the 50% change threshold in both management zones over the time period assessed. It should be noted that a shorter time period was assessed for the SA data due to data confidentiality issues creating gaps in the time-series.

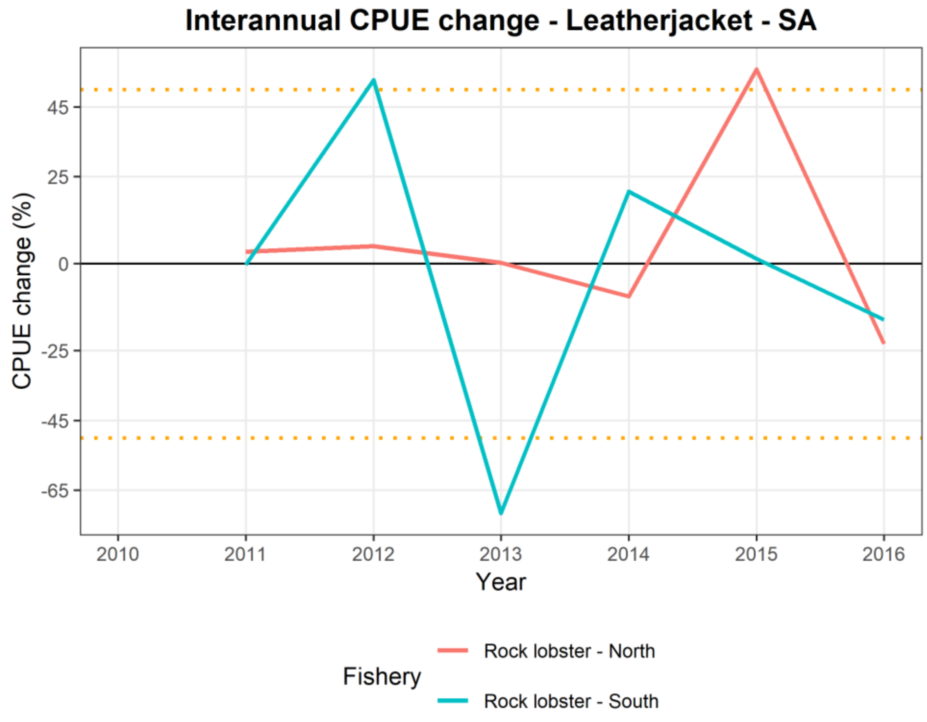

Figure 38. Inter-annual CPUE change for leatherjackets in each management zone in SA. Some years in the time-series were excluded due to data confidentiality issues. Dotted orange lines show a 50% change reference point.

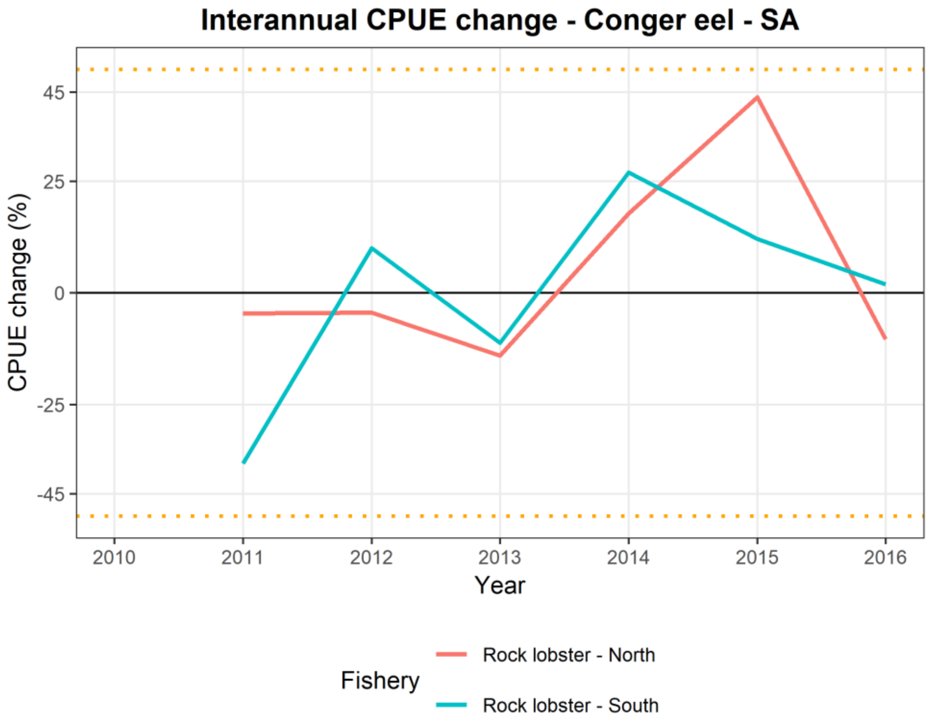

Figure 39. Inter-annual CPUE change for Conger Eel in each management zone in SA. Some years in the time-series were excluded due to data confidentiality issues. Dotted orange lines show a 50% change reference point.

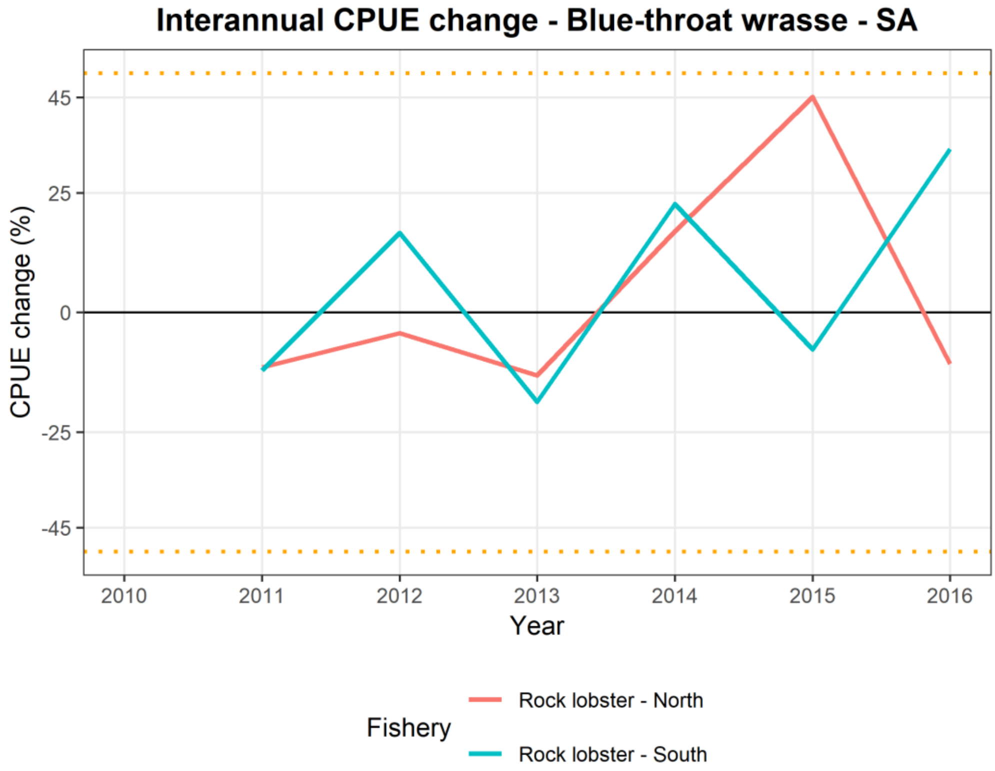

Figure 40. Inter-annual CPUE change for Blue-throat Wrasse in each management zone in SA. Some years in the time-series were excluded due to data confidentiality issues. Dotted orange lines show a 50% change reference point.

Victoria

Sufficient data was available to examine inter-annual change in CPUE in VIC for: leatherjackets (Figure 41), Conger Eel (Figure 42) and Blue-throat Wrasse (Figure 43). Proportion change was calculated using the maximum CPUE in the time-series.

Inter-annual change in CPUE was particularly variable for all species, with occasional breaches of the 50% threshold. For Blue-throat Wrasse and leatherjackets the variability decreased in more recent years, most likely due to higher catch rates early in the time series which were used as a reference. The large spikes in the time-series for Conger Eel in the EZ should be seen in light of the typically very low catch rates in the EZ and therefore small fluctuations in catch result in large inter-annual variation.

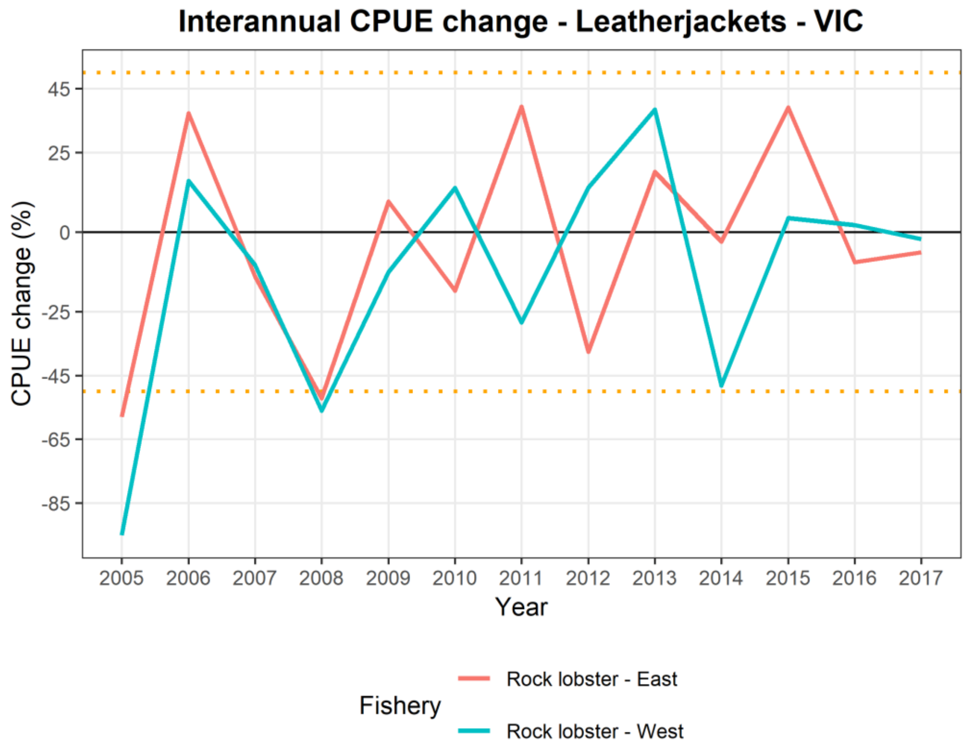

Figure 41. Inter-annual CPUE change for leatherjackets in each management zone in VIC. Dotted orange lines show a 50% change reference point.

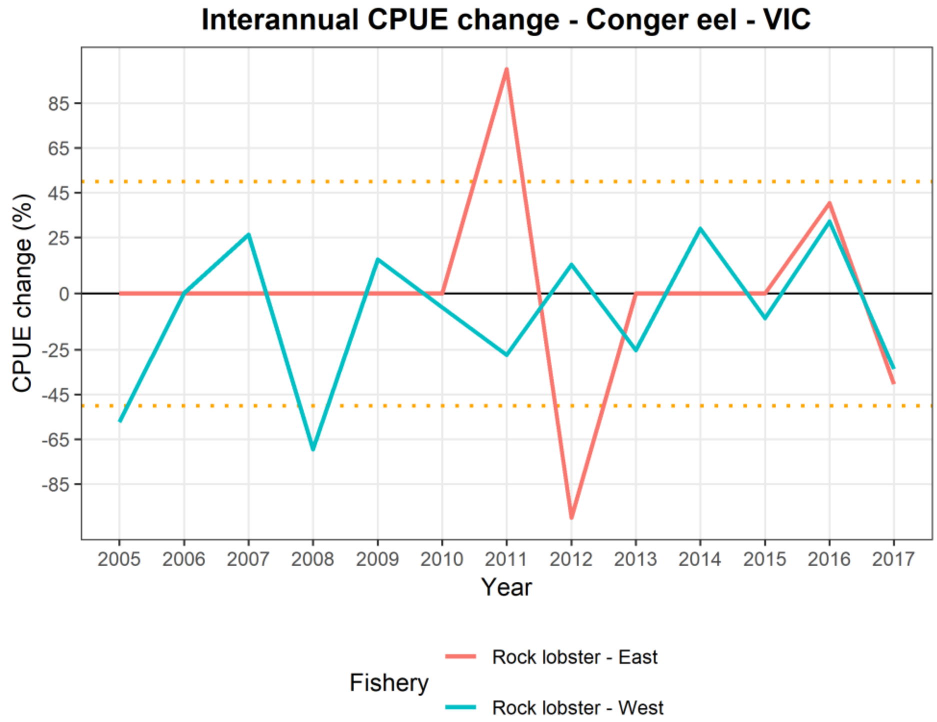

Figure 42. Inter-annual CPUE change for Conger Eel in each management zone in VIC. Dotted orange lines show a 50% change reference point.

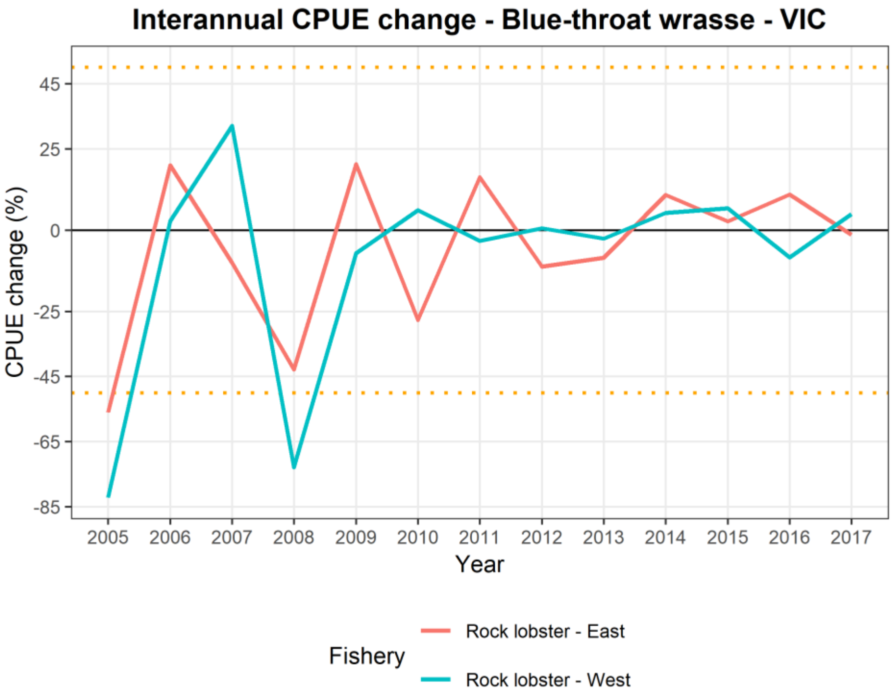

Figure 43. Inter-annual CPUE change for Blue-throat Wrasse in each management zone in VIC. Dotted orange lines show a 50% change reference point.

Tasmania

Sufficient data was available to examine inter-annual change in CPUE in TAS for: leatherjackets (Figure 44), Conger Eel (Figure 45) and Blue-throat Wrasse (Figure 46) and Ocean Perch (Figure 47).

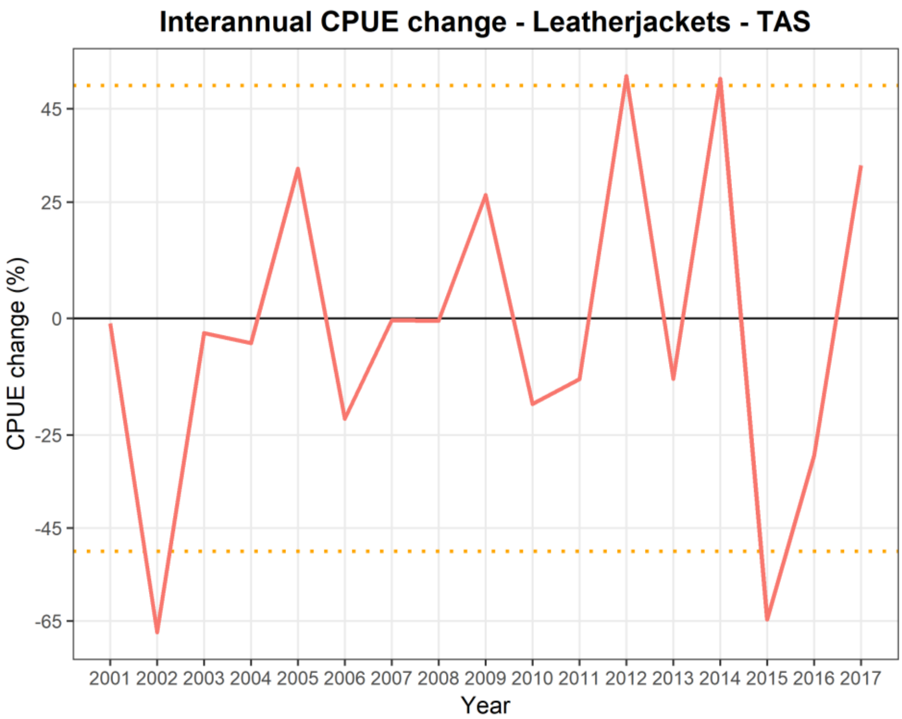

Figure 44. Inter-annual CPUE change for leatherjackets in TAS. Dotted orange lines show a 50% change reference point.

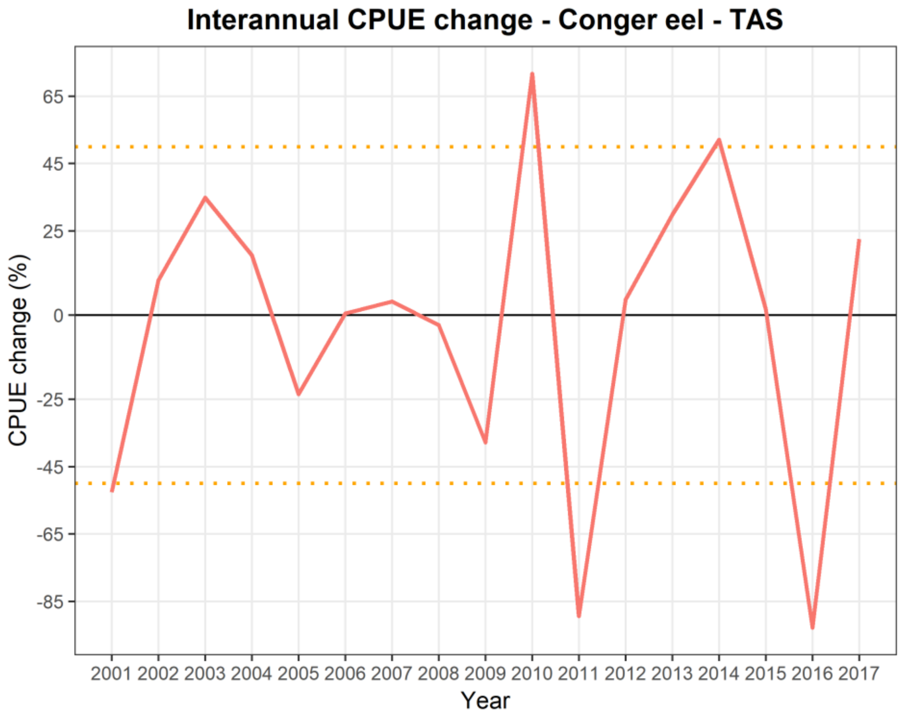

Figure 45. Inter-annual CPUE change for Conger Eel in TAS. Dotted orange lines show a 50% change reference point.

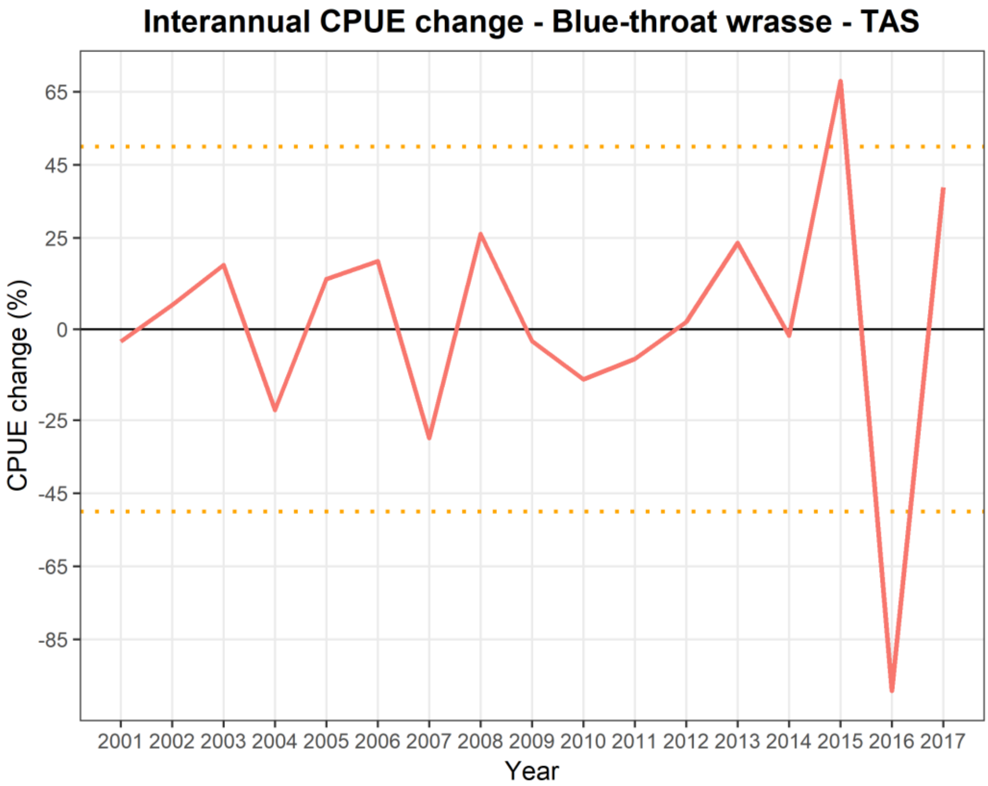

Figure 46. Inter-annual CPUE change for Blue-throat Wrasse in TAS. Dotted orange lines show a 50% change reference point.

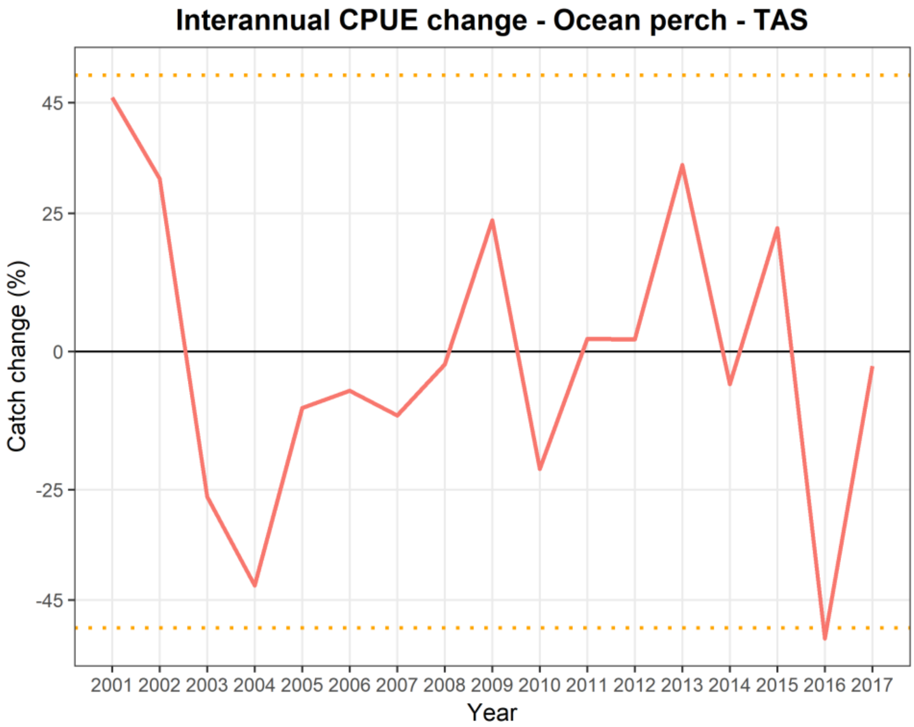

Figure 47. Inter-annual CPUE change for Ocean Perch in TAS. Dotted orange lines show a 50% change reference point.

Trends in the time-series of bycatch

Significant trends in the time-series of bycatch data were more evident when analysing the entire length of the time-series rather than just the last 5 years of data (Table 15). Significant trends here refer to a statistically significant “smooth” for the estimated time-series of total bycatch for a species. This indicates that the trend is significantly different to zero, and examination of the smooth trends showed the direction of the trend. In the last 5 years of bycatch data, no trends in total estimated bycatch of any species (or group) was found to be statistically significant. When analysing the entire time-series, a significant increasing trend was found in the total estimated bycatch for Conger Eel in the EZ of VIC ($p = 0.009$); although this increase was small and related to extremely low catch rates early in the time series (see Figure 29). Significant decreasing trends were found for Blue-throat Wrasse in both the EZ ($p = 0.035$) and WZ ($p= 0.004$) of VIC. A significant decreasing trend was also found for leatherjackets in the WZ of VIC ($p = 0.000003$). Significant decreasing trends in the total estimated bycatch were found for Conger Eel in both the NZ ($p = 0.00002$) and SZ ($p=0.0002$) of SA over the entire time-series of bycatch data.

Table 15. Results of Generalized Additive Model analysis for the last 5 years of bycatch data and the entire time-series of bycatch data for key species in each state/management zone. Statistically significant trends at the p=0.05 level are marked with an *. The direction of the trend is positive when marked with ↑, and negative when marked with ↓.

Species	**State**	**Management zone**	**GAM p-value (5 years)**	**GAM p-value (entire time-series)**
Draughtboard Shark	VIC	EZ	0.284	0.772
		WZ	0.090	0.762
	TAS		0.763	0.481
Leatherjackets	SA	NZ	0.212	0.086
		SZ	0.267	0.815
	VIC	EZ	0.260	0.051
		WZ	0.193	0.000003* ↓
	TAS		0.393	0.261
Horseshoe Leatherjacket	SA	NZ	0.240	0.233
		SZ	0.483	0.239
Conger Eel	SA	NZ	0.540	0.00002* ↓
		SZ	0.506	0.0002 * ↓
	VIC	EZ	0.213	0.009* ↑
		WZ	0.511	0.060
	TAS		0.346	0.517
Ocean Perch	TAS		0.118	0.224
Blue throat Wrasse	SA	NZ	0.086	0.730
		SZ	0.059	0.059
	VIC	EZ	0.176	0.035 * ↓
		WZ	0.285	0.004 * ↓
	TAS		0.892	0.708

Discussion

Pot fisheries such as the SRLF are generally considered to have lower ecological impacts on bycatch compared to other fishing methods such as trawl fisheries (Kelleher 2005). The results of this project are in general concordance with this view. For example, the risk assessment did not identify any bycatch, byproduct or TEP species under high risk from the impacts of the operations of the SRLF. However, some species of bycatch in the SRLF are also target or bycatch species in other fisheries or may be subject to other environmental impacts and therefore may be subjected to cumulative impacts. From this perspective, the ongoing monitoring of species identified in this report as important bycatch species in these fisheries is warranted, along with periodic reviews of changes in bycatch composition throughout time. Our research also highlights the knowledge gap in life history traits for some key species, making a full assessment of risk problematic. We have identified some areas for improvement in the way data is collected and handled across the fishery. And adopting a standardised approach to sampling protocols and data analysis that take into account the key recommendations made in this report will provide stakeholders with the assurance that bycatch monitoring and management in the SRLF is occurring at internationally best practice standards.

Ensuring data collection and management is best practice

The assessment of the current bycatch monitoring program operating in the SRLF against the United States' Tier Classification Scheme (TCS) provided a means of identifying areas in need of improvement. Overall scores placed each state in tier 2 out of a possible 5 tiers, which is an average position when considering the size of the fishery. It should be noted that scoring the highest points achievable in some sections of the TCS would be unrealistic for this fishery. For example, in order to score full points in the 'sampling design' and 'sampling coverage' sections would require a full census of pots by observers which would be logistically and financially unachievable. While gaining the highest tier classification may not be an achievable goal, a number of areas were identified that will raise the standards of bycatch monitoring in the SRLF to best practice. We discuss these in detail below.

Improvements to the observer program

Independent observer programs are a cornerstone of bycatch monitoring worldwide as they provide a fishery independent means of gaining an unbiased sample of bycatch, byproduct and TEPS interactions. The importance of a well-designed observer program is reflected in the weight it is given in the TCS scoring, with 33 out of a total 73 points being related to the observer program. A well-designed observer sampling program requires randomisation of sampling, including stratification of data collection among vessels, trips and hauls. That is, ensuring that sampling is randomised within each of these strata means that data can be concluded as being representative of the entire fishery. This also means that the subsequent scaling up of estimates across a fishery will be representative. If samples are biased, for example due to a low number of participating vessels, it is likely that inaccuracies will exist with respect to the spatial and temporal coverage of the program and also in terms of encompassing industry practices regarding, for example, the keeping of species for bait or for consumption.

Current design issues in the observer programs in each jurisdiction in the SRLF became apparent when comparing these to what is considered best-practice in the TCS. In the SRLF, observer programs have been running in each state for over 15 years, and therefore the longevity of the observer programs in each jurisdiction scored highly. However, deficiencies in the sampling design and spatial coverage of the programs were apparent across the fishery due at least in part to: (i) the opportunistic nature of observer coverage that is necessitated by the need to work only with fishers/vessels that can accommodate observers on board over extended periods at sea, and (ii) the large number of vessels operating across a large fishing area (e.g. 250 licences across 230,000km^2 in SA).

The highest scores for an observer program as defined by the TCS include a full census of hauls, which is unrealistic for this fishery. Participation in the observer programs in this fishery is currently voluntary in all jurisdictions and sampling typically comprises a small proportion of the fleet making the design of a randomised sampling program problematic. Increasing the number of vessels participating in the observer program would improve the spatial and temporal coverage of sampling. Depending on the number and location of vessels participating in the observer program, a fully randomised or stratified randomised sampling design could be employed to partition observer effort according to the timing and location of fishing operations. The importance of this was highlighted in TAS, where the jurisdiction is the whole management zone, where we identified significant spatial differences in bycatch numbers and composition. In particular, bycatch was noted to be high in the Bass Strait islands. Therefore, there is a need to ensure that sampling is spread across the entire fishery in any given season. One possible way to do this would be to partition TAS into four zones (NE, NW, SE and SW) for bycatch monitoring purposes and allocate at least one observer sampling trip in each of these zones in a season.

The need for consistent species level identification across the fishery was identified as another area for improvement in the observer programs. PSA and higher level quantitative analyses need to take into account the life history traits of individual species. In data across all jurisdictions there were species that were identified to the group level rather than species level. This was particularly the case for leatherjacket and wrasse species. Because a number of leatherjacket species and Blue-throat Wrasse were identified as potentially higher risk species, confidence in subsequent analyses was reduced by uncertainty in the proportion of species in a grouped identity that should be attributed to a particular species. Improved species identification could be achieved through the ongoing training of observers, or alternatively could involve subsequent photo identification for species whose identification was uncertain.

Another issue raised with the observer data during the current project was that the fate of bycatch was typically not recorded in the observer programs. While industry reporting of species kept as byproduct is mandatory across all jurisdictions in the SRLF, verification of the proportion of catch kept for bait or consumption would greatly aid in estimating the potential impact of the fishery on individual species. Also, observer data regarding species susceptibility to barotrauma or their likelihood of being preyed upon when discarded is important in assessing risk to those species. While some of this information was captured during the risk assessment workshop discussions, changes in current data collection protocols to include this information would allow for increased confidence in future risk assessments.

Observer programs all recorded numbers of individual bycatch species in sampled pots rather than weights. Weights are the standard international way of expressing bycatch (see Kelleher 2005; NMFS 2011; Gray & Kennelly 2018; Pérez Roda *et al.* 2019) and enable extrapolated estimates of discards to be expressed as a percentage of retained catches (the usual way discard rates are provided). In this project, mean calculated or estimated lengths were converted to an average weight using species level length-to-weight conversion parameters. These mean weights were then used to extrapolate total bycatch amounts as well as biomass by season for individual species. But we acknowledge that this introduces increased uncertainty in our estimates that has not been accounted for. Recording the weight of bycatch directly would place an increased burden on observers, but could perhaps be conducted for a subset of samples in order to increase the confidence in estimates.

There were inconsistencies in the way TEPS interactions were defined and recorded in the observer programs in each jurisdiction. For example, TEPS reporting is left to mandatory industry reporting in SA, where only direct interactions with gear are reported. In TAS, as well as mandatory reporting of TEPS interactions by industry, observers also recorded interactions as “positive”, “negative” or “benign” in TAS, and include sightings and an estimation of numbers recorded . This indicates the need for a consistent definition of TEPS interactions across all jurisdictions and the implementation of a consistent methodology for the recording of interactions. Gathering consistent information about TEPS interactions would allow an assessment of whether there were spatial or temporal changes in interaction types or rates. We therefore recommend that data sheets for TEPS interactions are made consistent for observers across all jurisdictions, and include the recording of when no interactions have occurred. Such information is crucial for management plans to show whether negative interactions with TEPS are an issue and to detect changing trends in encounter rates. For example, Humpback Whale populations have been steadily increasing in Southern Australian waters, and so there is an increased potential for interactions. If observers (and ideally industry) were to record sightings as well as an interactions with gear, changes in risk of entanglements through time and space could be better quantified and management measures implemented if necessary.

Additional improvements in bycatch reporting: industry and supplemental data, database considerations and the analytical approach

In addition to the focus on observer programs, the TCS also scores industry collected and supplemental data that is available for bycatch estimates. Industry bycatch data is collected in a number of small voluntary programs in each state. However, the main focus is often the collection of size data for the target species, and thus the collection of bycatch data is sometimes inconsistent. Preliminary analysis of the program in SA indicated that levels of bycatch reporting by industry were often less than reported by observers due to the focus on reporting of undersize SRL as a recruitment measure. Therefore, improvements need to be made if industry data is to be considered reliable enough to be included in bycatch analyses and reporting. Also, there is the need for industry to improve reporting of TEPS interactions on compulsory reporting sheets as low reporting rates compared to observer data were evident. Improved TEPS reporting would lead to improved spatial and temporal resolution of TEPS interactions and would be useful to better inform any mitigation strategies if they became necessary.

The major issue identified during this project with respect to database and IT considerations was the inconsistency in the way data was recorded between the different jurisdictions. As previously noted, the lack of consistent species identification and different common names used resulted in extra resources being put to standardising, collating and analysing bycatch datasets. There were also inconsistencies in the way TEPS interactions were recorded between the different states and therefore the types of data available for risk analyses differed between jurisdictions. The need for data standardisation between states has been noted previously (see Linnane & Walsh 2011). A bycatch species list distributed to all states along with the relevant data structure required for analyses identified during this project could aid in consistent data collection and enhanced analyses in the future.

The low scores awarded in the TCS for the analytical approach for bycatch data in the SRLF is primarily a product of the lack of historical focus on detailed estimation and reporting of bycatch for this fishery. Methods outlined in this project for scaling up bycatch estimates and data poor stock assessments are commonly used in estimating and reporting on bycatch. Estimates of uncertainty and potential biases can be quantified with these approaches. Therefore the use of these approaches are likely to pass a peer review process and result in a higher score for the “analytical approach” portion of the TCS. Suggested improvements to data collection protocols would also aid in removing biases and improving confidence in estimates.

Risk assessment for bycatch, byproduct and TEP species: factors affecting risk

The PSA and data poor stock assessment methods highlight that potential ecological risk to bycatch species in the SRLF can be attributed to four important factors:

(i) The ‘actual’ risk posed to species is related to the extent the fishery overlaps with a species’ range, habitat and depth preference and the species’ likelihood of incurring post-capture mortality through either being retained as byproduct or through barotrauma and/or subsequent predation. While we have explored this risk to the extent the data allows, in many cases this risk needs better quantification.

(ii) There is currently a lack of information for some of the life history parameters for bycatch species that potentially leads to a ‘false positive’ risk score for that species. That is, a higher risk category is conservatively awarded to species through the PSA risk analysis process due to a lack in knowledge about a species’ life history population traits and how they interact with fishing related mortality to influence population growth.

(iii) There are further data collection issues that prevent the quantification of risk at a higher level. In particular, the lack of species level identification for some groups of bycatch was a hindrance in this respect.

(iv) There is a current lack of scientific understanding of the selectivity and catchability of bycatch species in pot fishing methods.

One of the key factors in scoring the susceptibility aspect of risk in the PSA is the likely level of post-capture mortality (PCM) of individual species. Species that are preferentially kept as bait are obviously have a high level of PCM. Feedback in the workshops during this project indicated that wrasse and leatherjacket species are likely to be preferred bait species for many fishers, and therefore a large proportion of their total estimated catch may actually be retained. However, this assumption is currently based on limited data, therefore making accurate quantification of fishing mortality impacts problematic. Better understanding as to the fate of bycatch is required to understand the likely impacts on populations of bycatch species. The historical underreporting of species used as bait has been noted elsewhere (see Hartmann, Gardner & Hobday 2013), and is further supported by our estimates of total bycatch compared to reported byproduct. One potential solution is that observers record the fate of bycatch (such as an estimate of condition upon release – e.g. whether dead, alive, etc.) as part of their operations. However, it should be noted that the quantifying the post capture survival of discarded bycatch is complicated as it would require tracking of individual discards for a number of days to fully assess survival rates (see Gilman *et al.* 2013).

Barotrauma was also identified as a risk factor for a number of species. Better understanding this factor would aid in future assessments of risk for these species. If observers recorded the barotrauma state of bycatch species, a more detailed analysis could be made for the relationship between depth and the likelihood of barotrauma for key species. For example, previous research has suggested that post-capture survival of Blue-throat Wrasse due to barotrauma shows a strong correlation with the depth of capture (Rimmer & Franklin 1997).

Missing life history parameters for a number of species further complicated assessing risk. This was especially the case for leatherjacket species, with a large number of species having at least three missing life-history parameters necessary for the PSA. To be precautionary, the PSA assigns high risk where there is missing information, thereby creating a higher risk score than may be warranted. For those leatherjacket species where life history parameters were available, risk scores were typically lower indicating that obtaining this missing information should also reduce risk scores for the others. Conversely, any. For bycatch species, where a lack of information relating to life history characteristics is available, further research conducted to obtain information about those traits is required to increase confidence in PSA risk scoring. Acquiring this information might be best achieved through academic research, with species of interest targeted for sampling as part of observer programs in future fishing seasons. Such information would increase the reliability of future bycatch risk assessments in the SRLF. Also, more detailed life history information would allow alternative risk assessment approaches to be undertaken. For example, Sustainability Analysis for Fishing Effects (SAFE), has been shown to provide less biased estimates of risk to species (Zhou et al. 2016). SAFE was not undertaken as part of this project due to the current lack of life history information for a large number of important bycatch species in the SRLF.

We found that a significant proportion of bycatch species were only identified to a grouped taxonomic level, precluding quantification of risk for individual species within those groups. This was particularly problematic for leatherjacket and wrasse species, both identified as being important grouped components of the bycatch taxa in the SRLF. Species within the same taxonomic genus or family may have different life history traits such as maximum age, age at maturity or fecundity that make them vary in susceptibility to the impacts of fishing mortality. Consequently, resolving species-level identification is critical in assessing species population risk. For the observer programs in each state we therefore recommend that ongoing training is conducted in identifying to the species level. Electronic logbooks or species identification kits that contain pictures of bycatch would aid in this respect. The ability to photograph unidentifiable species in the field for subsequent verification would also assist in resolving species identification.

Quantifying the likelihood of the fishing gear (pot) to catch and retain different bycatch species (i.e. selectivity) posed another challenge during the risk assessment process. Such methods have not been developed for pot fisheries and we therefore found it necessary to use expert opinion gathered in workshops to aid the scoring of selectivity risk. The PSA has been historically applied to trawl fisheries and methods for quantifying selectivity have been based around the mesh size of the gear. Research examining the total abundance of species across a reef compared to the proportion that are likely to be caught and retained in a pot (both with and without escape gaps), and the impact of repeated fishing effort across a reef, would greatly aid in assessing pot selectivity/species catchability. Studies combining underwater visual survey techniques with potting experiments across the same reef systems could also provide the data necessary to estimate catch selectivity in SRL fishing operations. Some data may already be available in some cases. For example, research potting in TAS has been conducted in the Maria Island marine reserve, where a time-series of diver survey data also exists. Analysis of this and other data, and a study in depths beyond SCUBA limits using ROVs and/or pots with cameras could be used to understand the selectivity of pot fishing across the fishery.

Our research and previous research suggests that escape gaps are likely to have a very large effect in reducing bycatch levels, particularly for some species. For example, bycatch of leatherjackets and Blue-throat Wrasse was estimated to be reduced by 97.5% and 93.3% respectively in TAS. However, this assessment was made by comparing observer data from commercial trips (open escape gaps) with research cruise data (generally closed gaps), and sampling biases not accounted for may be present. The escape gap effect estimated in TAS was considerably larger than that estimated in the NZ management area of SA (Linnane *et al.* 2011). The estimates of Linnane et al. (2011) were from fishery dependent data collected immediately before and after the introduction of escape gaps in the NZ. It is likely that percentage reductions in bycatch from escape gaps may actually be higher given the relatively higher bycatch reporting rates reported by observers in this fishery. An updated assessment of the escape gap effect after over 15 years of escape gap introduction would be informative as to the current level of effect. If large reductions in bycatch were noted without negative impacts on catch of the target species, management could move towards the introduction of mandatory escape gaps for the NZ.

The finding of large reductions in bycatch when using escape gaps is supported by other research that has suggested overall bycatch reduction up to 80% (Frusher & Gibson 1999). Also, behavioural characteristics of species when pots are hauled has been to shown to significantly affect catch rates of certain species (Asanopoulos et al. 2017). A further understanding of the effect of escape gaps for individual species would aid in better quantifying the selectivity and catchability of species and thus the risk posed to those species.

No TEPS were found to be at high risk from fishing operations of the SRLF in the PSA. However, similarly to the PSA for bycatch and byproduct species, no account is taken of actual population levels of TEPS species. For species that are under threat of extinction due to low and/or declining populations the risk of even one fatality may have serious consequences. One such example is the Australian Sea Lion in SA, where any additional pressure on already low population numbers could lead to extinction (Goldsworthy, Hamer & Page 2007). Management measures, including seal exclusion devices have been put in place in SA to mitigate any potential impacts. If ongoing monitoring reveals issues with juvenile seal entrapment in other areas such as fishing grounds around seal colonies, then managers should consider the introduction of similar measures to reduce risk. Also, while no TEPS was found to be at high risk through operations by the SRLF, it was also noted in workshops that the potential for negative publicity and impacts on social licence to operate due to a single negative incidence involving TEPS is significant. Therefore, the risk from the fisheries' perspective is high, and thus any measures to mitigate risk to TEPS should be considered a priority.

Discarding rates across the management zones in the fishery

The results from this study provide the best available overall estimates of discarding in the SRLF. The only previous estimate came from Kennelly (2018) who found that, of the 84 fisheries examined across 4 jurisdictions (NSW, TAS, Queensland and the Northern Territory), the 2nd ranked fishery in terms of discard rates, was the TAS Rock Lobster Fishery where it was estimated that 66% of the catch was discarded. Because only numbers of individuals were recorded, Kennelly's (2018) estimate relied on the assumption that the average weight of discarded individuals was one third that of retained individuals. The new estimates from the current study are considered to be far more accurate. Discard rates, including undersized SRL in TAS were estimated as 77.9%. For other fisheries examined in this study, the discard rates were lower (40–58%).

Most people would consider lobster trapping as a reasonably selective fishing method, yet these results show that this trap fishery has quite significant levels of discarding. However, if undersize conspecifics are excluded from discard rates, discard levels are far lower, showing that the main contributor to such high levels of discards are undersize lobsters:

Tasmania: 77.9% (incl. undersized RL), 14.3% (excl. undersized RL)

Victoria (EZ): 44.8% (incl. undersized RL), 33.5% (excl. undersized RL)

Victoria (WZ): 58.4% (incl. undersized RL), 13.6% (excl. undersized RL)

South Australia (NZ) 49.6% (incl. undersized RL), 36.3% (excl. undersized RL)

South Australia (SZ) 40.0% (incl. undersized RL), 9.5% (excl. undersized RL)

Under the most commonly used definition of “bycatch” (and that used in this study), the discard of undersize, unwanted conspecifics is still regarded as bycatch and, if possible, such discards should be reduced as much as possible in order to minimize any incidental mortality due to fishing. However, it is important to note that such a high rate of discarding for lobster fisheries does not necessarily reflect the actual incidental mortality of these animals. It is considered likely that the discard mortality of species like lobsters is minimal (see Mills, Gardner & Johnson 2006; Green & Gardner 2009) - a point commonly made regarding the discarding of animals with hard exoskeletons (Gilman *et al.* 2013).

Notwithstanding the possible survival of many of these conspecifics, at least some mortality may be expected due to such practices - from predators as the discarded animals make their way through the water column, and as they seek new habitats and food sources after being displaced (Gilman *et al.* 2013). Clearly it would be ideal, and in the best interests of the SRL fishing industry who rely on subsequently catching these discarded conspecifics, to reduce such discarding as much as possible. The best way to achieve this is by not catching undersize lobsters in the first place. The relatively high contribution of undersize SRL to bycatch levels reported in this study indicates more research is required to assess how escape gaps in pots is required for (or other devices/activities) can further minimise undersize bycatch in this fishery.

Quantitative assessment of key bycatch species

Quantitative analysis of the ten short-listed species for further analysis provided additional insights into the ‘actual’ risk posed to these species. PSA is known to be precautionary in nature, returning a high proportion of false positives - that is, a higher risk is assigned than is warranted (Zhou *et al.* 2016). This is in part due to the fact that the approach does not take into account levels of catch, the size of the population, the likely exploitation rate or any management actions that may already be in place to mitigate risk to that species (Hobday *et al.* 2007). Rather, the idea of PSA is to act as a filter to narrow down the pool of species that warrant more detailed analysis. By considering estimated levels of total catch for the ten species, comparison with catches from targeted fisheries and outputs of data poor stock assessment could be taken into account. Below we discuss the implications of outputs of the quantitative analyses conducted for the ten key species identified as part of the PSA.

Draughtboard Shark

Draughtboard Shark are a key bycatch species for the SRLF, especially in TAS and VIC. In TAS, Draughtboard Shark are one of the dominant bycatch species, particularly when considering the total biomass caught, with an average estimated bycatch of over 200 tonnes, or one-sixth of the TAC for the fishery. While our analysis shows considerable variability in the year-to-year estimate of bycatch, no consistent trend is evident in the 17- and 13-year time-series of bycatch data examined for TAS and VIC, respectively. Industry reporting of byproduct for this species indicates that it is not a preferred byproduct species, and workshop feedback from both observers and fishers indicated that the majority of Draughtboard Sharks caught as bycatch are returned to the water in good condition. Research also suggests that this species is likely to have a high post-capture survival rate (Awruch et al. 2012 and C. Awruch pers. communication). A recent study that examined the cumulative risk posed to Draughtboard Shark as a bycatch species across 19 fisheries (although notably not including the SRLF) concluded that the risk posed to this species was likely to be minimal (Zhou et al. 2019). Therefore, we conclude that added impact from the small amount of byproduct historically reported from the SRLF is unlikely to have had a major impact on populations of this species.

Leatherjackets

The importance of leatherjackets as a bycatch species in rock lobster pots has been discussed in previous research (e.g. Frusher & Gibson 1999; Brock *et al.* 2007; Asanopoulos *et al.* 2017) and is further confirmed by our results; however, further conclusions are hampered by a lack of detailed species level bycatch data. Treating leatherjacket species as a group indicated that total catches in SRLFs are likely to be comparable or sometimes exceed amounts taken in the targeted scalefish fishery in each state. A recent stock assessment in TAS indicate a decline in catch rates of leatherjackets in recent years, where they are primarily a byproduct in the wrasse trap fishery (Moore, Lyle & Hartmann 2019). Low catch rates were primarily attributed to a lack of market demand. While this would indicate that populations are not under high levels of fishing pressure, defining stock status of individual species via the use of bycatch data remains difficult due to individual species identification not being resolved. For the two species of leatherjacket that there was sufficient data to scale-up estimates (Horseshoe Leatherjacket and Degen's Leatherjacket), results were inconclusive as it was still unclear what proportion of unidentified leatherjacket were likely to belong to those species. However, the available data indicates that Horseshoe Leatherjackets are likely to be the most abundant leatherjacket species in SA, suggesting that this species should be a key bycatch reference species there.

Actual levels of post-capture mortality are currently unknown; however, if this proportion is high then our results indicate that leatherjacket species should be a part of ongoing monitoring of bycatch in the SRLF. Workshop discussions indicated that leatherjackets are often kept for bait and may also be susceptible to barotrauma when caught in deeper waters. Different depth preferences for species within families may imply different levels of risk. For example, Degen's Leatherjackets tend to occupy deeper water and therefore may be more susceptible to barotrauma issues.

The risk assessment also revealed that there were important life history parameters missing for a significant proportion of leatherjacket species (see Appendix B). Missing life history parameters were mostly related to age at maturity, maximum age and fecundity. Improved species identification and knowledge of the missing life history parameters for leatherjacket species would greatly aid in refining the assessment of risk and in setting future management reference points for these species. Improved species identification for leatherjackets has also been identified as a recommendation to assist stock assessment of leatherjacket species in the Tasmanian scalefish fishery (Emery et al. 2015).

Blue-throat Wrasse

Alongside leatherjackets, wrasse species are often noted to be dominant as bycatch species in pot fisheries (e.g. Asanopoulos et al. 2017). The scaled-up estimates of total bycatch of Blue-throat Wrasse estimated in this study indicate that in some seasons, catches may represent a significant proportion of the catch taken by jurisdictional scalefish fisheries in some seasons. As stock assessments already exist for this species in each state, it seems reasonable that the estimated bycatch in the SRLF be incorporated into stock assessments in each state. Despite the fact that the total catches of Blue-throat Wrasse are likely to have been underestimated in respective state scalefish stock assessments, the long-term stability of populations of Blue-throat Wrasse have been noted in recent stock assessments (Moore, Lyle & Hartmann 2019).

During the workshops conducted as part of this project is was noted that Blue-throat Wrasse was a preferred bait species for some fishers across the fishery. Blue-throat Wrasse were also noted to be particularly susceptible to barotrauma, and a more detailed assessment of the depth distribution of the fishery in relation to bycatch of this species would be informative in knowing the risk of barotrauma posed to this species from SRL fishing operations. Due to these combined risks it seems likely that a considerable proportion of the total estimate reported here may be subject to post-capture mortality. Gaining a better understanding of the extent of barotrauma risk on post release survival and the use of this species as bait is key to understanding potential risks posed to Blue-throat Wrasse populations.

Ocean Perch

Bycatch of Ocean Perch in the SRLF was found to be primarily restricted to TAS, with only small amounts reported in VIC and SA, where it may also be grouped into catches of unidentified perch species. Projected catch rates of approximately 10 tonnes per season in

TAS indicate that total catch of this species may warrant ongoing monitoring. In workshop discussions, this species was noted to be particularly susceptible to barotrauma, and is often caught in deeper water potentially amplifying the risk to population viability. A more thorough assessment of the spatial and depth distribution of this species when taken as bycatch would be informative in order to better assess the risk of bycatch mortality for this species.

Conger Eel

Total estimated bycatch of Conger Eel was much higher in TAS than for VIC and SA. Also, analysis of trends in the bycatch data indicate declines in catch for this species in VIC and SA. Historical byproduct reporting, if accurate, suggests that perhaps 10% of the bycatch is kept as byproduct. The potential vulnerability of this species associated with its life history traits (slow growth, etc.) indicates the need for further research to be done to understand the proportion of the fleet that is retaining the species as byproduct and the potential market demands for this species. Furthermore, anecdotal evidence suggests that Conger Eels are often difficult to remove from pots, and therefore some fishers use hooks or gaffs for this purpose. Therefore, there is the potential for increased PCM for this species when discarded. Observer recording of the fate and condition of bycatch would help in quantifying the impacts of handling practices for this species.

Reference points for ongoing monitoring of bycatch

We tested three approaches to setting reference points for our key bycatch species in this study: Data poor stock assessments, inter-annual variation in CPUE, and testing for statistically significant trends in time-series of estimated catch levels. All three methods were limited by the data quality available for analysis, and in particular the considerable noise in the estimated catch and CPUE for bycatch species. This limitation is primarily due to the small sample sizes that are typically obtained when sampling bycatch in fisheries. For example, Tuck, Knuckey and Klaer (2013) in assessing trends in bycatch across Commonwealth fisheries noted that data was often insufficient to assess trends across many of the fisheries. Below we discuss each of the metrics in turn, both in terms of the results for the species we applied them to and their advantages and limitations as tools for ongoing monitoring.

The data poor stock assessment approach (Catch-MSY) indicated that recent catch levels of Blue-throat Wrasse in VIC and TAS were unlikely to push the stock to the limit reference point of 20% of the estimated virgin biomass. This result was not surprising given the relatively small proportion that our estimated bycatch represented when compared to the scalefish fishery in each state. The results of the VIC analysis would be considered more reliable due to the much longer time-series of catch available to estimate the model parameters. Typically, a time-series of catch greater than 25 years is desirable for the Catch-MSY approach (Haddon *et al.* 2019), which was only available for VIC. While the limits assessed could be used to set trigger points, the catch landed in the scalefish fishery in VIC is a far more significant influence on stock status. Given that stock assessments are already conducted for Blue-throat Wrasse in all states, it is recommended that estimates of bycatch and subsequent PCM for this species are incorporated into future stock assessments.

The data poor stock assessment results for leatherjackets indicate that estimated bycatch could potentially have an unacceptably high probability of stock status moving towards the limit reference point. However, a key underlying assumption in the Catch-MSY approach are likely to be providing a misleading result. In particular, the assumption that the historical catch directly reflects the stock status is unlikely to be valid. For example, market demand for leatherjackets in TAS is low (Moore, Lyle & Hartmann 2019) and recent catch levels are therefore more likely reflective of market demands rather than actual levels of stock. This situation is further complicated by the fact that species are grouped together. Furthermore, longer time-series of catch were not available that would have strengthened the outcomes from this assessment.

Analysis of the inter-annual variation in CPUE for key bycatch species revealed that the reference point of 20% change between years (see DAWE 2018) is unlikely to be useful for bycatch in the SRLF due to high variability in the data. We found it was necessary to use a reference point of the maximum catch in the time-series, and then compare inter-annual change relative to this in order to reduce the variability in CPUE. Even then, a 50% inter-annual change level appears more sensible for consideration as a reference point, with most species analysed exceeding the 50% level at least once in the time-series of data.

Therefore, using a 50% inter-annual change in CPUE as a reference point could provide a useful trigger to examine the data in more detail to see whether further investigation is warranted. However, it should be noted that for rarer species, catch rates will be low and with small sample sizes inter-annual variability will be high. Therefore, we recommend that if a 50% reference point is breached then there should be examination of the data in more detail, and in particular potential spatial biases in sampling should be investigated. Improved sampling design for the observer program may also reduce some of the inter-annual variability by addressing some of these types of sampling biases.

We found that in analysing the time series of bycatch data for trends, detecting trends in the last 5 years of data returned few significant results, whereas analysing a longer time-series found more significant trends. Due to the inherent noise in bycatch data, analysis of the trend over short time frames such as the suggested 5 years for Commonwealth fisheries (DAWE 2018) may be problematic. While catch or CPUE may fluctuate from year-to-year due to factors such as natural variability, sampling bias or noise due to small sample sizes, longer term trends such as the entire time-series of bycatch data in each state (~15 years) will reveal whether temporal trends are consistent.

Analysis of trends in bycatch (decreasing/increasing) provides perhaps the most intuitive means of creating a trigger for assessing the potential need for management intervention. While caution should be taken in attributing such trends directly to the impacts of the fishery on stock status of these bycatch species in the SRLF, any longer term declines may signal the need to assess the additional impact that these bycatch levels may be having. The species and states/management zones that we found significant longer term trends in bycatch biomass variation displayed trajectories in the total estimated bycatch that were visually evident in the raw plotted data. Therefore, we propose that analysis of statistically significant trends that incorporate sufficiently long time series of bycatch data should form part of ongoing analysis of bycatch data in the SRLF. For example, analysis of the previous 10 years of catch data on a regular basis would help to ascertain whether previously identified trends were continuing and indicate if further analysis as to the driver of that change was necessary. The outcomes of this analysis could inform whether management action for the SRLF is required. We also note that the analysis conducted here was simple, and factors such as temporal autocorrelation were not accounted for. Ongoing analyses could explore more sophisticated methods for detecting long-term trends in bycatch.

Key species for ongoing monitoring

Finfish, including leatherjackets (various species), Blue-throat Wrasse, Conger Eel and Ocean Perch emerged as being key species for ongoing monitoring through the risk assessment and quantitative analyses. We summarise these key species in Table 16, and suggest that these species should be key to ongoing reporting, subject to periodic review. We also include some species of secondary importance in Table 16, and note that although these species were either determined to be at lower risk, or not assessed in a more detailed quantitative analysis, they are nonetheless important bycatch species for the reasons listed. Species of secondary importance could perhaps be recorded as numbers rather than measured or weighed.

Table 16. Species of primary and secondary importance for ongoing monitoring in the SRLF and reasoning for their importance.

Species of primary importance	
Species	**Reasons for ongoing monitoring**
Leatherjackets • All species until a long (10+ years) time series is established at the species level	• Common bycatch • Often kept for bait • Susceptible to barotrauma • Current lack of life history information for many species
Conger Eel	• Relatively common bycatch • Sometimes kept for bait or may be subjected to rough handling • Life history traits make it more susceptible to fishing pressure • Identified declines in catch in EZ of VIC and NZ of SA
Blue-throat Wrasse	• Common bycatch • Commercial fishery exists and estimates of bycatch should be included in stock assessments • Particularly susceptible to barotrauma • Declining trends of bycatch detected in VIC

Continues…

Ocean Perch	• Important bycatch species in Tasmania • Particularly susceptible to barotrauma, and a deeper water species hence high PCM likely • Commercially exploited, but catch varies significantly. Bycatch should be incorporated into stock assessments.
Species of secondary importance	
Species	**Reasons for ongoing monitoring**
Gurnard Perch	• Reasonably prevalent species across all states • Particularly susceptible to barotrauma • May be used as bait
Butterfly Perch	• Reasonably prevalent species across all states • Particularly susceptible to barotrauma • May be used as bait
Barber Perch	• Reasonably prevalent species across all states • Particularly susceptible to barotrauma • May be used as bait
Ocean Jacket	• Important bycatch species in SA • Likely to be used for bait
Bearded Cod/Red Cod	• Relatively common bycatch species in all states • Likely to be kept for bait • Susceptible to barotrauma
Purple Wrasse	• Reasonably common bycatch • Likely to be kept for bait • Susceptible to barotrauma
Snapper	• Reasonably common bycatch species, especially in SA and VIC • Commercially and recreationally important species • Bycatch may be important to include in stock assessments
Draughtboard Shark	• Dominant bycatch species in TAS and VIC • Byproduct kept has fluctuated over time and new markets may become available • Counts unlikely to be time consuming
Port Jackson Shark	• Dominant bycatch species in TAS and VIC • Byproduct kept has fluctuated over time and new markets may become available • Counts unlikely to be time consuming

Conversely, many bycatch species were identified as having low risk from the impacts of the SRLF through the risk assessment. Also, more detailed analysis revealed that risk to some species (e.g. Draughtboard Shark) are likely to be lower than the PSA indicated. As funds for monitoring bycatch will always be limited, de-emphasizing ongoing detailed data collection for species deemed to be low risk would free up resources to focus on species of higher risk. Narrowing down a list of key species for ongoing monitoring allows for larger sample sizes to be collected for those species and hence improved data quality and estimates of actual risk and potential impacts.

Our analyses showed that the time-series of the composition of bycatch in each jurisdiction was relatively stable in over a decade of data collection. The dominant species we have reported have also been noted to be prevalent in previous studies on bycatch in TAS (Frusher & Gibson 1999) and SA (Brock *et al.* 2007; Asanopoulos *et al.* 2017). Current data collection protocols record numbers of all bycatch species, and for species that are considered low risk, this is considered to be unwarranted. A good example of this is Hermit Crabs which are a dominant bycatch across the majority of the fishery. Counting Hermit Crabs is time consuming and unnecessary considering they are returned to water and very likely to survive. Species that are deemed to be low risk such as Hermit Crabs could either be recorded on a presence-absence basis, or only recorded in periodic censuses of all species (see below).

The higher biological productivity of some of the dominant species makes them more likely to be able to sustain higher levels of take, and this is reflected in the risk assessment results. For example, octopus is the dominant byproduct species across the SRLF, however catches remain relatively stable through time indicating that the level of bycatch is unlikely to be detrimental to populations. However, impacts on species with lower productivity or where productivity is uncertain may be higher and need better quantification.

The above key bycatch species identified for focussed monitoring are based on the best available current data and thus should not be seen as a fixed list, but should be subject to periodic review. For example, recording bycatch data at a species level and completing life history information may reduce the potential risk of some of these ratings obtained through the analysis used in this report. Changes in market demands also mean that impacts on certain species could change as the amount of these species being retained as byproduct is likely to follow market demands. For example, Velvet Crab was found to be a key bycatch species in all states, but only a small market currently exists and so most bycatch is discarded. However, if new markets were to open then monitoring the byproduct levels of Velvet Crab may be warranted. Shifts in the supply of bait could also vary through time altering the amount of byproduct used as bait. Where possible, this information should be incorporated into assessing ongoing risk.

A periodic complete census of all bycatch species, for example on a 5 year basis, would allow the detection of any major shifts in the composition of bycatch. For example, a multivariate analysis of bycatch composition every 5 years could allow the detection of significant shifts in assemblages. This would allow a re-assessment to see whether any changes need to be made to the focal species list. However, it should be noted that many bycatch species are targeted by other fisheries and subject to environmental fluctuations that are likely to impact on their abundance. Therefore, changes may not necessarily reflect impacts from the SRLF. That is, the key species identified through the risk assessment process are considered to be more likely to be susceptible to impacts, and thus changes in their biomass are may reflect those impacts. However, these impacts need to also be considered in light of other fishery and environmental pressures.

Conclusion

This report provides the most comprehensive assessment of bycatch in the SRLF to date. We have utilised the long (> 15 years) time series of bycatch data collected in each jurisdiction in order to quantify the important bycatch species and trends in their level of catch over time.

A full assessment of the monitoring program operating in each jurisdiction was conducted and compared to international best practice standards. Through this assessment, various improvements to current practices were identified, including the need for improved sampling design of the observer program and the need for consistent data collection practices across the states.

The risk assessment undertaken for bycatch species confirmed that the SRLF is unlikely to be having major impacts on the populations of bycatch species. However, improved data quality, species level identification and information regarding the fate of bycatch will allow for improved confidence in future assessments. The risk assessment also refined a subset of priority species for ongoing monitoring, with susceptibility risks being primarily related to the keeping of those species for bait or consumption and barotrauma issues.

More detailed quantitative analyses showed that bycatch data for bycatch species is typically noisy, and therefore setting reference points is difficult as inter-annual variation may just reflect sampling noise or natural variation. We therefore recommend that examination of longer-term trends is the preferred approach when assessing whether management action is required.

In summary, this report provides a comprehensive overview of bycatch across the entire SRLF, makes recommendations for improvements in the monitoring program and analysis of bycatch data and provides guidance around assessing performance indicators for ongoing monitoring.

Implications

The outputs of this report have direct implications for fisheries managers, the SRLF industry and the general public. We provide guidelines to help improve the monitoring programs in each state, a list of key focal species for reporting, and reference points and metrics for ongoing monitoring.

By adopting these recommendations, managers can be assured that they are meeting industry best practice in regard to bycatch monitoring. This is also of direct benefit to the industry as regular assessments and export accreditation can be made more streamlined and wider ecosystem impacts of the fishery can be better quantified. Improved assessment of bycatch in the SRLF is also of direct benefit to the wider public as this will lead to increased certainty on sustainable management of SRLF including impacts on bycatch species, noting that both SRL and bycatch species are a publicly owned resource.

Recommendations

Recommendation 1: Improvements be made in the collection and verification of observer data, including recording the weight and fate of bycatch, improved species identification, consistent recording of TEPS interactions and ongoing observer training. The lack of species-level identification was noted to be of particular concern as it compromised risk assessments for certain species. The ability to photograph unidentified species in the field, or the introduction of an electronic logbook system would be of particular benefit.

Recommendation 2: An increase in the current number of vessels involved in the observer program is necessary to allow a fully randomised sample of the fishery's bycatch resulting in less bias and better spatial and temporal coverage. Note that this does not necessarily entail an increase in the number of observer trips, but rather an allocation of effort to ensure optimal spatial and temporal coverage. For TAS, where bycatch levels vary substantially throughout space, at a minimum one observer trip should be conducted in the NE, NW, SE and SW in each season.

Recommendation 3: Missing life history parameters are collected for some of the key species identified in this report. In particular, missing information for several leatherjacket species (see missing attributes column in Appendix B) introduced uncertainty into risk analyses, potentially inflating PSA risk scores. Species samples to assist with this could be collected as part of observer operations to support this research.

Recommendation 4: Species identified in this report as being of both primary and secondary are prioritised for ongoing monitoring through the existing observer programs, while species deemed as low risk may no longer be monitored on a regular basis but form part of a periodic census (see recommendation 5). In contrast, the enumeration of invertebrates such as Hermit Crabs and Velvet Crabs could be ceased to free up valuable observer time. Weights of important bycatch species should be recorded for at least a subset of these important species so that total bycatch biomass can be quantified with more certainty.

Recommendation 5: A periodic census of all bycatch species should be conducted every 5 years to detect any major shifts in the overall assemblage of bycatch species. Ideally this should include counts for all bycatch species in pots sampled by observers in that season.

Recommendation 6: A consistent data recording system be adopted across the three state jurisdictions to allow comparison between states and assessments of bycatch across the whole fishery. In particular, consistent species lists should be adopted as well as a consistent methodology for the recording of TEPS interactions. This could be achieved through having consistent data recording sheets and training to ensure protocols are consistently followed.

Recommendation 7: Analysis of longer time-series (at least 10 years) of trends in bycatch levels is adopted as the preferred metric for management review of potential bycatch impacts. Reference points of an inter-annual change in CPUE of 50% as an initial reference point for further investigation into long-term trends is proposed as a starting point. Trends already identified as being significant in this report are verified to occur over the next period of monitoring and may lead to management actions.

Recommendation 8: Further study into the efficacy and sizing of escape gaps for conspecifics be conducted. While this has been an area of previous research, the large numbers of undersize lobsters that were identified as discarded bycatch in this study is a concern for the SRLF because any mortality incurred by these animals have a direct impact on subsequent catches in the fishery. Furthermore, given the efficacy of escape gaps as a bycatch mitigation measure reported here and elsewhere, improvements in escape gap sizing and design are also likely to impact on bycatch levels.

Recommendation 9: Estimates of bycatch for individual species are incorporated into stock assessments already conducted for those species where appropriate. This may necessitate the smoothing of estimates, which our analyses tended to show as quite noisy.

Recommendation 10: Increased industry participation in the collection of TEPS interactions is encouraged. While our results show that ecological risk to TEPS from the SRLF is likely to be low, having data that quantifies both the sightings and the encounter rates with gear, and whether these rates are changing through time would be useful for ongoing assessments.

Appendices

Appendix A: Length frequency data used to determine mean lengths and subsequent weights for species

Data from observer trips in Victoria and Tasmania. Expert length was the average expected length for the species based on the input of two ecologists and then averaged. This was used where no length data was available for a species.

Common name	No. observations	Mean length	SD length	state	Expert length	Calculated weight (kg)
Cod, Southern Rock	246	38.54	9.01	VIC	30	1.13
Cod, Unspecified	10	41.50	6.87	VIC	30	1.00
Cow Fish, Ornate	1	17.00	NA	VIC	12	0.04
Eel, Conger	72	124.93	27.38	VIC	100	2.56
Gurnard, Unspecified	20	24.05	6.31	VIC	27.5	0.32
Herring cale	1	32.00	NA	VIC	35	0.46
Horseshoe Leatherjacket	32	29.75	4.09	VIC	27.5	0.43
Knife jaw	4	30.50	11.15	VIC	30	0.56
Leatherjacket	2078	29.32	7.92	VIC	22	0.44
Ling, Rock	13	58.54	17.75	VIC	100	0.84
Little Scorpionfish	1	27.00	NA	VIC	8	0.006
Magpie perch	5	29.80	10.03	VIC	30	0.42
Morwong, Banded	4	35.00	10.80	VIC	40	0.70
Morwong, Jackass	15	36.00	6.22	VIC	35	0.61
Morwong, Unspecified	5	33.00	6.28	VIC	NA	0.51
Mullet, Red	3	26.33	6.43	VIC	25	0.22
Mullet, Unspecified	8	22.13	4.36	VIC	25	0.16

Old Wife	2	26.00	2.83	VIC	21.5	0.13
Perch, Barber	4	18.25	7.54	VIC	15	0.05
Perch, Ocean	42	25.52	7.83	VIC	25	0.37
Perch, Unspecified	191	21.04	6.14	VIC	NA	0.08
Ray, Fiddler	1	60.00	NA	VIC	80	1.01
Ribaldo	3	45.33	4.16	VIC	40	1.00
Shark, Blue Whaler	1	171.00	NA	VIC	NA	41.19
Shark, Dog	2	55.50	4.95	VIC	75	0.68
Shark, Draughtboard	478	80.25	11.27	VIC	100	2.70
Shark, Gummy	17	90.88	24.12	VIC	120	1.91
Shark, Other(Unspecified)	3	143.33	66.58	VIC	NA	6
Shark, Port Jackson	468	71.47	20.23	VIC	100	2.37
Shark, White-Spotted Dogfish	1	40.00	NA	VIC	75	0.25
Shark, Wobbegong	7	136.57	22.37	VIC	130	9.14
Silkie fish	42	18.02	5.41	VIC	NA	0.05
Sixspine Leatherjacket	5	34.80	6.22	VIC	30	0.63
Snapper	213	31.39	6.61	VIC	40	0.59
Warty Prowfish	1	21.00	NA	VIC	20	0.17
Wrasse, Bluethroat	6	36.83	6.91	VIC	30	1.32
Wrasse, Unspecified	557	30.68	7.95	VIC	18	0.52
BASTARD TRUMPETER	5	38.50	3.20	TAS	37.5	0.73
Blackstriped goatfish	1	21.50	NA	TAS	25	0.22
BLUE-THROAT WRASSE	74	41.02	6.53	TAS	30	1.33
BROWN-STRIPED LEATHERJACKET	23	30.72	5.43	TAS	23.5	0.44
CARDINAL FISH	1	12.00	NA	TAS	12	0.03

DRAUGHTBOARD SHARK	1548	78.47	11.27	TAS	100	2.70
EASTERN ROCK LOBSTER	21	15.72	18.87	TAS	NA	1.50
GUMMY SHARK	2	102.50	24.75	TAS	120	1.91
JACKASS MORWONG	31	24.24	2.40	TAS	35	0.61
LEATHERJACKET - UNIDENTIFIED	9	30.79	11.12	TAS	22	0.53
PINK LING	2	65.50	12.02	TAS	100	1.23
PIPEFISH	1	5.00	NA	TAS	10	0.0006
PORT JACKSON SHARK	1	78.00	NA	TAS	100	2.37
PURPLE WRASSE	14	35.15	5.14	TAS	29	0.55
ROSY WRASSE	1	74.00	NA	TAS	17.5	0.21
SCHOOL SHARK	1	118.00	NA	TAS	120	7.57
SENATOR WRASSE	2	28.50	0.71	TAS	18	0.07
SIX-SPINE LEATHER JACKET	67	33.25	6.18	TAS	30	0.63
SOUTHERN CONGER EEL	14	114.84	33.94	TAS	100	2.56
STRIPEY TRUMPETER	2	51.25	8.84	TAS	45	1.28
TOOTHBRUSH LEATHERJACKET	7	27.86	1.75	TAS	21.5	0.35

Appendix B: Summary outputs for all species considered in the PSA risk assessment

Complete tables for all byproduct, bycatch and TEP species assessed in the PSA. Each component (byproduct, bycatch and TEPS) has been ordered in terms of overall risk and then in order of the susceptibility risk. This was done as susceptibility was used to filter out medium risk species for further analysis, with high or medium risk species with a susceptibility score > 1.5 being filtered for further analysis.

Note that risk scores are potential risk only, and not actual risk as this method does not account for "the level of catch, the size of the population, or the likely exploitation rate" (Hobday et al. 2007, pp. 135). Also, the method is precautionary and risk scores are elevated where there are missing attributes.

Byproduct species

Species Name	n missing attributes -Number of missing attributes from Table 1	Productivity total (additive) -Sum of productivity attributes from Table 1	Susceptibility total (multiplicative) -Product of susceptibility attributes from Table 1	2D Overall Risk Value (P&S multiplicative) -Euclidean distance from origin on 2D axes of productivity and susceptibility	2D Risk ranking -Overall risk ranking
Blue-throat Wrasse	0	1.29	2.33	2.66	Med
Six Spine Leatherjacket	3	2.14	2.33	3.16	Med
Toothbrush Leatherjacket	1	1.43	2.33	2.73	Med
Degen's Leatherjacket	3	2.14	2.33	3.16	Med
Horseshoe Leatherjacket	3	2.14	1.88	2.85	Med
Mosaic Leatherjacket	3	2.14	1.88	2.85	Med
Ocean Perch	1	2.00	1.88	2.74	Med
Gunn's Leatherjacket	3	2.14	1.88	2.85	Med
Draughtboard Shark	2	2.57	1.65	3.06	Med
Southern Conger Eel	2	2.43	1.65	2.94	Med
Gummy Shark	0	2.29	1.43	2.69	Med
Harlequin Fish	1	2.29	1.43	2.69	Med
Nannygai	1	2.29	1.38	2.67	Med
Wobbegong	0	2.57	1.20	2.84	Med
Broadnose Sevengill Shark	0	2.57	1.05	2.78	Med
White-spotted Dogfish	0	2.57	1.05	2.78	Med
Green-eyed Dogfish	1	2.43	1.05	2.65	Med

Blue Shark	0	2.57	1.00	2.76	Med
Bearded Rock Cod	1	1.71	1.88	2.54	Low
Brownstriped Leatherjacket	1	1.57	1.88	2.45	Low
Velvet Leatherjacket	1	1.71	1.88	2.54	Low
Barber Perch	3	2.00	1.65	2.59	Low
Giant Crab	0	2.00	1.65	2.59	Low
Bridled Leatherjacket	3	2.00	1.65	2.59	Low
Yeollowstriped Leatherjacket	3	2.00	1.65	2.59	Low
Velvet Crab	0	1.43	1.58	2.13	Low
Eastern Rock Lobster	0	1.86	1.58	2.44	Low
Banded Morwong	0	1.43	1.43	2.02	Low
Bastard Trumpeter	0	1.71	1.43	2.23	Low
Purple Wrasse	1	1.71	1.43	2.23	Low
Senator Wrasse	1	1.71	1.43	2.23	Low
Dusky Morwong	0	1.43	1.43	2.02	Low
Cuttlefish	1	1.86	1.43	2.34	Low
Gurnard Perch	3	2.14	1.43	2.57	Low
Striped Trumpeter	0	1.86	1.43	2.34	Low
Luderick	1	1.71	1.43	2.23	Low
Marblefish	3	2.00	1.43	2.46	Low
Pale Octopus	1	1.57	1.43	2.12	Low
Maori octopus	1	1.86	1.43	2.34	Low
Gloomy Octopus	1	1.57	1.43	2.12	Low
Red Cod	1	1.71	1.43	2.23	Low
Crimson Cleaner Wrasse	3	2.00	1.43	2.46	Low
Snapper	0	1.86	1.38	2.31	Low

Rosy Wrasse	1	1.71	1.28	2.14	Low
Grey Morwong	0	1.29	1.28	1.81	Low
Port Jackson Shark	1	2.29	1.28	2.62	Low
Ocean Jacket	0	1.43	1.28	1.91	Low
Sergeant Baker	3	2.29	1.28	2.62	Low
Elephantfish	0	1.71	1.28	2.14	Low
Australian Salmon	0	1.57	1.20	1.98	Low
Herring Cale	4	2.14	1.20	2.46	Low
Blacklip Abalone	1	1.43	1.20	1.87	Low
Sand Flathead	0	1.43	1.20	1.87	Low
Ribaldo	2	2.29	1.20	2.58	Low
Little Gurnard Perch	3	2.00	1.20	2.33	Low
Butterfly Gurnard	0	1.29	1.20	1.76	Low
Common Stargazer	1	2.14	1.20	2.46	Low
Southern Calamari	0	1.43	1.20	1.87	Low
Jackass Morwong	0	1.43	1.13	1.82	Low
Pink Ling	0	1.86	1.13	2.17	Low
Rock Ling	1	2.00	1.13	2.29	Low
Spider Crab	3	2.14	1.13	2.42	Low
Rusty Catshark	2	2.29	1.13	2.55	Low
Barracouta	0	1.57	1.13	1.93	Low
Arrow Squid	2	1.86	1.13	2.17	Low
Latchet	3	2.29	1.13	2.55	Low
Common Sand Crab	4	2.14	1.13	2.42	Low
Sweep	0	1.43	1.13	1.82	Low
Jack Mackerel	0	1.29	1.13	1.71	Low
Yellowtail Kingfish	0	1.71	1.13	2.05	Low
Swallowtail	1	2.14	1.13	2.42	Low

John Dory	2	2.00	1.08	2.27	Low
Knifejaw	3	2.14	1.05	2.39	Low
Albacore	1	2.14	1.05	2.39	Low
Oilfish	0	1.71	1.03	2.00	Low

Bycatch (Discard) Species

Species Name	**n missing attributes** -Number of missing attributes from Table 1	**Productivity total (additive)** -Sum of productivity attributes from Table 1	**Susceptibility total (multiplicative)** -Product of susceptibility attributes from Table 1	**2D Overall Risk Value (P&S multiplicative)** -Euclidean distance from origin on 2D axes of productivity and susceptibility	**2D Risk ranking** -Overall risk ranking
Fiddler Ray	1	2.71	1.13	2.94	Med
Melbourne Skate	4	2.57	1.13	2.81	Med
Western Blue Groper	2	2.43	1.13	2.68	Med
Thresher Shark	0	2.57	1.00	2.76	Med
Rough Rock Crab	4	2.14	1.43	2.57	Low
Butterfly Perch	0	1.29	1.43	1.92	Low
Eastern Orange Perch	0	1.29	1.43	1.92	Low
Red Hermit Crab	4	2.14	1.28	2.49	Low
Red Mullet/Blue-lined Goatfish	0	1.14	1.28	1.71	Low
Warty Prowfish	4	2.29	1.20	2.58	Low
Globe Fish	3	2.29	1.20	2.58	Low
Cleft-Fronted Shore Crab	3	2.00	1.20	2.33	Low
Mado Sweep	3	2.00	1.20	2.33	Low
Yelloweye Mullet	1	1.57	1.20	1.98	Low

Long-spined Sea Urchin	0	1.43	1.20	1.87	Low
Magpie Perch	0	1.29	1.20	1.76	Low
Blackstriped Goatfish	4	2.29	1.13	2.55	Low
Shaw's Cowfish	4	2.29	1.13	2.55	Low
Ornate Cowfish	4	2.29	1.13	2.55	Low
South Georgia spiny plunderfish	4	2.29	1.13	2.55	Low
Common Bullseye	4	2.29	1.13	2.55	Low
Biscuit Seastar	4	2.14	1.13	2.42	Low
Wavy Periwinkle	4	2.14	1.13	2.42	Low
Decorator Crab	4	2.00	1.13	2.29	Low
Barred Toadfish	2	1.86	1.13	2.17	Low
Old Wife	3	2.14	1.08	2.40	Low
Silverbelly	1	1.43	1.08	1.79	Low
Thetis Fish	4	2.29	1.05	2.52	Low
Whiptail	4	2.29	1.05	2.52	Low
Great Cormorant	1	2.14	1.05	2.39	Low
Whelk	4	2.14	1.05	2.39	Low
Nectria Seastar	4	2.14	1.05	2.39	Low
Little Pied Cormorant	1	2.00	1.05	2.26	Low
Banded Stingaree	0	1.71	1.05	2.01	Low
Red Rock Cod (southern red scorpion fish)	1	1.43	1.05	1.77	Low
Silver Dory	0	1.29	1.05	1.66	Low
Cardinal Fish	4	2.29	1.03	2.51	Low
Broadgilled Hagfish	2	2.14	1.03	2.38	Low
Commercial Scallop	2	1.71	1.03	2.00	Low
Tailor	1	1.57	1.03	1.88	Low
Blue Swimmer Crab	1	1.29	1.03	1.64	Low

Southern Bluefin Tuna	0	2.00	1.00	2.24	Low

TEPS

Species Name	**n missing attributes** -Number of missing attributes from Table 1	**Productivity total (additive)** -Sum of productivity attributes from Table 1	**Susceptibility total (multiplicative)** -Product of susceptibility attributes from Table 1	**2D Overall Risk Value (P&S multiplicative)** -Euclidean distance from origin on 2D axes of productivity and susceptibility	**2D Risk ranking** -Overall risk ranking
New Zealand Fur-seal	0	2.43	1.58	2.89	Med
Australasian Gannet	1	2.29	1.58	2.78	Med
Australian Fur Seal	0	2.29	1.58	2.78	Med
Australian Sea-lion	0	2.43	1.28	2.74	Med
Shy Albatross	1	2.43	1.20	2.71	Med
Bottlenose Dolphin	0	2.86	1.18	3.09	Med
School Shark, Tope shark	0	2.57	1.13	2.81	Med
Humpback Whale	0	2.71	1.08	2.92	Med
Leatherback Turtle	1	2.57	1.08	2.79	Med
Gibson's Albatross	1	2.86	1.05	3.04	Med
Antipodean Albatross	1	2.86	1.05	3.04	Med
Tristan Albatross	1	2.86	1.05	3.04	Med
Campbell Albatross	1	2.71	1.05	2.91	Med
Pacific Albatross	1	2.71	1.05	2.91	Med
White-capped Albatross	1	2.71	1.05	2.91	Med
Southern Dogfish	4	2.71	1.05	2.91	Med
Buller's Shearwater	3	2.57	1.05	2.78	Med

Pink-footed Shearwater	3	2.57	1.05	2.78	Med
Providence Petrel	3	2.57	1.05	2.78	Med
Southern Royal Albatross	1	2.57	1.05	2.78	Med
Wandering Albatross	1	2.57	1.05	2.78	Med
Northern Royal Albatross	1	2.57	1.05	2.78	Med
Indian Yellow-nosed Albatross	1	2.57	1.05	2.78	Med
Salvin's Albatross	1	2.57	1.05	2.78	Med
Chatham Albatross	1	2.57	1.05	2.78	Med
Amsterdam Albatross	1	2.57	1.05	2.78	Med
Fairy Prion	3	2.43	1.05	2.65	Med
Flesh-footed Shearwater	1	2.43	1.05	2.65	Med
Short-tailed Shearwater	1	2.43	1.05	2.65	Med
Blue Petrel	3	2.43	1.05	2.65	Med
Kerguelen Petrel	3	2.43	1.05	2.65	Med
Black Petrel	2	2.43	1.05	2.65	Med
Gould's Petrel	3	2.43	1.05	2.65	Med
Great-winged Petrel	2	2.43	1.05	2.65	Med
Soft-plumaged Petrel	3	2.43	1.05	2.65	Med
Buller's Albatross	1	2.43	1.05	2.65	Med
Grey-headed Albatross	1	2.43	1.05	2.65	Med
Black-browed Albatross	1	2.43	1.05	2.65	Med
Light-mantled Albatross	1	2.43	1.05	2.65	Med
Black Swan	1	2.43	1.05	2.65	Med
Ducks,geese and swans	1	2.43	1.05	2.65	Med

Eagles, Hawks, Kites and Sea-eagles	1	2.43	1.05	2.65	Med
Antarctic Minke Whale	1	2.86	1.03	3.04	Med
Minke Whale	0	2.86	1.03	3.04	Med
Sei Whale	0	2.86	1.03	3.04	Med
Bryde's Whale	0	2.86	1.03	3.04	Med
Fin Whale	0	2.86	1.03	3.04	Med
Short-finned Pilot Whale	0	2.86	1.03	3.04	Med
Long-finned Pilot Whale	0	2.86	1.03	3.04	Med
Killer Whale	0	2.86	1.03	3.04	Med
False Killer Whale	1	2.86	1.03	3.04	Med
Pygmy Sperm Whale	0	2.86	1.03	3.04	Med
Sperm Whale	0	2.86	1.03	3.04	Med
Arnoux's Beaked Whale	0	2.86	1.03	3.04	Med
Southern Bottlenose Whale	1	2.86	1.03	3.04	Med
Andrew's Beaked Whale	1	2.86	1.03	3.04	Med
Blainville's Beaked Whale	0	2.86	1.03	3.04	Med
Gray's Beaked Whale	1	2.86	1.03	3.04	Med
Hector's Beaked Whale	0	2.86	1.03	3.04	Med
Strap-toothed Beaked Whale	1	2.86	1.03	3.04	Med
True's Beaked Whale	0	2.86	1.03	3.04	Med
Tasman Beaked Whale	1	2.86	1.03	3.04	Med
Cuvier's Beaked Whale	0	2.86	1.03	3.04	Med
Southern Right Whale	0	2.71	1.03	2.90	Med

Pygmy Right Whale	1	2.71	1.03	2.90	Med
Southern Right Whale Dolphin	1	2.71	1.03	2.90	Med
Dwarf Sperm Whale	0	2.71	1.03	2.90	Med
Blue Whale	0	2.57	1.03	2.77	Med
Loggerhead Turtle	2	2.57	1.03	2.77	Med
White Shark	0	2.86	1.00	3.03	Med
Leopard Seal	0	2.71	1.00	2.89	Med
Elephant Seal	0	2.71	1.00	2.89	Med
Blue Warehou	0	1.29	1.43	1.92	Low
Black Faced Cormorant	1	2.29	1.28	2.62	Low
Eastern Blue Groper	2	2.14	1.28	2.49	Low
Common Weedfish	3	2.29	1.20	2.58	Low
Common Dolphin	0	2.29	1.18	2.57	Low
Dusky Dolphin	0	2.29	1.18	2.57	Low
Common Dolphin, Long-beaked	1	2.29	1.18	2.57	Low
Little Penguin	1	2.14	1.13	2.42	Low
Sooty Shearwater	1	2.29	1.05	2.52	Low
Southern Giant-Petrel	1	2.29	1.05	2.52	Low
Northern Giant-Petrel	1	2.29	1.05	2.52	Low
White-chinned Petrel	1	2.29	1.05	2.52	Low
Grey Petrel	1	2.29	1.05	2.52	Low
Sooty Albatross	1	2.29	1.05	2.52	Low
Crested Tern	1	2.29	1.05	2.52	Low
Caspian Tern	1	2.29	1.05	2.52	Low
Pacific Gull	1	2.29	1.05	2.52	Low
Little Shearwater (Tasman Sea)	2	2.14	1.05	2.39	Low
Fluttering Shearwater	2	2.14	1.05	2.39	Low
Hutton's Shearwater	2	2.14	1.05	2.39	Low

White-bellied Storm-Petrel (Tasman Sea),	1	2.14	1.05	2.39	Low
Black-bellied Storm-Petrel	1	2.14	1.05	2.39	Low
Sooty Tern	1	2.14	1.05	2.39	Low
Common Tern	1	2.14	1.05	2.39	Low
Silver Gull	1	2.14	1.05	2.39	Low
Australian Pelican	1	2.14	1.05	2.39	Low
Herons and Egrets	1	2.14	1.05	2.39	Low
Wilson's Storm Petrel (Subantarctic)	1	2.00	1.05	2.26	Low
White-faced Storm-Petrel	1	2.00	1.05	2.26	Low
Little Tern	1	2.00	1.05	2.26	Low
Australian Fairy Tern	1	2.00	1.05	2.26	Low
Common Diving-Petrel	1	1.86	1.05	2.13	Low
Green Turtle	1	2.43	1.03	2.64	Low
Sawtooth Pipefish	0	1.57	1.03	1.88	Low
Leafy Seadragon	0	1.57	1.03	1.88	Low
Weedy Seadragon, Common Seadragon	0	1.57	1.03	1.88	Low
Pot Bellied Seahorse	0	1.43	1.03	1.76	Low
Short-head Seahorse	0	1.43	1.03	1.76	Low
Bullneck Seahorse	0	1.43	1.03	1.76	Low
Big-bellied / Southern Potbellied Seahorse	0	1.43	1.03	1.76	Low
Hairy Pipefish	0	1.43	1.03	1.76	Low
Javelin Pipefish	0	1.43	1.03	1.76	Low
Briggs' Crested Pipefish, Briggs' Pipefish	0	1.43	1.03	1.76	Low
Knife-snouted Pipefish	0	1.43	1.03	1.76	Low

Brushtail Pipefish	0	1.43	1.03	1.76	Low
Deep-bodied Pipefish	0	1.43	1.03	1.76	Low
Half-banded Pipefish	0	1.43	1.03	1.76	Low
Australian Smooth Pipefish, Smooth Pipefish	0	1.43	1.03	1.76	Low
Spotted Pipefish	0	1.43	1.03	1.76	Low
Wide-bodied Pipefish, Black Pipefish	0	1.43	1.03	1.76	Low
Ring-backed Pipefish	0	1.43	1.03	1.76	Low
Pug-nosed Pipefish	0	1.43	1.03	1.76	Low
Mollison's Pipefish	0	1.43	1.03	1.76	Low
Australian Long-snout Pipefish, Long-snouted Pipefish	0	1.43	1.03	1.76	Low
Tucker's Pipefish	0	1.43	1.03	1.76	Low
Upside-down Pipefish	0	1.43	1.03	1.76	Low
Rhino Pipefish, Macleay's Crested Pipefish	0	1.43	1.03	1.76	Low
Trawl Pipefish, Kimbla Pipefish	0	1.43	1.03	1.76	Low
Red Pipefish	1	1.43	1.03	1.76	Low
Mother-of-pearl Pipefish	0	1.43	1.03	1.76	Low
Booth's Pipefish	0	1.43	1.03	1.76	Low
Port Phillip Pipefish	0	1.29	1.03	1.64	Low

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