

design, trial and implementation of an integrated, long-term bycatch monitoring program

road tested in the Northern Prawn Fishery



FINAL REPORT
FRDC Project No. 2002/035



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DEDICATION to Dr Burke Hill

The Bycatch Monitoring Project team dedicates this report to the memory of Dr Burke Hill – an eminent scientist and valued colleague – for his commitment and groundbreaking work on the impacts and management of fishing off northern and eastern Australia. Burke was a crustacean ecologist and fisheries biologist, and former acting chief of CSIRO Division of Fisheries. His early research in southern Africa mainly focused on fish and crustacean ecology and behaviour in lakes and estuaries, and he was largely responsible for developing a quantitative approach to estuarine ecology in the region. In Australia, he continued his crustacean research, particularly on prawns and crabs, and developed groundbreaking projects to study the elements and impacts of fishing off northern and eastern Australia.



Burke was strongly involved in the development of the much respected management arrangements within the Northern Prawn Fishery today, which included representation as the scientific member on NORMAC, the development of the NPF; a management design to reduce the impact of fishing on the spawning of prawns and Australia's first Bycatch Action Plan, publications on the fate of bycatch discarded from trawlers and critical research into vulnerable sawfish species in northern Australia. Burke encouraged the Australian Fisheries Management Authority, research and industry to embrace ecosystem approaches to fisheries management, long before the concept was broadly accepted. His contribution to fisheries management was recognized in 1986 with an award from the Queensland Commercial Fishermen's Organisation. In 1992 Dr Hill was appointed to the FRDC board, a position he held until 1997. He also served on the board of the Queensland Fisheries Management Authority.



He played a major role in the redevelopment of the Cleveland laboratory in 1991-92 and in the general growth and development of CSIRO Fisheries and Marine Research.

Dr Burke Hill is greatly missed as a researcher, mentor and friend.

1. NON TECHNICAL SUMMARY

2002/035	Design, trial and implementation of an integrated, long-term bycatch monitoring program, road tested in the Northern Prawn Fishery
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Objectives

1. To design, trial and implement an integrated long-term bycatch monitoring program; that addresses (i) total amount of bycatch, (ii) protected species and (iii) high risk species in the most cost-effective manner possible using the NPF as an example.
2. To transfer ownership, momentum and responsibility of ongoing monitoring to AFMA and NORMAC.
3. To develop a new, innovative, quantitative method for defining the risk to the sustainability of bycatch species from prawn trawling, and apply the model to the bycatch of the NPF (original objective: to validate the risk assessment of the Northern Prawn Fishery bycatch species recognised as 'high risk').
4. To provide the first description of the bycatch from the Joseph Bonaparte Gulf.

Outcomes Achieved To Date

<p>This project has designed a process to develop a cost-effective and acceptable bycatch monitoring program for a fishery with diverse and abundant bycatch. We have designed a new quantitative risk assessment technique and a bycatch monitoring program that is focused mainly on the species of concern. The monitoring program has been established in the industry now for four years, although the sampling effort levels need to be increased to ensure adequate species coverage in the years ahead.</p>

General Approach

The need for a bycatch monitoring program in Australia's Northern Prawn Fishery (NPF) stemmed from the EPBC Act 1999 which requires the maintenance of biodiversity and demonstration of sustainability for all species and habitats impacted by Australian industry activities; and the NPF's commitments under its Strategic assessment by the Department of Environment and Heritage (DEH). An important underlying premise is that all species, communities and habitats are valuable and demonstrating sustainability should not be limited only to those of particular charisma, notoriety or concern. This meant that a program is required to assess and manage all species, communities and habitats impacted by NPF activities.

The NPF consists of two fishing seasons; a 'banana' season primarily targeting banana prawns (*Penaeus merguianus* and *P. indicus*), and a 'tiger' season primarily targeting two species of tiger prawns (*P. semisulcatus* and *P. esculentus*) and two species of endeavour prawns (*Metapenaeus endeavouri* and *M. ensis*) and the red-legged banana prawn (*P. indicus*) (Dichmont et al., In press). NPF bycatch includes six species of sea turtles, more than twelve species of protected sea snakes, about 50 species of sharks, rays and sawfish, and hundreds of species of teleost fish and epibenthic invertebrates (Stobutzki et al., 2001; Brewer et al.,

2006). The diversity and abundance of this bycatch has required an innovative new approach to assessing the sustainability of all species in a cost-effective and robust manner.

Firstly, a quantitative risk assessment – Sustainability Assessment for Fishing Effects (SAFE, described below) – was developed and used to select species that are ‘at risk’ from this fishing activity. This allows the bycatch monitoring program to focus on species of concern only, greatly reducing the cost and difficulty of demonstrating sustainability for all or a more arbitrary suite of bycatch species. Secondly, a comparison of different monitoring methods was made in order to develop the most cost-effective bycatch monitoring strategy for the broad range of bycatch species caught in the NPF. Descriptions of the bycatch from NPF banana prawn fisheries were also made to complete our knowledge of the species composition of the main sub-fisheries in the NPF. This approach, along with a collaborative model and protocols between AFMA and CSIRO for managing and delivering the bycatch monitoring program, has provided the information allowing recommendations for implementing of a cost-effective bycatch monitoring program for the NPF.

Selecting an effective sampling regime for monitoring diverse trawl bycatch

We compared five potential methods for monitoring bycatch – logbooks, requested industry collections, crew-member observers (CMOs), scientific observers and fishery-independent surveys – over two years and 4 seasons; tiger season 2003, banana season 2004, tiger season 2004 and banana season 2005. After the first year the data collection was examined and altered, where necessary, to improve the design.

Logbook data from 23,470 vessel days was reliable and comparable with previous studies for sea turtles but not for other groups, but has the advantage of collecting the large amounts of data required to detect changes in rarer species. Requested industry collections returned bycatch sub-samples from 370 trawls, but these had a high and unpredictable rate of bias due to being collected from the hopper on more than one-third of occasions (see Heales et al., 2003). CMOs participated in training workshops each year and collected data from 5,633 trawls. They demonstrated ability to collect reliable and accurate data for sea turtles, sea snakes, sawfish and other selected elasmobranchs. Scientific observers collected reliable and accurate bycatch data from 148 trawls for all targeted species groups. Fishery-independent surveys also collected reliable and accurate bycatch data from 493 trawls during this study. The bycatch monitoring program will be able to combine the high sampling power of the fishery-dependent methods to collect monitoring data for most of the targeted species groups (sea turtles, sea snakes, syngnathids, sawfish, ‘at-risk’ elasmobranchs and fish), with the higher acceptance of fishery-independent methods' ability to provide additional data and validation.

Assessing the composition and structure of species communities is reported to be a key indicator for ecosystem-based fisheries management (Rochet and Trenkel, 2003; Fulton et al., 2004a, 2004b; Hall and Mainprize, 2004). The dominance and high diversity of bycatch species in trawl catches warrant ongoing assessment to determine whether trawl impacts change the composition and structure of these demersal communities (see Sainsbury et al., 1988), but without necessarily putting individual species at risk. The assessment of bycatch species composition can only be based on data collected by scientific observers or fishery-independent surveys as the other methods collected an unacceptably high proportion of inaccurate data.

Fishery-independent bycatch surveys can control and minimise spatial variation and diel and lunar periodicity from year to year, providing more precise assessments. They can piggyback on the current NPF prawn monitoring mid-year surveys and provide certainty in data collection from year to year. They are also the only method that can provide an option for collecting control data (samples outside the high effort areas) to interpret whether any changes in species composition and structure are due to fishing impacts or some other source (e.g. climate change – see McFarlane et al., 2000).

Estimates of total bycatch are also an accepted indicator of fishery impact and can be made by all methods available to the bycatch monitoring program. However, models using these data based on fleet effort patterns are needed to assess fishery impacts on bycatch in the future. Research proposals for the construction of these models are currently under consideration.

Implementation, effort and cost scenarios

Species ‘at risk’ from NPF fishing activity are recommended for inclusion in the bycatch monitoring program (BMP) along with listed Threatened, Endangered and Protected (TEP) species. The BMP will then collect medium and long-term data sets on each to determine whether (i) there is ongoing risk (= remain in the monitoring program); (ii) the risk is not real or removed (= cease monitoring); or (iii) the impact is unsustainable. In the latter case the fishery should formulate and instigate a specific threat abatement plan to remove this risk (e.g. use a specific bycatch reduction device to reduce its catch and improve survival). The risk assessment should be repeated periodically (e.g. five-yearly) to incorporate any new data and provide up-to-date assessment of risk. The remaining many hundreds of species need only be re-assessed if there are major changes to effort or spatial management of the fishery.

A combination of sampling methods can provide a cost-effective bycatch data collection program. It relies on using the sampling power of the fishing fleet to collect adequate sample sizes for a range of TEP and ‘at-risk’ species. It also provides a high level of broader stakeholder acceptance by including annual training for CMOs and validation of data for all species groups using a combination of methods including scientific observers and fishery-independent surveys.

The total number of trawls that the entire fleet is capable of monitoring is about 33,600 trawls (70 vessels x 120 days x 4 trawls), based roughly on 2006 effort levels. This represents the logbook data collection capability. One CMO can provide data from 384 nets per year and one scientific observer from 864 nets per year. CMOs and scientific observers can combine to collect acceptable and validated data for sea snakes, syngnathids, sawfish, and selected elasmobranchs and fish. However, the proportion of these species that can be assessed by the BMP will depend on the level of effort supported in the program.

A reduced effort scenario of ten CMOs and one scientific observer will allow the detection of declines for two species of sea turtles, nine of eleven species of sea snakes, one of sawfish, none of the three species of ‘at-risk’ elasmobranchs, and both species of ‘at-risk’ teleost fish. This scenario costs between \$262,000 and \$565,000 per year, depending on the level of inclusion and coverage of species composition and structure assessments, but excludes three of the selected TEP and ‘at-risk’ species.

The recommended BMP (15 CMOs and 3 scientific observers) can deliver sustainability assessments for two species of sea turtles, ten (of eleven) species of sea snakes, one species of sawfish, two of three species of ‘at-risk’ elasmobranchs, both species of ‘at-risk’ teleost fish,

species composition and structure and total bycatch discards. This recommended scenario would cost an estimated \$728,000 per year, and includes fishery-independent surveys, with control regions in both the Gulf of Carpentaria (GoC) and Joseph Bonaparte Gulf (JBG). This program without assessment of species community structure and composition in any region costs an estimated \$425,000 per year.

The collection of bycatch data during the BMP will provide detailed information on trends in catches for each species included in the program. The program also requires pre-determined responses to help interpret trends and manage species sustainability (e.g. species-specific limit reference points, triggers and management actions). However, these have not been previously developed for bycatch species where limited biological information is available and the BMP will participate in national Ecological Risk Assessment forums for guidance and consistency across fisheries on these issues.

For some of the rarer TEP or 'at-risk' bycatch species there is not enough sampling capability in the industry to detect changes in their populations based on catch information. Or for syngnathids, they are too poorly known at this stage to include in this sampling effort scenario. In both these cases alternative management strategies should be considered in order to enhance the long term sustainability of these species and provide protection for demersal communities impacted by this fishery. As an example, closing low effort areas as a way of promoting rebuilding of sessile benthic habitats may provide a measure of protection for a range of species and hence some insurance for long-term sustainability of these species and habitats.

Validating the NPF ecological risk assessment

The NPF has been using a semi-quantitative attribute-based ecological risk assessment method to focus the bycatch monitoring program on the highest risk bycatch species. However, to validate this method we incorporated new data to assess the change in sustainability of species following the introduction of Turtle Excluder Devices (TED) in Australia's Northern Prawn Fishery. Population recovery ranks changed for 19 of the 56 elasmobranch species post-TEDs. However, ten species unexpectedly showed a decrease in sustainability. This was due to TEDs successfully excluding large animals from the catch, resulting in a lower mean length at capture, which reduced the recovery ranks for two criteria relying on length data. This falsely indicates that TEDs increase the impact on pre-breeding animals, thus reducing the recovery potential of these species. Hence, the existing attribute-based risk assessment methods are likely to be inadequate for reflecting even the most obvious changes in fishing impacts on bycatch species. Industry and management can benefit greatly from an approach that more accurately estimates absolute risk and this approach is described below.

SAFE - A new quantitative ecological risk assessment approach

A critical part of the approach to providing relevant, cost-effective and ongoing management of the NPF bycatch monitoring program has been the development of a new quantitative ecological risk assessment. We refer to this method as a Sustainability Assessment for Fishing Effects (SAFE). Considering the high diversity, rarity, limited data, and low economic value of NPF bycatch, we took a unique approach that could accommodate these difficult issues, but also provide biological meaningful results. The SAFE approach broadly consists of two separate components: 1) determining the fishing mortality rate of a species based on their spatial overlap with the fishery and their vulnerability to being capture by prawn trawls, and

2) use of basic life history parameters to assess the status of the population relative to a biological reference point at the estimated fishing mortality rate.

We attempted to make the SAFE method as conceptually simple, and operate using as few data as possible, in order for it to be transferable to other fisheries. Hence, the first component of SAFE is based on simple detection-nondetection (or presence-absence) data, which is generally cheaper to collect and more widely available than count data. This innovative model estimates the abundance of randomly or aggregated populations from detection-nondetection data collected from repeated surveys, by incorporating detection probability and site occupancy. The model forms the basis of the first component in our SAFE method, but should have wide reaching ecological application.

Separate risk assessments were undertaken on elasmobranch and teleost bycatch species, primarily due to the differences in their general life history traits. Although SAFE may be used in isolation as a fisheries management tool, owing to large number of bycatch species in the NPF and the high uncertainty around some estimates of life history parameters used in the model, we used the SAFE as a 'screening' tool to identify 'at-risk' species. These species were then nominated as candidates for the long-term monitoring program in order to gather more detailed biological and time-series catch data. Their sustainability could then be assessed in a more rigorous fashion than could be achieved by only using SAFE.

A total of 51 elasmobranch and 478 teleost species were recorded as trawl bycatch in the NPF managed region in various surveys conducted by CSIRO and state fisheries agencies between 1979 and 2003. Of these species, only six were identified as being 'at risk' of becoming unsustainable, since the fishing mortality rate on these species exceeded a biological reference point, μ_{crash} . This reference point is the minimum fishing mortality rate that would eventually drive a population to extinction. This reference point is used in stock assessment of target species and usually requires more biological and catch data than is generally available for low-value, little-studied bycatch species. However, by using natural mortality as a surrogate for fishing mortality (Thompson, 1993) we were able to establish a practical and biologically meaningful reference point for bycatch that can be easily incorporated into existing fishery management strategies.

The 'at-risk' species comprise three elasmobranch species: Banded Wobbegong (*Orectolobus ornatus*), Blotched fantail ray (*Taeniura meyeni*) and Porcupine ray (*Urogymnus asperrimus*), and two teleost species: Dwarf lionfish (*Dendrochirus brachypterus*) and Raggy scorpionfish (*Scorpaenopsis venosa*). However, in order for the true sustainability of species to be assessed the cumulative impacts of all fisheries in the NPF managed region needs to be the focus of future work. This will be particularly important for elasmobranchs since they are most vulnerable to decline from fishing activities, mainly due to their generally slow growth, low natural mortality rate and low reproductive potential (Stevens, 1997; Walker, 1998). Although the vast majority of the NPF elasmobranch bycatch species were assessed as sustainable – including narrow and green sawfish – many are target species in state and Commonwealth-regulated gillnet and longline fisheries (e.g. Qld N3 and N9 fisheries) (Zellar and Snape, 2006) and more recently, targeted in Illegal, Unregulated and Unreported (IUU) fisheries for their fins. Because our SAFE approach is primarily based on the geographic distribution of a species, it is likely to successfully quantify cumulative impacts of fisheries in a discrete region by simply incorporating fishery-specific catchability, escapement and post-capture survival.

To transfer ownership, momentum and responsibility of ongoing monitoring to NORMAC and AFMA

Responsibility and ownership for the long-term monitoring program was gradually transferred to AFMA throughout the collaborative AFMA and CSIRO project. In order to maintain the momentum of the long-term monitoring program, a series of detailed protocols have been developed to ensure that corporate knowledge gained during the project on the best ways to manage the long-term monitoring objectives are clearly defined. Protocols that are critical to the continued successful running of both CMO and scientific observer programs will be easily accessible to staff, particularly new staff, in order to ensure ownership and continuity of the program. Protocols defining the different responsibilities for both AFMA and CSIRO are clearly described to guide the organisation, maintenance and reporting for this program.

Transfer of responsibility and ownership to NORMAC will occur with the presentation (in early December 2006) of the projected monitoring budget for the 2007/08 financial year. The response of NORMAC to the projected budget and work plan will provide future direction and funding for bycatch monitoring program in the NPF. This program requires adequate, long term funding to ensure ongoing participation and enthusiasm of fishers and staff and the subsequent sampling power to allow the ongoing assessment of the species targeted in this program. AFMA will endeavour to secure continued funding for the training of CMOs and will allocate 40% of a full time employee to run this part of the program. This position will ensure regular contact with the participants to encourage them and to maintain interest and ensure long-term commitment. AFMA will also allocate funding for scientific observer coverage during both fishing seasons in the NPF.

Bycatch of the Joseph Bonaparte Gulf and Gulf of Carpentaria banana prawn fisheries

The Joseph Bonaparte Gulf banana prawn sub-fishery is an important component of Australia's Northern Prawn Fishery. However, the species composition of the large volumes of bycatch caught in this region is poorly known. We sampled the prawn trawl bycatch and described 195 taxa from 85 families. The species composition of this bycatch is distinctly different from that of other tropical regions, including the neighbouring Gulf of Carpentaria in the NPF. The estimated 4,868 t of bycatch taken annually in the JBG consists of mainly teleosts (4486 t), invertebrates (382 t) and small elasmobranchs (66 t). Eight species have never been recorded from other bycatch studies in northern Australia.

Bycatch of Gulf of Carpentaria banana prawn trawl fishery was also described for the first time. An estimated 1502 t of bycatch is taken annually in this fishery and comprises 43.5% of the total catch per trawl. It is characterised by small teleosts (75.3%), invertebrates (1%) and small to medium sized elasmobranchs (23.1%). While the overall bycatch was highly diverse (226 spp.), the assemblage structure was significantly different to the tiger fishery bycatch. The differences in bycatch between these fisheries are due to differing gear and fishery operations required for each target species.

Keywords: bycatch, bycatch monitoring, risk assessment, Joseph Bonaparte Gulf, sea turtles, sea snakes, syngnathids, sawfish, at-risk.

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3. BACKGROUND

There are increasing requirements for Australian fisheries to demonstrate ecologically sustainable development. This is essential for fisheries to conform to legislative requirements and to maintain a high level of acceptance of their practices with national stakeholders and in the current global market place. Under Australian legislation, in particular the Environmental Protection and Biodiversity Conservation (EPBC) Act 1999, all Commonwealth and export fisheries must have management practices in place that address the long-term sustainability of all species impacted. There are also increasing pressures from the international marketplace that imported products are sourced from industries that can demonstrate environmentally sound practices. For example, the United States does not import prawns from trawl fisheries that cannot demonstrate similar exclusion rates of protected sea turtles to those required in the US – a practice recently approved by the World Trade Organisation. An increasing number of fisheries are seeking certification by organisations such as the Marine Stewardship Council, which provide internationally accepted standards and certification of sustainable practice that can be used to value-add products in many international marketplaces.

An integral part of demonstrating ecologically sustainable practices in fisheries is the measurement of impact and reduction of identified negative impacts on the marine environment. A significant part of this is effective long-term monitoring of all target and bycatch species. However, this is yet to be implemented in most fisheries partly due to the complexity of monitoring a potentially large numbers of rare, low value and data-poor bycatch species. Designing and implementing such a program requires significant investment in consultation, trialling options and undertaking cost-benefit analysis of potential options. Undertaking this in one fishery will provide a general monitoring framework that other fisheries can build on.

Prawn trawling is one of the least selective forms of fishing (Alverson, 1994). Assessing and demonstrating that it can be an ecologically sustainable practice has been an important part of recent fisheries research and management (e.g. Milton, 2001; Stobutzki et al., 2001, 2002; Diamond, 2005; Hall and Mainprize, 2005). In particular, the Northern Prawn Fishery (NPF) has been a national leader in addressing bycatch sustainability issues via its Bycatch Action Plan (BAP). It has also strongly supported the development and funding of several research projects, which provide assessments of the sustainability of non-target species including (i) the joint BRS/CSIRO/FRDC (1998/202) sea turtle monitoring project, (ii) the CSIRO/QDPI/FRDC (1996/257) project to assess the sustainability of bycatch species caught by NPF trawlers, and (iii) the CSIRO/AMC/Industry/FRDC (2000/173) project to assess the performance of BRDs and TEDs in the NPF, (iv) the CSIRO/FRDC project to quantify the effects of trawling on seabed fauna (FRDC2002/102). These projects have made critical steps towards measuring or subsequently reducing many of the identified negative impacts on the marine environment, and beginning a culture of accountability of environmental impacts imposed by the fishery. The NPF now requires a long-term integrated bycatch monitoring program to demonstrate that previous research and subsequent management strategies implemented in the fishery have been successful in ensuring the fishery is ecologically sustainable. Given the complexity of the NPF, this provides an ideal opportunity in which to develop general monitoring principles and procedures for other Australian fisheries to adopt. This project aimed to design, trial and implement such a program in the NPF, and hand it over to NORMAC and AFMA for its long-term responsibility. This research capitalises on the FRDC's and the NPF's past investments in bycatch research by providing a mechanism for the NPF to measure its ongoing impact on the bycatch taken by the fishery, and its intention

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to ensure ecological sustainability of these species. This research provides a guide for other Australian fisheries on how to implement and design such a program. The outputs include evaluations of different long-term monitoring approaches, the effectiveness and benefits of each, and how to combine approaches and procedures to ensure effective, integrated, long-term monitoring in Australian fisheries.

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4. NEED

The design and implementation of an integrated, long-term bycatch monitoring program has not been undertaken in any Australian prawn trawl fishery. The diversity and complexity of bycatch issues in most fisheries means that designing such a program is a significant challenge (Smith et al., 1997; Diamond, 2005; Heales et al., 2003, in press). Undertaking this in one fishery will develop a tested framework, protocols and procedures that can be transferred to other fisheries, particularly due to the collaboration with AFMA.

This project addressed a range of issues currently faced by Australian fisheries, including (i) how to establish a long-term monitoring program for non-target species; (ii) how the industry can best measure its impact on bycatch species; (iii) how to increase industry's ability to provide validated and high quality data on non-target species, that are acceptable to all stakeholders; and (iv) how to establish and manage fishery-dependent monitoring programs that will be successful in the long term.

The Northern Prawn Fishery (NPF) has been proactive in supporting research aimed at ensuring ecological sustainability, particularly regarding bycatch. This has shown significant benefit during the 'road tests' of EA's strategic assessment guidelines, with the NPF being the first Australian fishery to undertake an ecological risk assessment for bycatch (FRDC 1996/257). The risk assessment identified species that were at highest risk from trawling, and there is now a need to take the next step and collect information on these species to determine the actual extent of this risk. There has also been a substantial amount of pilot research evaluating the most suitable monitoring methods for this fishery. This project provides the NPF with a cost effective process to demonstrate to the Australian and international community its willingness and ability to monitor its impacts on the species groups of most concern. This is an important step that will also have to be made by other Australian fisheries to conform to relevant fisheries and environmental legislation such as the *Environment Protection and Biodiversity Conservation Act (EPBC) 1999*.

This research forms part of an overall ecosystem-based fisheries management plan for the NPF (See NPF Management Plan, 1995 and NPF Bycatch Action Plan, 2003). The monitoring program will be designed to meet the commitments of the NPF Bycatch Action Plan and EA strategic assessment guidelines. This research will also address NORMAC's high priority research area of the Effects of Fishing. It specifically addresses the priorities of assessing bycatch, ensuring the sustainability of bycatch and the development of monitoring programs for assessing BRDs under commercial conditions. The project also addresses FRDC Effects of Trawling subprogram priorities regarding bycatch: "Methods for measuring and monitoring bycatch and the quantification of the direct impacts on associated populations and communities of bycatch species".

The project also takes the vital first step towards understanding the bycatch issues of the Joseph Bonaparte Gulf (JBG) and Gulf of Carpentaria (GoC) banana prawn trawl fisheries. The JBG represents about 20% (\$10-20 million/y) of the total NPF banana prawn catch and approximately 65% of the NPF's red-legged banana prawn catch (Loneragan et al., 2002). However, the bycatch of these sub-fisheries are very poorly known. A process for monitoring and managing the impacts of the NPF must include both the JBG and GoC banana fisheries. In order for this to be successful, a basic knowledge of the species impacted and their variability in space and time is required to prevent inadequate coverage of the bycatch species. The fishing strategies employed in these sub-fisheries also differ, which may have important implications for the effectiveness of monitoring options.

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5. OBJECTIVES

1. To design, trial and implement an integrated long-term bycatch monitoring program; that addresses (i) total amount of bycatch, (ii) protected species and (iii) high risk species in the most cost-effective manner possible using the NPF as an example.
2. To transfer ownership, momentum and responsibility of ongoing monitoring to NORMAC and AFMA.
3. To develop a new, innovative, quantitative method for defining the risk to the sustainability of bycatch species from prawn trawling, and apply the model to the bycatch of the NPF (original objective: to validate the risk assessment of the NPF bycatch species recognised as ‘high risk’).
4. To provide the first description of the bycatch from the Joseph Bonaparte Gulf.

After the start of the project, in January 2004, a proposal by CSIRO was accepted by FRDC to amend the third objective and include the development of a new quantitative method for defining the risk to the sustainability of bycatch species. This and the original objective are included above and addressed in Section 5.4.

Objective 4 was also expanded to take advantage of sampling opportunities to include the first detailed description of the bycatch of the Gulf of Carpentaria banana prawn trawl fishery (see Section 5.5).

5.1 General Approach

The need for a bycatch monitoring program in Australia’s Northern Prawn Fishery (NPF) stemmed from the *Environment Protection and Biodiversity Conservation Act* (EPBC) 1999 which requires the maintenance of biodiversity and demonstration of sustainability for all species and habitats impacted by Australian industry activities. Furthermore, a bycatch monitoring program is needed to satisfy the requirements of the NPF Bycatch Action Plan (NORMAC, 2003), which is to minimise the impact on bycatch species and protect habitats of vulnerable species of marine life. An important underlying premise in this project is that all species are equally valuable (*sensu* EPBC Act 1999) and demonstrating sustainability should not be limited only to those of particular charisma, notoriety or concern. Consequently, a program was required to assess, where possible, all species groups impacted by NPF activities. This project aims to provide a bycatch monitoring program that includes all bycatch species including sea turtles, sea snakes, sharks, stingrays, sawfish and all the small fish, invertebrates and other organisms.

The NPF consists of two temporally separated sub-fisheries; the ‘banana prawn fishery’ primarily targets banana prawns (*Penaeus merguianus* and *P. indicus*), and the ‘tiger prawn fishery’ primarily targets two species of tiger prawns (*P. semisulcatus* and *P. esculentus*) and two species of endeavour prawns (*Metapenaeus endeavouri* and *M. ensis*) and the red-legged banana prawn (*P. indicus*) (Dichmont et al., In press). Although each fishery opportunistically targets other species, depending on the availability of the main target species, the two seasons are considered quite different based on gear type and fishing method. This is explained in detail in Section 5.5.4. Both fisheries catch significant quantities of diverse bycatch and

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methods for monitoring were compared during both seasons over two years: tigers 2003, bananas 2004, tigers 2004 and bananas 2005. After the first year (end of bananas 2004), the data collection approach was examined and altered, where necessary, to improve the design, or to further test the ability and limitations of a particular method of collecting data.

NPF bycatch includes six species of sea turtles, more than twelve species of protected sea snakes, about 50 species of sharks, rays and sawfish, and hundreds of species of teleost fish and epibenthic invertebrates (Stobutzki et al., 2001b; Brewer et al., 2006; Section 5.5.2; and Section 5.5.4). The diversity and abundance of this bycatch has necessitated an innovative new approach to assessing the sustainability of all species in a cost-effective and robust manner.

Firstly, a quantitative ecological risk assessment method was developed, (called a Sustainability Assessment for Fishing Effects – SAFE) and used to identify individual species at potential risk from fishing in the NPF (Section 5.4). This method supersedes previous qualitative and semi-quantitative attribute-based ecological risk assessment methods developed for data-limited bycatch species (Milton, 2001; Stobutzki et al. 2001a; 2002; Walker, 2004; Astles et al., 2006; Hobday et al., 2006). These methods provide only a relative ranking of risk amongst the suite of species in question and give little indication of a species' true risk of becoming unsustainable due to fishing. Furthermore, some are not sensitive to changes in the catchability of species as a result of changes in fisheries management strategies, and inadequately reflect even the most obvious changes in risk to individual species (Griffiths et al., 2006). However, the new SAFE method estimates the fishing mortality rate of individual species from the relative abundance of animals caught in fished, relative to unfished areas, and then assessing sustainability using a simple biological reference point (μ_{crash}) based on basic life history parameters. This allows the bycatch monitoring program to focus on individual species of concern, greatly reducing the cost and difficulty of demonstrating sustainability for all species or arbitrary ecological 'functional' groups.

Secondly, a comparison of different monitoring methods was made in order to develop the most cost-effective bycatch monitoring strategy for the broad range of bycatch species caught in the NPF. Five methods were compared using a range of criteria to assess their reliability, accuracy, feasibility and acceptability. The methods compared were fishery logbooks, requested industry collections, crew-member observers, scientific observers, and fishery-independent surveys. A range of power calculations were also made to determine the levels of sampling required to adequately detect levels of change for different species in the monitoring program. These assessments are described in Section 5.2 and provide the basis for the cost scenarios and recommendations (Section 6) for the NPF long-term bycatch monitoring program.

The use of observers in the fishery during the project provided opportunities to improve our knowledge of NPF bycatch in the banana prawn sub-fisheries. Section 5.5 provides the first comprehensive descriptions of the bycatch of both the Gulf of Carpentaria and Joseph Bonaparte Gulf sub-fisheries. This information will play an important part in the management of impacts on NPF bycatch by providing new information for risk assessments, since the new quantitative SAFE model is spatially explicit, baseline data for future comparisons, and a greatly improved knowledge of the interaction between fishing and these marine communities.

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This project has been a strongly collaborative effort between CSIRO, AFMA and the fishing industry. CSIRO and AFMA have jointly conducted observer training workshops, made ports visits, attended fishery observer conferences, and along with stakeholders from industry and conservation, formed the projects steering committee. During the course of the project, AFMA has taken an increasing responsibility for running the crew-member observer and scientific observer programs, in particular. A plan to transfer ownership, momentum and responsibility of the long-term bycatch monitoring program to AFMA and NORMAC, including ongoing collaboration with CSIRO and industry, is described in Section 5.3.

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5.2 Objective 1 – To design, trial and implement an integrated long-term bycatch monitoring program; that addresses: (i) total amount of bycatch; (ii) protected species; and (iii) high risk species in the most cost-effective manner possible using the NPF as an example

5.2.1 Selecting an effective sampling regime for monitoring diverse trawl bycatch

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Abstract

Demonstrating sustainability for tropical trawl bycatch, like that of the Northern Prawn Fishery, has been hampered by the challenges of dealing with large numbers of species at once. Traditional approaches to assessing their population sustainability would be relatively complex and expensive, especially as the species involved are usually much more poorly understood than target species. In this study we used a new approach to cost-effectively demonstrate sustainability for all bycatch species, by firstly using a new, innovative risk assessment (Section 5.4) to focus the monitoring program on species ‘at risk’ as well as other species of concern only – the Threatened, Endangered and Protected (TEP); and secondly by comparing different monitoring methods to find the most cost effective combination that can collect reliable, accurate and acceptable data for the wide variety of bycatch species involved.

We compared five methods – logbooks, requested industry collections, crew-member observers, scientific observers and fishery-independent surveys – over two years and four seasons. Logbook data from 23,470 vessel days was reliable and comparable with previous studies for sea turtles, but not for other groups, and has the advantage of collecting the large amounts of data required to detect changes in rarer species. Requested industry collections returned bycatch sub-samples from 370 trawls, but these had a high and unpredictable rate of bias due to being collected from the hopper on more than one-third of occasions. Crew-member observers participated in training workshops each year and collected data from 5,633 trawls. They demonstrated an ability to collect reliable and accurate for sea turtles, sea snakes, sawfish and other selected elasmobranchs. Scientific observers collected reliable and accurate bycatch data from 148 trawls for all targeted species groups. Fishery-independent surveys also collected reliable and accurate bycatch data from 493 trawls during this study. The bycatch monitoring program will be able to combine the high sampling power of the fishery-dependent methods to collect monitoring data for most species groups (sea turtles, sea snakes, syngnathids, sawfish, ‘at-risk’ elasmobranchs and fish), with the higher acceptance of fishery-independent methods to provide additional data and validation.

Introduction

There is considerable concern that global fisheries are having a serious, long-term impact on marine biological communities. In most cases, this concern has stemmed from assessments of target species being fished at unsustainable levels (FAO, 1999). However, there is increasing concern for species caught as bycatch which are usually poorly understood in terms of both their ecology and sustainability under impacts of fishing. Most concern is for threatened and endangered species such as albatross caught in longline tuna fisheries (Melvin et al., 2004), sea turtles caught in demersal trawls (Poiner and Harris, 1996), sea lions caught in midwater and demersal trawl fisheries (Fritz and Ferrero, 2002) and cetaceans caught in tuna purse seine operations (Bordino et al., 2002). These scenarios have resulted in targeted monitoring programs, development of threat abatement plans and implementation of bycatch reduction

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devices to ensure that fishery impacts on these species are limited to levels that allow population recovery or sustainability.

However, many fisheries impact a more diverse suite of bycatch species than these more simple scenarios, and in most cases, without knowing whether their populations can sustain the impact of the fishery or combination of fisheries in question. For example, trawl fishing is relatively non-selective and can catch hundreds of unwanted and poorly understood species that are mostly killed before being discarded at sea (Hill and Wassenberg, 1990, 2000; Wassenberg and Hill, 1989; Machias et al., 2001). Most of these fisheries have little or no effective, demonstrable bycatch management actions in place despite the possibility of species loss and the public's desire for sustainable development (Aslin and Byron, 2003).

The intention of most governments is to achieve sustainable development by creating policies and laws that promote and require fisheries to ensure sustainability of impacted species and ecosystems. This is reflected in the FAO Code of Conduct for responsible Fisheries (FAO, 2003), in fishery certification processes and in specific legislation of many countries. For example, Marine Stewardship Council certification requires a fishery to be conducted in a manner that does not lead to over-fishing or depletion, and allows for the maintenance of the structure, productivity, function and diversity of the ecosystem. This label provides fisheries with a 'clean green' marketing advantage in many countries. Australia's *Environment Protection and Biodiversity Conservation Act 1999* requires fisheries to demonstrate the sustainability of all impacted species and habitats. The United States has banned imports of prawns (shrimps) from countries not using Turtle Excluder Devices as effectively as in the USA (Hall and Mainprize, 2005). Despite these recent initiatives, few fisheries with diverse bycatch have implemented programs that provide a robust demonstration of sustainability for all populations impacted.

Demonstrating sustainability for a diverse suite of species is usually hampered by the challenges of dealing with large numbers of species at once. Data collection and analyses are relatively complex and expensive and the species involved are usually much more poorly understood compared to the target species. Consequently, there are few fisheries with diverse bycatch that collect adequate information on these species, and even fewer that are able to demonstrate that all impacted populations are fished at sustainable levels. Instead, some fisheries use more cost effective, but less precise techniques to manage impacts on bycatch. These include implementation of Turtle Excluder Devices (Brewer et al., 2006), Seal Excluder Devices (Gibson and Isakssen, 1998), streamer lines to reduce seabird catches (Melvin et al., 2004) or bycatch reduction devices that are used to exclude a wide range of unwanted species from catches (Broadhurst, 2000; Brewer et al., 2006; Courtney et al., 2006). Other fisheries have limits on the total allowable take of either specific species (e.g. short-tailed albatross and Pacific salmon in Alaskan fisheries; Hall and Mainprize, 2005) or for the total amount of bycatch discarded (Diamond, 2005), and use these limits to close the fishing season, providing strong incentive to reduce catches of these species.

However, demonstrating that populations of species impacted by fishing are sustainable requires species-specific and quantitative approaches; in particular, quantitative risk or stock assessments, or long term monitoring programs. Development of a cost effective approach to demonstrating sustainability for a diverse suite of species is essential in order for many fisheries to move towards a world's best practice approach to management, as specified in much of the guidelines and policy currently guiding them.

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The diverse nature of Northern Prawn Fishery (NPF) bycatch – including several groups that are Threatened, Endangered and Protected (Stobutzki et al., 2001; Brewer et al., 2006), does not lend itself to an obvious single most effective method for monitoring all species. Some of the more vulnerable species, such as sea turtles and sawfish are easily recognised and identified, lending themselves to onboard processing and data collection. However, others such as syngnathids are small and difficult to identify and require different sampling and processing procedures. The main aim of this study was to compare a range of bycatch monitoring methods – fishery logbooks, requested industry collections, crew-member observers, scientific observers and fishery-independent surveys – and assess the most effective method(s) to include in a long-term bycatch monitoring program for complex, diverse bycatch. This research found that a combination of fishery logbooks, crew-member observers, scientific observers and fishery-independent surveys, in association with a robust quantitative risk assessment process provides the most cost-effective strategy for assessing the sustainability of bycatch species in Australia's NPF.

Methods

General approach

An important underlying premise in this study was that all species are equally valuable and demonstrating sustainability should not be limited only to those of particular charisma, notoriety or concern. This required a method that could adequately assess all species groups.

NPF bycatch includes six species of sea turtles, more than twelve species of protected sea snakes, about 50 species of sharks, rays and sawfish, and hundreds of species of teleost fish and epibenthic invertebrates (Stobutzki et al., 2001; Brewer et al., 2006; Section 5.5.2; and Section 5.5.4). The diversity and abundance of this bycatch has required an innovative new approach to cost-effectively demonstrating sustainability for all bycatch species, by focusing the monitoring program on species of concern only. These are species listed as Threatened, Endangered and Protected (TEP) and other species assessed to be 'at risk'. The following approach allows the demonstration of sustainability for all species in a cost-effective and robust manner.

Firstly, a quantitative ecological risk assessment method was developed, (called Sustainability Assessment for Fishing Effects – SAFE) and used to identify species at potential risk from fishing in the NPF (Section 5.4). This method supersedes previous qualitative and semi-quantitative attribute-based ecological risk assessment methods developed for data-limited bycatch species (Milton, 2001; Stobutzki et al., 2001a; 2002; Walker, 2004; Astles et al., 2006; Hobday et al., 2006). These methods provide only a relative ranking of risk amongst the suite of species in question and give little indication of a species' true risk of becoming unsustainable due to fishing. In contrast, the new SAFE method estimates the fishing mortality rate of individual species from the relative abundance of animals caught in fished, relative to unfished areas, and then assesses sustainability using a simple biological reference point (μ_{crash}) based on basic life history parameters (Section 5.4). This allows the bycatch monitoring program to focus on individual species of concern, greatly reducing the cost and difficulty of demonstrating sustainability for all species or arbitrary ecological 'functional' groups.

Secondly, a program has been developed to cost-effectively monitor this broad range of bycatch species, by using a combination of sampling methods. Five methods were compared using five different criteria. These methods were; fishery logbooks, requested industry

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collections, crew-member observers, scientific observers, and fishery-independent surveys. The assessment of these methods and design of the monitoring program is described below.

Data collection

Vessels fishing in both the banana and tiger prawn fishing seasons catch significant quantities of diverse bycatch, and methods for monitoring were compared during both seasons over two years: tigers 2003, bananas 2004, tigers 2004 and bananas 2005. After the first year (end of bananas 2004) the data collection was examined and altered, where necessary to improve the design. The five methods used are described below.

Fishery Logbooks: Completion of fishery logbooks ('logbooks' hereafter) is required by all vessels in the fishery and they include data on the number and species of sea turtles caught, and the total number of sea snakes and syngnathids caught (Table 1). Logbook data are routinely collected by the fishery manager – the Australian Fisheries Management Authority (AFMA) – and the logbook bycatch data from 2003 and 2004 were used for this study.

Requested Industry Collections (RICs): This method involved the voluntarily collection of bycatch samples and data by fishing skippers and crews, following written and face-to-face recruitment into the program by project staff. Instructions and sampling gear were given to fishers in ports before the 2003 and 2004 NPF fishing seasons. No other formal training was involved. RICs were requested to collect, store (frozen) and return subsamples of the 'small bycatch' (mixed bycatch excluding large animals >300 mm, Heales et al., 2003) during selected weeks, following phone calls from project staff. Along with these bycatch subsamples, they were also asked to record physical trawl data (date, time, location, depth), gear specifications, and estimate the total bycatch (bycatch + prawn) from the sampled net.

Crew Member Observers (CMOs): These were volunteer fishing crew who were recruited either by phone or during pre or post-season port visits. Where possible, a skipper and crew member from the same vessel were involved in the process. Thirteen and twelve CMOs attended two-day training courses during the NPF mid-season breaks in July 2003 and 2004, respectively. They were trained on data collection, selected species identification and other aspects of the bycatch monitoring program. CMOs were asked to record estimates of the total bycatch weight with validation where possible; collect subsamples of bycatch during specific times of the season; and record data on all sea turtles, sea snakes, sawfish and other selected sharks and rays caught (Table 1). Sea turtles were identified to species, lengths measured, and then released. Sea snakes were placed in a prawn basket with a 10 cm scale bar, photographed using a disposable camera and released. Photographed sea snakes were later identified to species (where possible) and their lengths estimated by specialist scientists in the laboratory. Sawfish and other selected sharks and rays (five species in total) were identified to species by the CMOs and their lengths either measured (if possible), or estimated. Other physical trawl data and gear specifications (see RIC method) were recorded.

Scientific Observers: These were trained specialists who boarded vessels throughout the fishing season with a primary objective of collecting a range of bycatch data. This included estimates of total bycatch weight with validation, the identification (photographed where possible) and life status of sea turtles, sea snakes, sawfish and selected sharks and rays (Table 1). Other physical trawl data and gear specifications (see RIC method) were recorded.

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Fishery-Independent Surveys: This method collected the same data as scientific observers (Table 1); however, using a pre-planned random, stratified sample design as used during the associated prawn monitoring cruises (Dichmont et al 2003). This contrasts with the other fishery-dependent methods which used sampling patterns designed to maximise prawn catches. These are typically much more targeted and clumped than a random design and determined by the fishing skipper from day to day. The fishery-independent surveys were conducted using chartered NPF fishing vessels and crew in order to be equivalent to any future fishery-independent monitoring program. Fishing sites were allocated in most of the major high effort fishing regions in the Gulf of Carpentaria (Figure 4.). Data from adjacent (deeper) regions was collected to provide baseline data on species composition from potential control regions to aid in the interpretation of any changes in bycatch during any future monitoring program.

RICs, CMOs and scientific observers were contacted regularly during the fishing seasons to promote targeted sampling windows, remind them of their requested tasks and assist them with any difficulties where possible.

Assessment criteria

Five criteria were used to compare and assess the bycatch collection methods.

Feasibility: Determining what data each method could collect was based on previous experience and consultation with fishers, observers and scientists. It was influenced mainly by what was reasonably achievable for each sampling method. Bycatch categories sampled by each method are described above and summarised in Table 1.

Data accuracy: Data accuracy was assessed using a range of measures (Table 2). RICs, CMOs and scientific observers returned subsamples of the bycatch used for assessing species composition and relative abundances of selected species to the laboratory. Subsample collection of diverse trawl bycatch is subject to different forms of bias, especially when seawater hoppers are used to help sort the catch (Heales et al., 2003). Subsamples from a well mixed catch are essential for comparing relative species abundances. Fishers and observers were asked to collect subsamples from a well mixed part of the catch before spilling it into the seawater hopper, and then later asked how often subsamples were taken incorrectly (directly from the hopper). CMOs were asked to identify selected species of sharks and rays, and then return them to the laboratory for validation. Accuracy was difficult to assess for logbook and RIC data.

Data reliability: Data reliability is broadly defined as how often tasks were completed compared to the requested or expected outcomes. It was also assessed using a range of others measures (Table 2). The number of sea turtles, sea snakes and syngnathids recorded in logbooks was compared to expected numbers from previous bycatch sampling (Brewer et al., 2006). The number of subsamples returned by RICs, CMOs and scientific observers was also compared to requested levels, as well counting the proportion of subsamples with adequate labels enclosed. The frequency of data recorded in key data fields for all methods was also assessed, as well as the number of sea snake photographs taken by CMOs and scientific observers that could be easily identified to species and their length estimated.

Stakeholder acceptance: This criterion was based on the acceptance of each method by a range of stakeholders including fishers, other industry representatives, conservation group representatives, industry management and the scientific community. It was measured quantitatively by our ability to recruit RICs and CMOs into the program, the number of

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fishers that participated compared to the number that were approached, and the number that collected adequate data compared to the number that agreed to participate. More qualitative information was received from other stakeholders from project steering committee meetings, feedback from project presentations and informal discussions.

Relative financial cost: Accurate costings were not considered as they varied appreciably throughout the period of the study. In general, the financial cost of bycatch data collection follows this pattern: logbooks < RICs < CMOs < scientific observers < fishery-independent surveys. This is based on the following major costs: logbooks (none); RICs (sampling materials); CMOs (sampling materials and training); scientific observers (sampling materials, training and salary); fishery-independent surveys (vessel charter, sampling materials and salaries). All methods require an approximately equal cost for data entry, processing, analyses and reporting. Cost was only considered when the data collection could be effectively achieved by more than one method, following assessment of the feasibility, accuracy, reliability and acceptance of methods (Section 5.2.6).

Data processing and analyses

Logbook and CMO data were returned to AFMA, while RIC, scientific observer and fishery-independent survey data were returned to CSIRO. Logbook data was entered into the existing AFMA database. Most logbook data used in this report came from AFMA data summaries compiled for annual and mid-season reports. AFMA developed a database and data entry forms for the CMO data, and provided CSIRO with data summaries in formats suitable for analyses. CSIRO developed a database and data entry forms for the RIC data, the scientific observer and the fishery-independent survey data.

CSIRO also developed a database and data entry forms for animals that were identified to species at CSIRO. These included photos of sea snakes and some sawfish returned from CMOs, scientific observers and fishery-independent surveys. CSIRO also developed a database and data entry forms to assess the success of the sampling for the small bycatch species. CSIRO matched the animals identified from photos, as well as small bycatch information to the AFMA CMO database. Data were checked by CSIRO for errors and summarised for comparative analyses.

Power calculations were completed in order to determine sample sizes necessary to detect change for different species during fishery-dependent data collection (Sections 5.2.2, 5.2.3), and to compare fishery-dependent with fishery-independent sampling patterns (Section 5.2.3). The information from these studies was used to help compare methods and included in the discussion, below.

Results

Logbooks

Logbook data was collated from a total of 23,470 vessel days during this study: 8,718 days (97 vessels), 3,419 days (95 vessels), 8,336 days (95 vessels) and 2997 days (85 vessels) during tigers 2003, bananas 2004, tigers 2004 and bananas 2005, respectively. This provided data from the highest number of trawls of any of the methods and returned the required bycatch data from all trawls. Fishing gear logs were returned on 78% of occasions. The spatial distribution of logbook data is shown in Figure 1.

However, logbooks only provide data for three bycatch categories and species-specific information for sea turtles (Table 1 and Table 2). A total of 51 sea turtles were recorded in

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logbooks during the study, an average catch rate of 0.00065/trawl. This catch rate appears low compared to data collected by CMOs (0.005/trawl, Table 2). CMOs were the only other method to collect comparable numbers of sea turtles, as the introduction of TEDs has greatly reduced their catch rates (Brewer et al., 2006). The life status was recorded for 98% of sea turtles from logbooks and 96% were recorded as alive when returned to the sea.

Logbooks were also required to record sea snake data but not for individual species. A total 22,283 sea snakes were recorded in logbooks (0.29 per trawl) during the study. Catch rates of total sea snakes were about half (55%) of those recorded by CMOs and less than half (40%) of the catch rate reported in Brewer et al. (2006) (Table 2). The life status of sea snakes was not required in NPF logbooks prior to 2004. Life status was recorded for 80.07% of sea snakes where requested (bananas 2004, tigers 2004 and bananas 2005) and 84.9% were alive when returned to the sea.

Logbooks also required data on all syngnathids encountered but not for individual species. Only 50 syngnathids were recorded in logbooks (0.00064 per trawl) during the study (Table 2). Catch rates were also low for this species group compared to other methods although sample sizes for scientific observers and fishery-independent surveys were small. Life status of syngnathids was not required in NPF logbooks prior to 2004. Life status was recorded for 56.25% of syngnathids where requested (bananas 2004, tigers 2004 and bananas 2005) and 22.2% were alive when returned to the sea.

Crew member observers

Crew member observers (CMOs) collected data from a total of 5,633 trawls during the study: 3,322 trawls (13 CMOs), 230 trawls (2 CMOs), 2,081 trawls (11 CMOs) and 0 trawls (0 CMOs) during tigers 2003, bananas 2004, tigers 2004 and bananas 2005, respectively. The spatial distribution of CMO samples is shown in Figure 2. They collected information for most bycatch groups with varying degrees of success. CMOs collected data from most days (81.7%) in the fishing season, but usually only from one of two nets (52% of all trawls and about 50% for sea turtles, sea snakes and sawfish) (Table 1).

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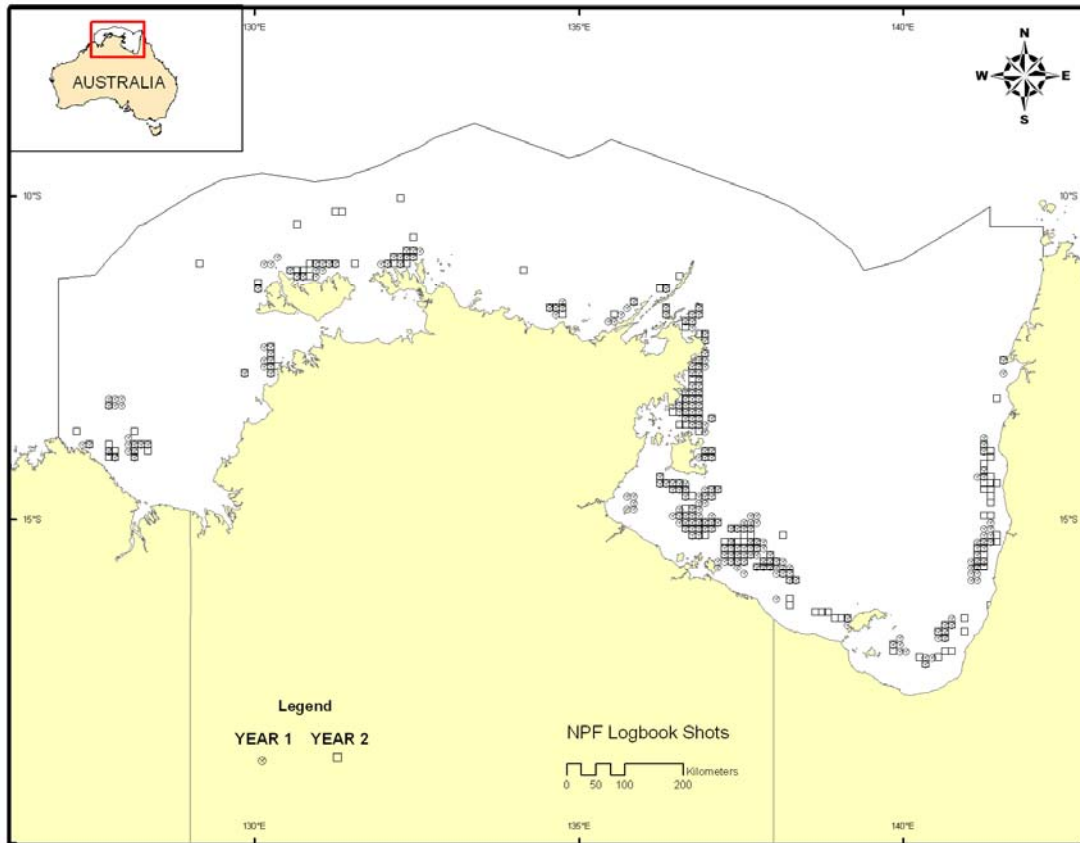


Figure 1. Grids where fishing was recorded in logbooks in the tiger fishery 2003 and banana fishery 2004 (year 1) and spring 2004 and autumn 2005 (year 2). To meet the confidentiality requirements, points shown are restricted to total number of boats >4 and total number of days fished >9)

CMO's recorded a total of 26 sea turtles during this study (0.005/trawl) – a much higher catch rate than surveyed in 2001 by Brewer et al. (2006) (0.0006/trawl), and about one quarter the number recorded by scientific observers (0.021/trawl) although only three were recorded in this program (Table 2). The catch rates of sea turtles are very low compared to those before the introduction of Turtle Excluder Devices (TEDs) in 2000 (e.g. 0.051, recorded in Poiner and Harris, 1996). A large number of sea turtles were not identified to species by CMOs (50%). These were usually sea turtles caught in the net before exclusion through the TED, but not brought on board where they could be identified. The life status was recorded for 77% of sea turtles and 90% were alive when returned to the sea.

CMOs recorded a total of 2,523 sea snakes during the study or 0.47 (± 0.012) per trawl. This catch rate is 27.1% lower than that of the 2001 survey of Brewer et al. (2006) and 41.4% lower than the earlier work of Wassenberg et al. (1994). Most sea snakes (96.3%) were photographed by CMOs and species identification and length estimates were possible from 98.4% of these. The life status was recorded for 76% of sea snakes and 76.3% were alive when returned to the sea.

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Table 1. Summary of assessment criteria used to compare the feasibility, accuracy, and reliability of five data collection methods for NPF bycatch. Ten data categories were used for the comparison, including six species groups. # = data collected at the requested day and correct number of time(s); \$ = bridge logs and sample labelling recorded correctly; @ = approximate data based on data loss due to gear failure, and other sample processing and collection errors; iu = identifications unvalidated; s04 = assessed in the 2004 tigers fishery only; tc = total counts only required.

Criteria	Logbooks	RICs	CMOs	SOs	FIS
1. Feasibility of data collection					
Sea turtles	yes	yes	yes	yes	yes
Sea snakes	no (tc)	no	yes	yes	yes
Syngnathids	no (tc)	no	no	yes	yes
Sawfish	no	no	yes	yes	yes
Other elasmobranchs	no	no	yes	yes	yes
Total bycatch estimates	no	yes	yes	yes	yes
Subsample collection	no	yes	yes	yes	yes
2. Accuracy					
i. Identification to species					
Sea turtles (iu)	94%	na	27%	66.7%	~99%
Sea snakes	na	na	98.5% (n=2,484)	100%	94% (iu)
Syngnathids	na	na	na	50%	50%
Sawfish	na	na	61% (n=94)	100% (iu)	93% (iu)
Other elasmobranchs	na	na	77.0% (n=35)	100%	100%
ii. Bycatch weight estimates validated	na	na	27% ^{s04}	100%	100%
iii. Samples collected without bias	na	30-70%	10-75%	>95% [@]	100%
3. Reliability					
Recruitment success	na	77.0%	15%	na	na
'Days/nights' sampled	100%	59.5%	81.7%	100%	na
Trawls sampled	na	59.5%	52.0%	100%	na
<i>a. Number recorded – see Table 2</i>					
<i>b. Nets checked</i>					
Sea turtles	100%	na	49.4%	100%	100%
Sea snakes	100%	na	50.4%	100%	100%
Syngnathids	100%	na	na	100%	100%

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Table 1 continued

Criteria	Logbooks	RICs	CMOs	SOs	FIS
Sawfish	na		49.4%	100%	100%
Other elasmobranchs	na	na	17.2%	100%	100%
Small bycatch species	na	na	na	>95%	100%
<i>c. Life status recorded (% alive in brackets)</i>					
Sea turtles	98% (96%)	na	77% (90%)	100% (66.7%)	-
Sea snakes	na	na	76% (76.3%)	88% (55%)	100% (97.8%)
Syngnathids	na	na	na	na	na
Sawfish	na	na	66% (26%)	93.1% (40.7%)	100% (98%)
<i>d. Number sea snakes photographed</i>	na	na	96.3% (n=2611)	90%	94%
<i>e. Total bycatch weight estimates made</i>	na	87%	98%	100%	na
<i>f. Subsample collection returns*</i>	na	59.5%	42.0%	100%	100%
<i>g. Subsamples collected at correct time[#]</i>	na	100%	100%	100%	100%
<i>h. Subsample data recorded correctly[§]</i>	na	74%	68%	>95% [@]	99% [@]
<i>i. Gear logs returned correctly</i>	78%	75.0%	50.0%	100%	100%

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Table 2. Comparison of catches (numbers caught per trawl ('tr')) for species groups between the four different methods and previous studies. Percentage species composition within each group is also presented. The number of individuals is in brackets; na = not applicable, either because this data was not collected, or different gear was used (as indicated). * = only data from trawls with TEDs and BRDs installed.

	Logbooks	CMOs	Scientific observers	Fishery independent surveys	Brewer et al., (2004)	Poiner and Harris (1996)	Wassenberg et al. (1994)
Sea turtles							
Total sea turtles	0.00065/tr (51)	0.005/tr (26)	0.021/tr (3)	0	0.0006/tr (1)*	na (no TEDs) (326)	
Flatback (<i>Natator depressus</i>)	38.3% (18)	30.8% (4)	-	-	29.9% (14)	63.2% (192)	
Loggerhead (<i>Caretta caretta</i>)	4.2% (2)	23.1% (3)	-	-	8.5% (4)	10.5% (32)	
Olive Ridley (<i>Lepidochelys olivacea</i>)	19.1% (9)	30.8% (4)	-	-	57.4% (27)	13.2% (40)	
Green (<i>Chelonia mydas</i>)	34.0% (16)	0	100% (1)	-	4.2% (2)	8.2% (25)	
Hawksbill (<i>Eretmochelys imbricata</i>)	4.2% (2)	15.3% (2)	-	-	0	4.9% (15)	
Unidentified sea turtles	4	13	2	-	37	22	
Sea snakes							
Total sea snakes	0.29/tr (22283)	0.470/tr (2,523)	0.365/tr (54)	0.18/tr (89)	0.644/tr (774)		~1.134 (1,247)
<i>Hydrophis elegans</i>	na	30.9% (767)	21.3% (10)	35.7% (30)	37.4% (304)		16.3% (207)
<i>Disteira major</i>	na	19.8% (492)	8.5% (4)	14.3% (12)	15.3% (124)		2.8% (35)
<i>Lapemis hardwickii</i>	na	14.9% (371)	57.5% (27)	26.2% (22)	17.1% (139)		53.5% (681)
<i>Astrotia stokesii</i>	na	9.8% (243)	0	4.8% (4)	7.5% (61)		4.6% (60)
<i>Aipysurus. laevis</i>	na	7.2% (180)	0	2.4% (2)	6.2% (50)		1.6% (20)
<i>Aipysurus eydouxii</i>	na	6.2% (155)	0	4.8% (4)	2.8% (23)		6.0% (76)
<i>Hydrophis ornatus</i>	na	4.5% (112)	6.4% (3)	3.6% (3)	6.5% (53)		0.9% (12)
<i>Hydrophis pacificus</i>	na	3.8% (94)	0	2.4% (2)	3.4% (28)		0
<i>Acalyptophis peronii</i>	na	2.1% (52)	0	3.6% (3)	2.2% (18)		1.7% (22)
<i>Disteira kingii</i>	na	0.48% (12)	2.1% (1)	0	0.5% (4)		2.6% (34)
<i>Hydrophis mcdowelli</i>	na	0.24% (6)	0	0	0.1% (1)		0.7% (9)
<i>Enhydrina schistosa</i>	na	0	2.1% (1)	0	0		8.1% (104)

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Table 2 continued

	Logbooks	CMOs	Scientific observers	Fishery independent surveys	Brewer et al., (2004)	Poiner and Harris (1996)	Wassenberg et al. (1994)
<i>Hydrophis caeruleus</i>	na	0	2.1% (1)	0	0		0.6% (8)
<i>Aipysurus duboisii</i>	na	0	0	2.4% (2)	1.0% (8)		0.4% (6)
Syngnathids	0.00064/tr (50)	na	0.0135/tr (2)	0.0101/tr (4)			
Sawfish							
Total sawfish	na	0.0298/tr (155)	0.196/tr (29)	0.0284 (14)	0.0157/tr (13)		
Narrow sawfish (<i>Anoxypristis cuspidata</i>)	na	97% (91)	97% (28)	100% (13)	100% (4)		
Dwarf sawfish (<i>Pristis clavata</i>)	na	2% (2)	0	0	0		
Green sawfish (<i>Pristis zijsron</i>)	na	1% (1)	3% (1)	0	0		
Unidentified sawfish	na	61	0	1	9		

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CMOs recorded 155 sawfish or 0.0298 (± 0.0027) per trawl, almost twice the catch rate surveyed in 2001 (Brewer et al., 2006; Table 2) but less than one-fifth of the catch rate recorded by scientific observers (Table 2). CMOs identified 94 (61%) sawfish to species as validated from photographs, with 61 remaining unidentified. This large number of unidentified were also mainly due to their being caught in the net before exclusion through the TED, but not brought on board where they could be identified. The life status was recorded for 66% of sawfish and 26% were alive when returned to the sea.

CMOs checked the nets for other selected sharks and rays on 17.2% of occasions and recorded data on 45 of these animals. They identified 35 (77%) of these to species as validated from labelled specimens sent back to the laboratory. Life status was not recorded for these species.

CMO's collected subsamples of small bycatch from 42.0% of trawls requested. All were collected at the time requested and 68% had correct written data and label information (Table 1). They estimated the weight of the total bycatch for 98% of trawls requested and validated these estimates by weighing the catch on 27% of the occasions that it was requested. However, a relatively high proportion of subsamples (10-75%) were collected in a biased way, directly from the seawater hopper, instead of taking the subsample from a part of the catch before it was spilled into the hopper.

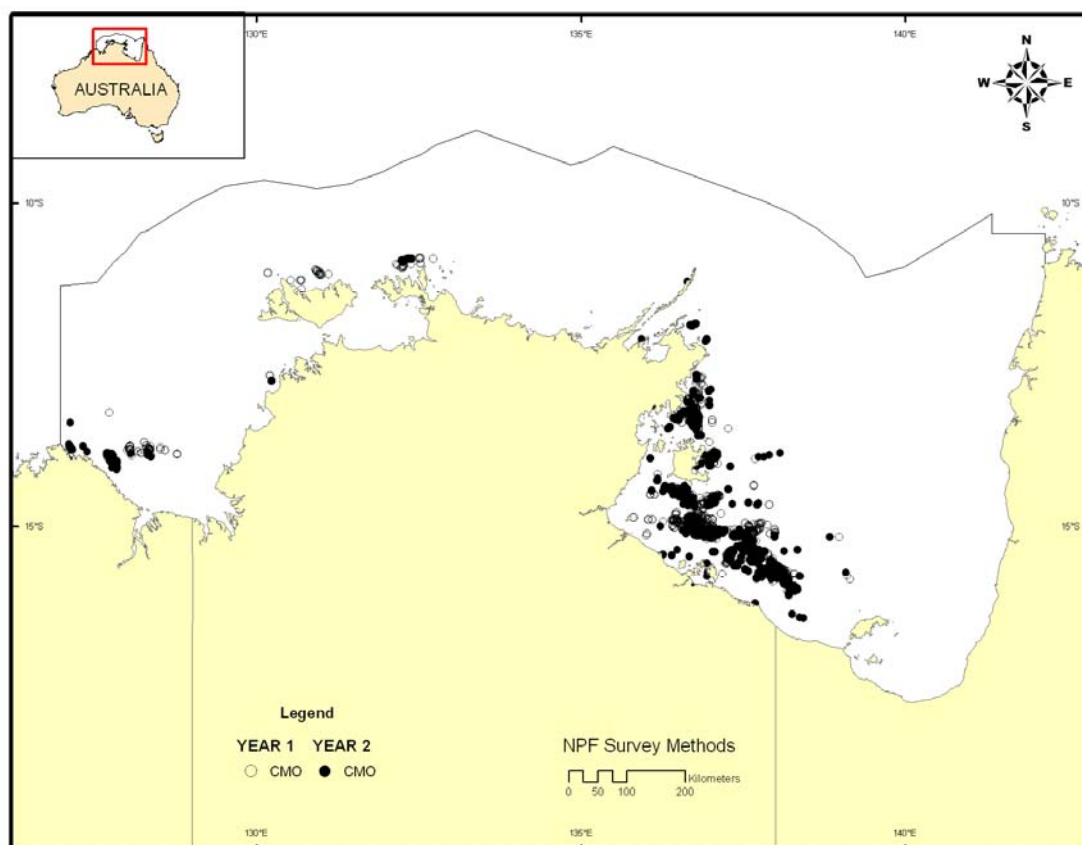


Figure 2. Sampling sites used during the crew-member observer (CMO) sampling in tigers 2003 and bananas 2004 (year 1) and spring 2004 and autumn 2005 (year 2) to collect bycatch data for comparison with other methods.

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Requested Industry Collections

Requested industry collections (RICs) collected data from a total of 307 trawls during the study: 170 trawls (32 vessels), 29 trawls (11 vessels), 91 trawls (22 vessels) and 17 trawls (24 vessels) during tigers 2003, bananas 2004, tigers 2004 and bananas 2005, respectively. The spatial distribution of logbook data is shown in Figure 3. They were not trained in, or requested to deal with new species groups but collected subsamples of small bycatch from 59.5% of trawls requested (Table 1). All of these were collected at the time requested and most (74%) recorded the data correctly on data sheets and subsample labels. Fishing gear data sheets were completed correctly on 75% of occasions. RICs estimated the weight of the total bycatch for 87% of trawls requested. However, a relatively high proportion of subsamples (~50%) contained bias in their species composition due to their collection directly from the seawater hopper (Heales et al., 2003), RICs were requested to avoid this by taking the subsample from a part of the catch before it was spilled into the hopper.

Scientific observers

Scientific observers collected data from a total of 148 trawls during the study: 27 trawls (two observers, one vessel), 70 trawls (two observers, two vessels), 0 trawls (0 observers) and 51 trawls (two observers, two vessels) during tigers 2003, bananas 2004, tigers 2004 and bananas 2005, respectively. The spatial distribution of scientific observer samples is shown in Figure 3. They collected information for all bycatch groups with a very high degree of reliability and accuracy. Scientific observers collected data from almost all days they were at sea in the fishing season, and always from both nets (100% of trawls for all bycatch groups) (Table 1).

Scientific observers recorded a total of three sea turtles during this study or 0.021 per trawl (± 0.0116) (total of one *Chelonia mydas* – green turtle; one *Natator depressus* – flatback turtle and one unidentified to species) – a much higher catch rate than the other methods and those surveyed in 2001 by Brewer et al. (2006) (0.0006/trawl). However, the sample size is too low to make valid comparisons. One was not identified to species as it was caught in the net before the TED, and not brought onboard. The flatback juvenile was caught in ‘try gear’ and was dead. The life status was recorded for all three sea turtles and two of these were alive when returned to the sea.

Scientific observers recorded a total of 54 sea snakes during the study or 0.365 (± 0.055) per trawl. This catch rate is lower than the CMO data and the previous studies of Brewer et al. (2004) and Wassenberg et al. (1994). However, the scientific observers collected much of their data in the banana trawl fisheries of the Joseph Bonaparte Gulf and the Gulf of Carpentaria (Section 5.5). All sea snakes encountered were photographed, identified to species and lengths estimated. The life status was recorded for 88% of sea snakes and 51% were alive when returned to the sea (10 snakes were retained under Department of the Environment and Heritage Permit Number: M2003/0009).

Scientific observers recorded 29 sawfish or 0.196 (± 0.0367) per trawl, a larger catch rate than either CMOs or Brewer et al. (2004) (Table 2). All (100%) were identified to species on board, based on a high degree of expert knowledge. Differences in sawfish catch rates between methods (scientific observers, CMO’s and fishery-independent surveys) were probably due mainly to differences in sawfish distribution between regions as well as the increased variability where there are lower sample sizes. The unidentified individuals were caught in the net and unable to be clearly viewed. The life status was recorded for 93.1% of sawfish and 40.7% were alive when returned to the sea.

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Scientific observers checked the nets for other selected sharks and rays on all occasions and recorded data on 400 of these animals (19 species). They identified 100% of these to species based on expert knowledge. Life status was not recorded for these species.

Scientific observers recorded two Syngnathids or $0.0135 (\pm 0.0135)$ per trawl. One (50%) was identified to species. Life status was not recorded for these species.

Scientific observers collected subsamples of small bycatch from 100% of trawls requested. All were collected at the time requested and 100% had correct written data and label information (Table 1). They estimated the weight of the total bycatch for 100% of trawls requested and validated these estimates by weighing the catch on 100% of the occasions requested. All subsamples were collected in an unbiased way.

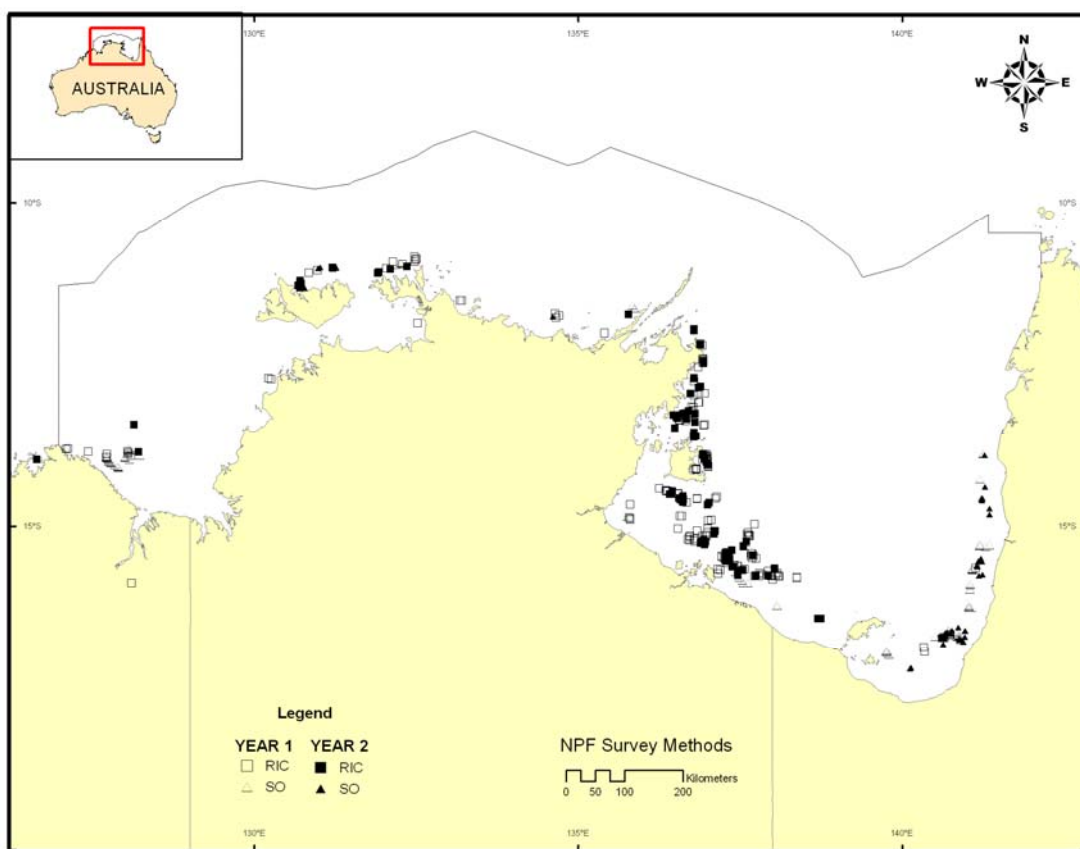


Figure 3. Sampling sites used during the requested industry collections (RICs) and scientific observer sampling in tigers 2003 and bananas 2004 (year 1) and tigers 2004 and bananas 2005 (year 2) to collect bycatch data for comparison with other methods.

Fishery-independent surveys

Fishery-independent surveys collected data from 493, 30 minute trawls during the study: 160 trawls (two observers, two vessels), and 333 trawls (two observers, two vessels, during pre-tigers 2002, and pre-bananas 2003, respectively). The spatial distribution of samples is shown in Figure 4. Fishery-independent surveys were dedicated to collecting scientific data and as such were completely reliable in collecting all required data from trawls (Table 1). They were also manned by highly specialised scientific staff and were accurate with all forms of sample processing.

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Fishery-independent surveys recorded interactions with sea turtles during this study. However, the sample size is too low to make valid comparisons.

Fishery-independent surveys recorded a total of recorded 89 sea snakes or 0.18 (± 0.022) per trawl. This catch rate is lower than the other methods and the previous studies of Brewer et al. (2004) and Wassenberg et al. (1994). Ninety-four percent of the sea snakes encountered were photographed, identified to species and lengths estimated. The life status was recorded for 100% of sea snakes and 98% were alive when returned to the sea.

Fishery-independent surveys recorded 14 sawfish or 0.0284 (± 0.008) per trawl, a similar catch rate to the CMOs, and a much higher rate than reported by Brewer et al. (2004) (Table 2). Thirteen (93%) were identified to species on board, based on expert knowledge. The life status was recorded for 100% of sawfish and 98% were alive when returned to the sea.

Fishery-independent surveys recorded data on other selected sharks and rays for 2194 of these animals (13 species). They identified all to species based on expert knowledge. Life status was not recorded for these species.

Fishery-independent surveys recorded four syngnathids or 0.0101 (± 0.00452) per trawl. Two (50%) were identified to species. Life status was not recorded for these species.

Fishery-independent surveys collected subsamples of small bycatch from 100% of trawls requested. The weight of the total bycatch was measured (not estimated) for 100% of trawls. All subsamples were collected in an unbiased way.

Differences in species composition

The species composition of sea turtles collected by logbooks and CMOs was different from the 2001 survey by Brewer et al. (2004) and the much larger data set described by Poiner and Harris (1996) (Table 2). Logbooks had a high relative number of Green turtles while CMOs had relatively high numbers of Loggerhead and Hawksbill turtles. However, both methods caught relatively few of these animals due to the introduction of TEDs in 2000, making robust comparisons difficult. The accuracy of the species identification in logbooks and by CMOs was not validated.

The species composition of sea snakes collected and photographed by CMOs and fishery-independent surveys are quite similar to the 2001 survey described by Brewer et al. (2004); with *Hydrophis elegans* comprising about one third of all sea snakes caught, while *Lapemis hardwickii* and *Disteira major* comprised the second and third highest abundances of all sea snakes caught. Most of the remaining 11 species also showed comparative contributions. However, there are some significant differences between these results and the species composition collected by scientific observers and Wassenberg et al. (1994), where *L. hardwickii* dominated catches and *D. major* was not one of the most abundant five species. These latter studies had a data collection bias towards the shallow banana prawn fishing grounds. In contrast, the CMOs, fishery-independent surveys and the survey by Brewer et al. (2004) were collected during the tiger prawn sub-fishery in deeper waters.

The species composition of sawfish by CMOs, scientific observers and Brewer et al. (2004) was almost identical, with *Anoxypristis cuspidata* comprising the overwhelming majority of sawfish encountered (Table 2).

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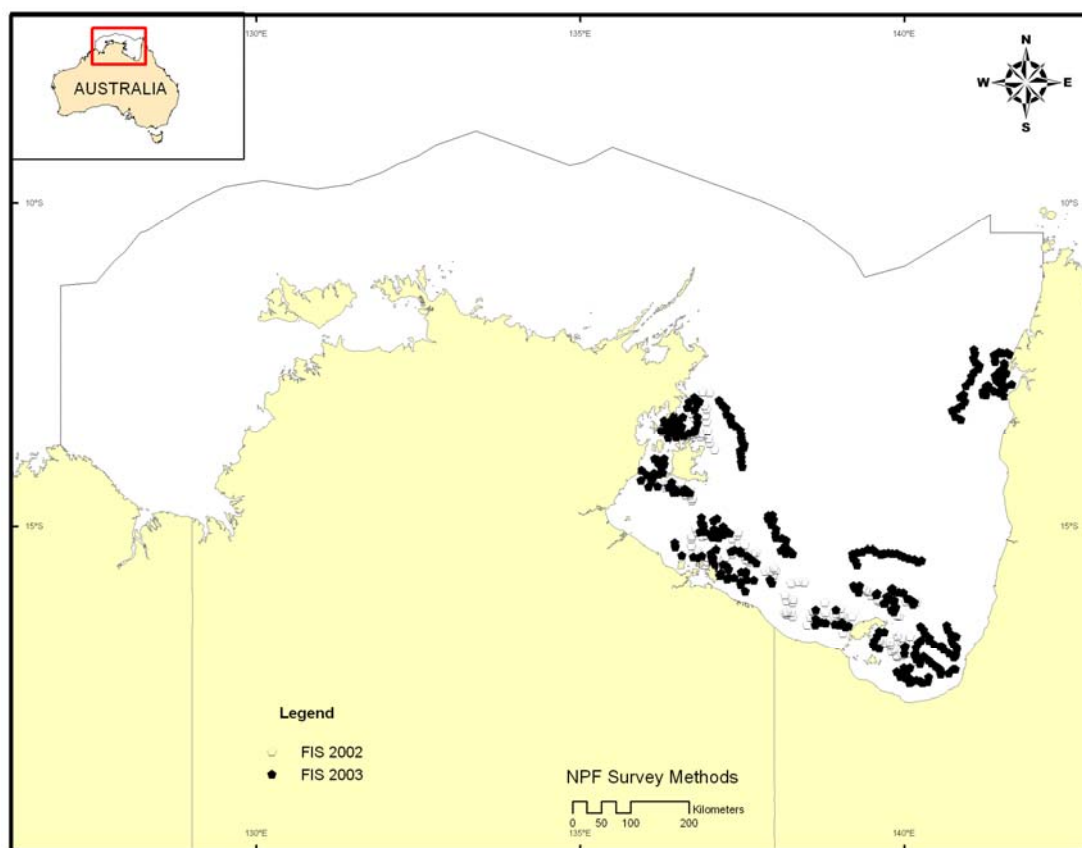


Figure 4. Sampling sites used during the fishery-independent surveys in 2002 and 2003 to sample bycatch species and communities for comparison with fishery-dependent methods.

Stakeholder acceptance

Discussions with fishery stakeholders during project steering committee meetings, industry committee meetings and specific discussions have shown a universally high level of trust associated with the data collected from fishery-independent methods (scientific observers and fishery-independent surveys) compared to fishery-dependent means. This is likely to be based on past experience in the fishery whereby logbooks, in particular, have gained a poor reputation for recording certain types of information; with bycatch data having a generally low priority with most skippers and crew.

More acceptable uses of fishery-dependent data collection include where the data can be validated by fishery-independent sources, or where a higher level of training and enthusiasm for data collection exists (e.g. CMOs).

Discussion

The comparison of methods for sampling bycatch has demonstrated a variety of strengths and weaknesses in each. Consequently, the most cost effective bycatch monitoring program is likely to be a combination of methods that takes advantage of the higher sample sizes that logbooks and CMOs, in particular are capable of, with the more accurate and acceptable methods that use highly trained and fishery-independent data collection.

Logbooks have the highest data collection ability, but only record data for three species groups (sea turtles, sea snakes and syngnathids). All of these recorded lower catch rates in

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logbooks than the other methods. However, sea turtles were comparable with one study (Brewer et al., 2006) and the comparison of sea turtle catch rates may be invalidated somewhat by the low sample sizes for most of the methods. The sea snake and syngnathid data from logbooks is not species specific and consequently of no use in assessing risk for individual species.

Fishers participating in the RIC method were not trained in any species-specific identification (other than their sea turtle logbook identification guides) and could only feasibly collect subsamples of the small bycatch. However, the use of hoppers on most vessels affected the ability of these collections to reflect the true species composition for both RIC and CMO subsample collections (Heales et al., 2003).

CMOs demonstrated an ability to collect data on a wide variety of bycatch groups and this was relatively reliable and accurate for sea turtles, sea snakes, sawfish and other selected elasmobranchs. These groups could be included in a bycatch monitoring program if validated by scientific observer data.

Scientific observers were dedicated to collecting bycatch data and collected reliable and accurate information for all targeted species groups, although higher numbers of observers are required to collect adequate data on the rarer species groups such as sea turtles, syngnathids and selected sharks and rays (Section 5.2.3). The small bycatch subsamples collected by observers were used to provide the first comprehensive descriptions of bycatch composition for both the Joseph Bonaparte Gulf red-legged banana prawn (*Penaeus indicus*) and the Gulf of Carpentaria common banana prawn (*P. merguensis*) sub-fisheries.

The fishery-independent surveys also collected reliable and accurate bycatch data for all targeted species groups, but also had a lower capacity to collect adequate data on the rarer species groups. The survival rates for sea snakes and sawfish were higher in fishery-independent surveys but most likely due to their shorter tow times (30 mins) compared to the commercial length tows (3-4 h) for the other methods. The collection of small bycatch subsamples was based around a random stratified design and hence more likely to provide the most accurate description the bycatch communities in different regions. This design is also able to better control for variability due to spatial, diel and lunar periodicity known to affect marine and bycatch species populations (King 1995).

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5.2.2 Detecting declines in of diverse trawl bycatch species, and implications for monitoring

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Abstract

Trawl fisheries around the world are being pressured to demonstrate that their impacts on both the target and bycatch species are sustainable in the long term. However, the effectiveness of sampling catches to assess the viability of populations of non-targeted species is rarely examined. We estimated the levels of fishery-dependent sampling effort required to detect declines in catch rates of prawn trawl bycatch from 95 commercial trawls in two regions of Australia's Northern Prawn Fishery. We explore a range of possible monitoring options including combining different sampling intensities, time-frames and levels of statistical power.

Poisson and negative binomial models were used to determine the number of trawls required to detect a range of declines. We found that between 15,536 and 24,933 trawls, depending on the region, would be required to detect a 20% decline in catch rates of the rarest taxa (<0.1 individuals h^{-1}), assuming a power of 90% and a level of significance of 5%. Assuming a lower detection power (70%), trawl numbers would drop to 9,126 and 14,646 respectively.

Using a model of a constant decline in relative abundance (over five years), data accumulated from modest-sized annual surveys (52 and 43 trawls in two regions) would provide increasing power to detect changes in catch rates. After three years, surveys of this size (and power of 70%), could detect declines of 99.9% in 72 – 81% of taxa, declines of 50% in 34 – 43% of taxa, and declines of 20% in 20 – 34% of taxa, depending on the region. After five years, the power to detect declines of 50% had increased to cover 43 – 72% of taxa, and declines of 20% in 34 – 43% of taxa. Our results indicate that the power to detect even quite large declines in catch rates of rarely caught species would only be possible after some years of modest sized annual surveys.

Keywords: Trawl bycatch; Monitoring; Power analysis; Sustainability; Prawn trawling; Shrimp trawling

Introduction

Managers of trawl fisheries worldwide are under increasing pressure to demonstrate that populations of both target species and bycatch are sustainable (FAO 1995). A variety of sampling methods (both fishery-dependent and fishery-independent) are used to assess the population status of the target species and demonstrate their sustainability. Fishery-dependent methods of measuring catches commonly combine data from commercial logbooks or landings with data collected by onboard observers. Combinations of these methods are used in fisheries like the North Pacific and Bering Sea groundfish trawl fishery, the Northwest Atlantic trawl fishery, and the Southeastern United States shrimp otter trawl fishery, Australia's South East trawl and Northern Prawn Fisheries (eg. McElderry *et al.* 1999, Knuckey and Liggins 1999, Dichmont *et al.* 2003).

However, the management of bycatch species is far more problematic and costly. This is mainly because bycatch is often not recorded in logbook information and the catches are mostly discarded at sea; the life histories of many bycatch species are unknown or poorly defined, making the estimation of sustainable levels of impact a difficult practice; and the

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distribution of bycatch species overlaps that of the target species but the proportion of each population impacted by fishing effort is unknown. As bycatch populations are not targeted by fishers (and in some cases, actively avoided), the catch rates are likely to be more variable than the targeted species' rates, with costly implications for the sampling effort required to detect catch rate declines. The high variability in sampling is further complicated by the lack of knowledge of how bycatch species react to diel and lunar cycles (King 1995).

Consequently, fishery-dependent methods that assess populations of target species (eg. CPUE-based, mark-recapture etc) are not necessarily directly applicable to the bycatch, particularly in fisheries with a very diverse bycatch (e.g. tropical prawn trawl fisheries, e.g. Stobutzki *et al.* 2001).

The accuracy and precision of fishery-dependent methods to assess populations of target species have been closely studied (Fox and Starr, 1996; Maunder and Punt, 2004). These methods rely on field catch rates being closely related to the population biomass. However, their ability to detect changes in bycatch populations has not been documented. For example, the bycatch of the US Gulf of Mexico prawn trawl fishery is monitored using fishery-dependent observer data (Nance *et al.* 1997), but the accuracy and precision of the monitoring have not been reported. In some countries, legislation requires that fishery managers demonstrate that their fishery is sustainable, both for target and bycatch species (e.g. in Australia, see Fletcher *et al.* 2005). However, few trawl fisheries even in such countries are currently collecting fishery-dependent or fishery-independent data that could be used to estimate the fishery impact on bycatch, other than a few threatened, endangered or protected species. Furthermore, no trawl fishery has reported on the performance of monitoring programs aimed at detecting changes in bycatch populations in diverse tropical prawn trawl fisheries catching hundreds of small bycatch species.

Establishing a fishery-dependent bycatch monitoring program would require careful design of sampling methods, including the level of sampling effort and precision needed to detect significant changes in catch rates. The level of effort needed for accurate monitoring would depend on the effect size to be detected, as well as between-trawl variability of sampling for individual species.

To assess the feasibility of monitoring population declines in bycatch by fishery-dependent methods, we examined data sets from surveys made by an observer on commercial trawlers in two regions of Australia's NPF. Based on these data, we calculated the level of sampling effort required to detect small (20%), medium (50%) and large (99.9%) declines in the catch rates of bycatch species. To counteract the likely high sampling effort required for monitoring some species, we examined the effects of extending time frames, and reducing the levels of detection power from 90% to 70%. These results are discussed in the context of their implications for monitoring.

Materials and methods

Field sampling

Between 25 May and 7 October 1997, a scientific observer collected subsamples of bycatch from 95 catches of two twin-rigged commercial prawn trawlers operating in the Northern Prawn Fishery (Figure 5), in both the North of Mornington Island region (52 trawls) and the North of Groote Eylandt region (43 trawls). Each trawler towed nets of 14 fathom headrope length, at a speed of around 3.2 knots.

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At the completion of each trawl and just before the codends were spilled, the total weight of the catch from one of the two nets was estimated using an electronic load cell. Both vessels were fitted with conventional above-deck sorting trays which separate the catches from each codend. After the codends were spilled, a single subsample (about 25kg) was collected from the unsorted catch of the weighed codend. After target species were removed, about 20kg of bycatch usually remained in the subsample. A total bycatch estimate was calculated as the total catch weight minus the target species weight for the sampled net, and then used to raise the subsample to the total catch weight. All bycatch samples were frozen on board and taken to the laboratory for processing. Individual fish and invertebrates from each subsample were identified, counted and weighed, and the data were entered into Oracle database tables.

Both trawlers completed four trawls each night during the sampling periods, with the trawl duration ranging from 2.00 to 3.75 h. The first trawl of the night usually included a half hour of twilight and the last trawl of the night included up to two hours of dawn or full daylight. Little is known about diel and lunar changes in the detectability of bycatch species in trawl catches, and particularly during the night–day transition (Salini *et al.* 2001). However, we have included all trawls in the analysis to fully represent the nature of commercial fishing practice.

Data description

For the purposes of this study, we treated the existing data collected over two weeks from each region, as if it represented the baseline survey data for future bycatch monitoring programs. In order to examine how well the trawls represented the bycatch species likely to be caught in the respective regions, we plotted species–area curves of the cumulative number of species recorded as increasing biomass of trawl subsamples were sorted.

There are many sources of both within and between–trawl variability in the catch rates of bycatch species. Within–trawl variability is greatly affected by subsampling techniques on-deck, usually caused by taking too small a subsample (Heales *et al.* 2000, 2003b). Between–trawl variability is affected by differences in species detectability between trawls, caused by combinations of patchiness in distribution of bycatch species; their behaviour related to feeding, reproduction, schooling and avoidance of predators; diel and lunar cycles; and differences in trawl speed, net sizes and weather conditions. All these sources of variability will be reflected in the efficiency of sampling individual species and are typical of most monitoring scenarios for non–target trawl species.

Abundance categories

Sample–size calculations were based on the mean numbers of species falling into five abundance categories that were determined from the survey data. The different abundance categories were based on the mean number per hour for each species i , caught by trawls in each region. This is represented by \bar{A}_i in Equation 1.

$$\bar{A}_i = \frac{\sum_{j=1}^{n_T} M_{ij} F_j}{n_T} \quad (1)$$

where M_{ij} represents the number of animals classed into species i and caught at the j -th trawl and n_T represents the total number of trawls carried out in the region. In Equation 1, F_j represents a scaling factor for the j -th trawl and it is comprised of the ratio between the total catch weight and the subsampled catch weight at trawl j (Q_j / W_j) divided by the duration of the trawl, D_j as shown in Equation 2

$$F_j = \frac{Q_j}{W_j} \times \frac{1}{D_j} \quad (2)$$

where Q_j represents the total catch weight (kg) and W_j represents the total weight (kg) of the subsample obtained from the j -th trawl.

Abundance categories were then formed by allocating each species \bar{A}_i into one of the following categories: very rare (≤ 0.1 animal per hour), rare ($0.1-1 \text{ h}^{-1}$), common ($1-10 \text{ h}^{-1}$), abundant ($10-100 \text{ h}^{-1}$), and very abundant ($>100 \text{ h}^{-1}$). We use the terms 'very rare' and 'rare', not in the traditional ecological context of the word, but as indicators of degrees of species occurrence in the surveyed trawl catches.

Calculating the number of trawls

In this section we outline the method by which the sample sizes to detect change for each abundance category were calculated. Although fishery managers are interested in both the direction and magnitude of change, preliminary analysis showed that detecting increases in catch rates would require lower levels of sampling effort than detecting declines (per same percentage). Consequently, we have presented the more precautionary approach of calculating the effort levels required to detect declines.

We outline the method by which the sample sizes for each abundance category were calculated. We adopt the null hypothesis (H_0) of “no change” in favour of the alternative hypothesis (H_a) that the relative abundance of a bycatch species has declined by a certain effect size, d , where the effect size is measured as a percentage. This is a standard approach used in prospective fishery studies, where the aim is to determine the size and direction of a change (Peterman, 1990). To achieve this, we have assumed an underlying generalized linear model and we describe the sample sizes necessary to give the likelihood ratio test of “no change” (where change represents a decline for bycatch species) for some specified power. The method is both approximate and asymptotic, but adequate to establish the minimum sampling effort required for practical planning purposes. Examples of these calculations are outlined in Cox and Hinkley (1974) and Self *et al.* (1992). We provide a summary of how these sample sizes were calculated below, details of which can be found in Appendix 1.

For bycatch species, our alternative hypothesis is that the mean count per trawl μ declines by a fixed proportion, (γ), per annum. This equates to $\gamma = 1 - d / 100$, a function of the effect size. So if $\gamma = 0.1$ then the mean declines by a proportion of 0.1 per year, resulting in a 90%

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decline or “effect size (d)” over the period. Due to the lack of studies that document the pattern of declines in catch rates of bycatch species, we chose a fixed proportion decline. However, we recognise that the patterns of declines in catch rates of bycatch species may exhibit threshold effects at some exploitation level. In what follows, i will be used to index the year, ranging from year 0 to year T . So at time 0 ($i = 0$), μ_0 and μ_a are both equal to the same mean count, λ .

$$\begin{aligned} H_0 : \mu_0 &= \lambda \\ H_a : \mu_a &= \lambda \gamma^i \quad i = 0, 1, \dots, T \end{aligned}$$

We considered using a range of suitable models including the Poisson and negative binomial models, as well as zero-inflated Poisson and zero-inflated negative binomial mixture models. Zero inflated mixture models may well be applicable to bycatch studies as the zeros in the data may be attributed to the inclusion of bycatch reduction devices, habitat and spatial location of the survey as well as competing species. However, for this study, there is no covariate information available that can describe the cause of the zero inflation. The negative binomial distribution is also highly appropriate for data that is zero inflated (Warton, 2005), having the capacity to account for excess zeros (as well as large counts) through the over dispersion parameter, ϕ . There are numerous examples of Poisson and negative binomial models being used in practice to estimate catch rates of fish and other species (See Smith and Richardson (1977), Jahn and Smith (1987), Cyr *et al.* (1992), Power and Moser (1999)). Consequently, we considered that these models provided an adequate framework for determining the minimum sample size requirements for each bycatch group in this study.

The Poisson model has density as shown in Equation 3, a distribution typically chosen for count data.

$$f(y; \mu) = \frac{\mu^y}{y!} \exp(-\mu), \quad y = 0, 1, \dots \quad (3)$$

In this Equation, y represents counts of a bycatch species and μ represents the corresponding mean count. The mean–variance relationship is such that the mean and variance of the Poisson distribution are equal.

$$E[Y] = \text{Var}[Y] = \mu \quad (4)$$

The negative binomial model has density as shown in Equation. 5.

$$f(y; \mu, \phi) = \frac{\Gamma(\phi + y)}{\Gamma(\phi)\Gamma(y + 1)} \left(\frac{\phi}{\mu + \phi} \right)^\phi \left(\frac{\mu}{\mu + \phi} \right)^y, \quad y = 0, 1, \dots \quad (5)$$

This distribution contains an additional over dispersion parameter, ϕ , which allows the variance of the distribution to exceed the mean. The Poisson is the limiting case of the negative binomial as the over dispersion parameter increases to infinity. Using this parameterisation of the negative binomial distribution shown in Equation 5, the mean and variance, respectively, are now

$$E[Y] = \mu, \quad \text{Var}[Y] = \mu + \mu^2/\phi \quad (6)$$

The smaller the estimate of ϕ , the more over dispersed the distribution is relative to the Poisson.

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We use the methods described by Self *et al.* (1992) to provide some insight into the number of trawls required through Equations for n_p (calculation based on Poisson model in Equation (3)) and n_{NB} (calculation based on negative binomial model in Equation (5)) for a given level of significance α , power $1 - \beta$, effect size d , corresponding proportion γ and number of years, T that monitoring takes place. These equations, which are developed in Appendix 1, are outlined below and can be coded into a statistical programming language such as R (Ihaka and Gentleman 1996)

$$n_p = \eta(\alpha, \beta) / \frac{1}{T+1} \sum_{i=0}^T 2 \{ \lambda \gamma^i \log \gamma^i - \lambda \gamma^i + \lambda \} \quad (7)$$

$$n_{NB} = \eta(\alpha, \beta) / \frac{1}{T+1} \sum_{i=0}^T 2 \left\{ \lambda \gamma^i \log(\gamma^i) + \log \left(\frac{\lambda + \phi}{\lambda \gamma^i + \phi} \right) (\phi + \lambda \gamma^i) \right\} \quad (8)$$

In both of these Equations, $\eta(\alpha, \beta)$ represents the non-centrality parameter, which for the non-central chi-square distribution with 1 degree of freedom, is represented as $\eta(\alpha, \beta) = (z_{1-\alpha/2} + z_{1-\beta})^2$. In this expression, $z_{1-\alpha/2}$ and $z_{1-\beta}$ represent the upper α and β critical values of the standard Normal (z) distribution as outlined in Rochon (1989).

Analyses

We used a power analysis approach to investigate the number of samples, n , required in each abundance category and in each region (North of Mornington Island and North of Groote Eylandt) to detect an effect size d corresponding to a specific level of power $1 - \beta$ and level of significance α . Here, an effect size d , represents a percentage of decline in species, which equates to a fixed proportion of decline γ , occurring per annum (equal to $1-d/100$) in Equations 5 and 6. Models chosen for each abundance category were based on the shape of the distribution of the numbers of individuals per trawl in each abundance category and corresponding summary statistics. In particular, we used the mean-variance relationship defined in Equation 6 to determine the value for the over dispersion parameter, ϕ . Where over dispersion was evident, (n_{NB}), Equation 8 was used to calculate the number of trawls required to detect a specific level of decline; otherwise the Poisson model ($\phi = \infty$) was assumed and the corresponding number (n_p) was evaluated (Equation 7).

We explored a range of effect sizes (d) at intervals of 10%, (eg, 10%, 20%, 30%) but for ease of reporting, we have chosen to only present effect sizes of 20%, 50% and 99.9% (effectively 100% but dividing by zero makes the calculation for 100% decline impossible). We investigated two levels of power, 70% and 90%. All scenarios were examined at the 5% level of significance. The non-centrality parameters corresponding to a significance level of 0.05 and powers of ($1 - \beta = 0.7$ and $1 - \beta = 0.9$) are 6.1721 and 10.5074, respectively. Reducing the power to detect change from 90% to 70% will increase the probability of a Type 2 error – that a bycatch species has actually suffered a real decline that we are unable to detect. This strategy should result in a reduction in the numbers of trawls needed to be sampled, hence a reduced annual cost of future monitoring plans. We investigated sample sizes after one, two, three, four and five years of hypothetical annual surveys based on 52 trawls at North of Mornington Island and 43 trawls at North of Groote Eylandt. The results are expressed as the numbers of trawls of average trawl duration that need to be sampled to detect given declines.

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However, for ease of reporting, we present only the one, three and five year options in this study.

Results

General patterns in the baseline data

The approximate areas of the regions sampled were 2096 km² at North of Mornington Island (NMI) and 1948 km² at North of Groote Eylandt (NGE) (Figure 5a, b). The distribution of trawl sites at NMI was much more even than at NGE, where the distribution was heavily biased towards the north–east corner of the grid; 30 (69.8%) of the 43 trawls were contained in one small area of about 99.5 km² (Figure 5b). These distribution patterns are typical of many fishery effort patterns.

In the NMI region, a total of 52 trawls representing 154.10 h of trawling provided the dataset on which the sample size calculations were based (Table 3). The total weight of bycatch taken from the sampled trawl (one of the two nets only) was 17,713 kg, for an average weight of 340.6 ± 18.5 kg and an average trawl duration of 3.00 h. The total weight of bycatch subsampled was 1,188.0 kg; we subsampled $8.3 \pm 0.7\%$ of each catch weight (Table 3).

In the NGE region, a total of 43 trawls representing 137.0 h of trawling provided the dataset (Table 3). The total weight of bycatch taken by the sampled trawl (one of the two nets only), was 7,397 kg, for an average weight of 172.0 ± 15.5 kg and an average trawl duration of 3.18 h. The total weight of bycatch subsampled was 1,022.6 kg; we subsampled $18.9 \pm 1.8\%$ of each catch weight (Table 3).

A total of 266 taxa (combined teleosts and invertebrates) were recorded in all subsamples in trawls from the NMI region and 295 taxa in trawls from the NGE region (Figure 6). The total number of taxa recorded at each region continued to increase with increasing numbers of trawls sampled, with a higher rate of increase in the NMI region. The slight tendency of both curves to become asymptotic indicates that most of the taxa (>90%) that were detectable by prawn trawling at that time of year, were sampled by these surveys, but also that further sampling will increase the number of species recorded (Figure 6).

Around 57% of taxa at NMI region and 66% of taxa at NGE were recorded at catch rates of ≤ 1 h⁻¹ (Figure 7, 'rare' and 'very rare' categories). Only 20% of taxa at NMI and 11% of taxa at NGE had catch rates of > 10 h⁻¹ (Figure 7, 'abundant' and 'very abundant' categories).

General patterns of sampling effort

Irrespective of the region, the number of trawls required to detect change decreases (a) as the statistical power to detect declines decreases, (b) as successive annual surveys are amalgamated into assessments, (c) as the size of the decline (to be detected) increases, (d) and with taxa that are more abundant.

Detecting 20% declines

The effect of lowering the power to detect declines from 90% to 70% is demonstrated in Table 4 and Table 5. The number of trawls needed to detect a 20% decline in just one year in the 'very rare' abundance category is reduced by 43%, from 24,933 to 14,646 at NMI, and from 15,536 to 9,126 at NGE (Table 4 and Table 5). Similar levels of reduction are demonstrated for all abundance categories.

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Combining lowered detection power of 70% with the postponement of assessments for three years, results in further effort reductions. The number of trawls needed to detect a 20% decline after three years in the 'very rare' abundance category is reduced by 90%, from 24,933 to 2,611 at NMI, and from 15,536 to 1,627 at NGE (Table 4 and Table 5). Similar levels of reduction are demonstrated for all abundance categories.

Similarly, combining lowered detection power of 70% and postponing the assessment for five years results in further effort reductions. The number of trawls needed to detect a 20% decline after five years in the 'very rare' abundance category is reduced by 95%, from 24,933 to 1,223 at NMI, and from 15,536 to 762 at NGE (Table 4 and Table 5). Similar levels of reduction are demonstrated for all abundance categories.

Detecting 50% and 99.9% declines

In general, the effect of lowering the power from 90% to 70% results in a reduction of 41% in the number of trawls needed to detect declines of both 50% and 99.9% in the 'very rare' abundance category in just one year in both regions (Table 4 and Table 5). Similar levels of reduction are demonstrated for all abundance categories.

The combination of lowered detection power (70%) and the postponement of assessments for three years, results in a reduction of 85% and 61% in the number of trawls needed to detect declines of 50% and 99.9%, in the 'very rare' abundance category in both regions (Table 4 and Table 5). Similar levels of reduction are demonstrated for all abundance categories.

Similarly, combining lowered detection power (70%) and postponing the assessment for five years results in a reduction of 90% and 65% in the number of trawls needed to detect declines of 50% and 99.9% respectively, in the 'very rare' abundance category in both regions (Table 4 and Table 5). Similar levels of reduction are demonstrated for all abundance categories.

Power of modest-sized surveys

Assuming a detection power of 70%, annual surveys of around 52 trawls (size of baseline survey) at NMI would be able to detect a 20% decline in none of the taxa after one year, in 20% after three years, and in 43% after five years. The same size annual survey would also be able to detect a 50% decline in 20% of the taxa after one year, and in 43% after three and five years. And it could detect a 99.9% decline in 43% of the taxa after one year, and in 81% after three and five years (Table 5).

Table 3. Summary of trawl survey data used to estimate the number of trawls required to detect declines in catch rates of bycatch species in two regions of the Northern Prawn Fishery.

Region	Total trawls (n)	Total trawl hours (n)	Mean duration (h) of trawls (\pm 1SE)	Total weight of bycatch (kg) (one net only)	Mean weight (kg) of bycatch per trawl (\pm 1SE) (one net only)	Total weight (kg) of subsamples (one net only)	Mean percentage of total bycatch subsampled (\pm 1SE)
North of Mornington Island	52	154.10	2.96 (\pm 0.1)	17,713	340.6 (\pm 18.5)	1188	8.30 (\pm 0.7)
North of Groote Eylandt	43	137.00	3.18 (\pm 0.1)	7,397	172.0 (\pm 15.5)	1022.6	18.90 (\pm 1.8)

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Assuming a detection power of 70%, annual surveys of around 43 trawls (size of baseline survey) at NGTE would be able to detect a 20% decline in 1% of the taxa after one year, in 34% after three years, and in 34% after five years. The same size annual survey would also be able to detect a 50% decline in 34% of the taxa after one year and three years, and in 72% after five years. And it could detect a 99.9% decline in 34% of the taxa after one year, and in 72% after three and five years (Table 5).

Discussion

Monitoring many hundreds of bycatch species to detect significant changes in populations is difficult but increasingly necessary in modern fisheries and ecosystem management. In Australia's Northern Prawn Fishery (NPF), a typical codend from a three-hour trawl contains around 300 kg of small bycatch, comprising over a hundred species. The collection of unbiased subsamples is critical to reducing subsampling errors in trawl catches (Heales *et al.* 2000, Heales *et al.* 2003a and b). Many species can only be identified in the laboratory, so samples must be frozen or preserved and taken to land for sorting, identifying, and processing. The cost of collecting and processing data on bycatch species is very high (Stobutzki *et al.* 2001), and high numbers of trawls need to be sampled to detect changes in the rarely caught species. The ability to detect even a relatively large annual decline of 50% in catch rates of the rarest species requires sampling about 1,920 trawls; or every trawl, from 30 trawlers for around 16 nights (four trawls per night) in each region of interest. The high cost of this level of sample collection and processing may be prohibitive to many fisheries seeking a more cost-effective monitoring strategy.

However, our study highlights an alternative long-term and more cost-effective bycatch monitoring strategy: (i) undertaking modest sized annual surveys (around the same level of effort as detailed in this study), (ii) postponing of population status assessments until sufficient sampling power has been accumulated, and (iii) accepting a lower power of 70%, to detect Type 2 errors.

Accumulating annual fishery-dependent bycatch surveys, of the size used to collect our baseline data, should be affordable for many trawl fisheries. Furthermore, these results are probably conservative for the levels of power achievable by fishery-independent surveys of a similar size, where more control over variation in detectability of species in catches is available. In many fisheries, there is annual monitoring of the target species (Shelton and Lilly, 2000, Dichmont *et al.* 2004). Simultaneously combining the objectives of monitoring both target and bycatch species would make this strategy even more attractive to fishery managers.

With even modest-sized annual surveys, every year there will be a suite of species where sufficient sampling power is available to detect some gross levels of decline in catch rates. But with every subsequent annual survey, the power to detect declines will increase. Postponing the period of assessment for three or even five years provides the power to detect quite small declines in many of the more rarely caught species.

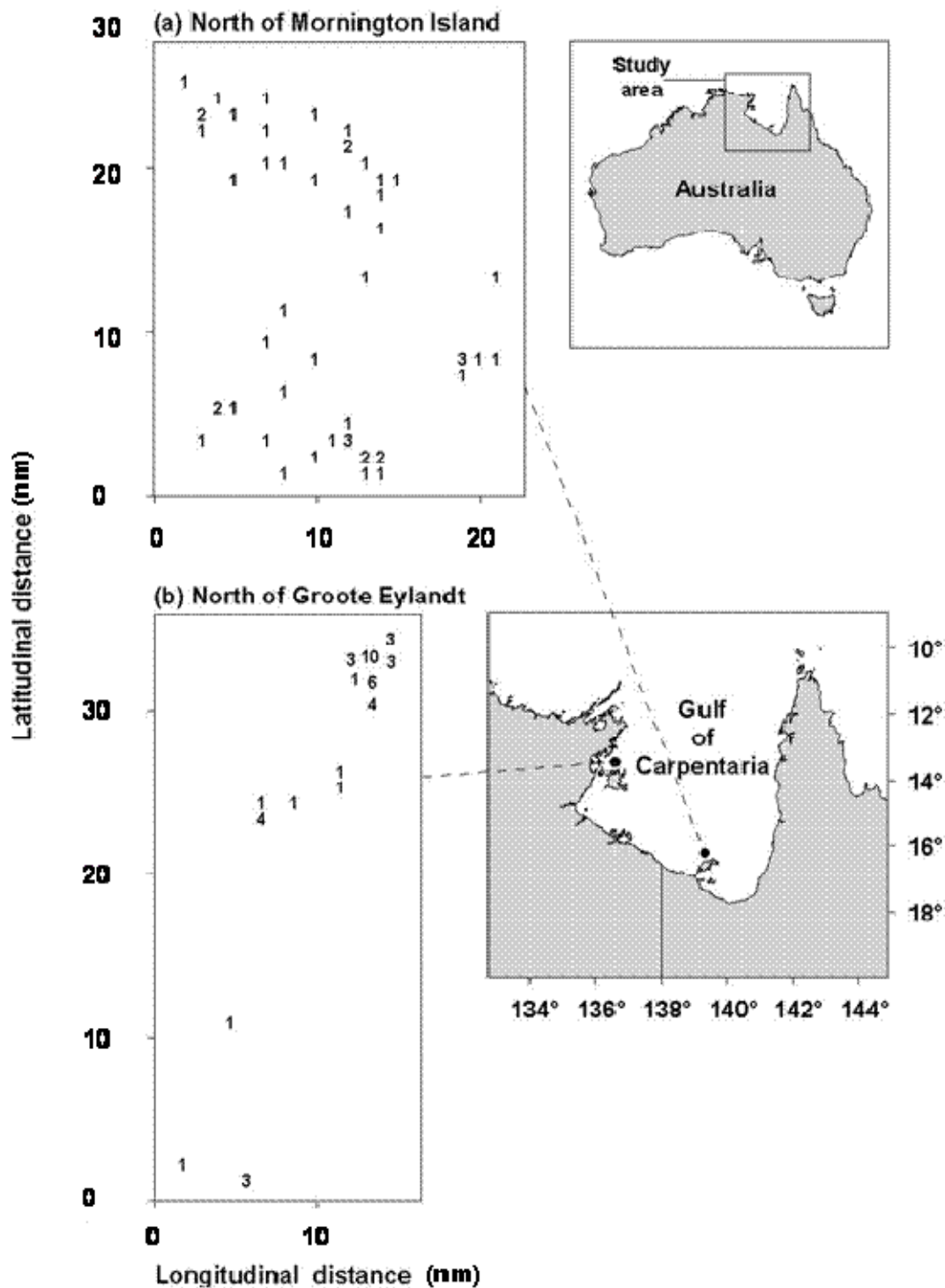


Figure 5. The distribution and numbers of trawls sampled at each site that were used to estimate the effort needed to detect declines in catch rates for bycatch species in two regions of Australia's Northern Prawn Fishery (a) North of Mornington Island (52 trawls) and (b) North of Groote Eylandt (43 trawls). The numbers on (a) and (b) refer to the number of trawls sampled at each site. Due to fishery confidentiality agreements, true latitude and longitude of sites are unable to be shown.

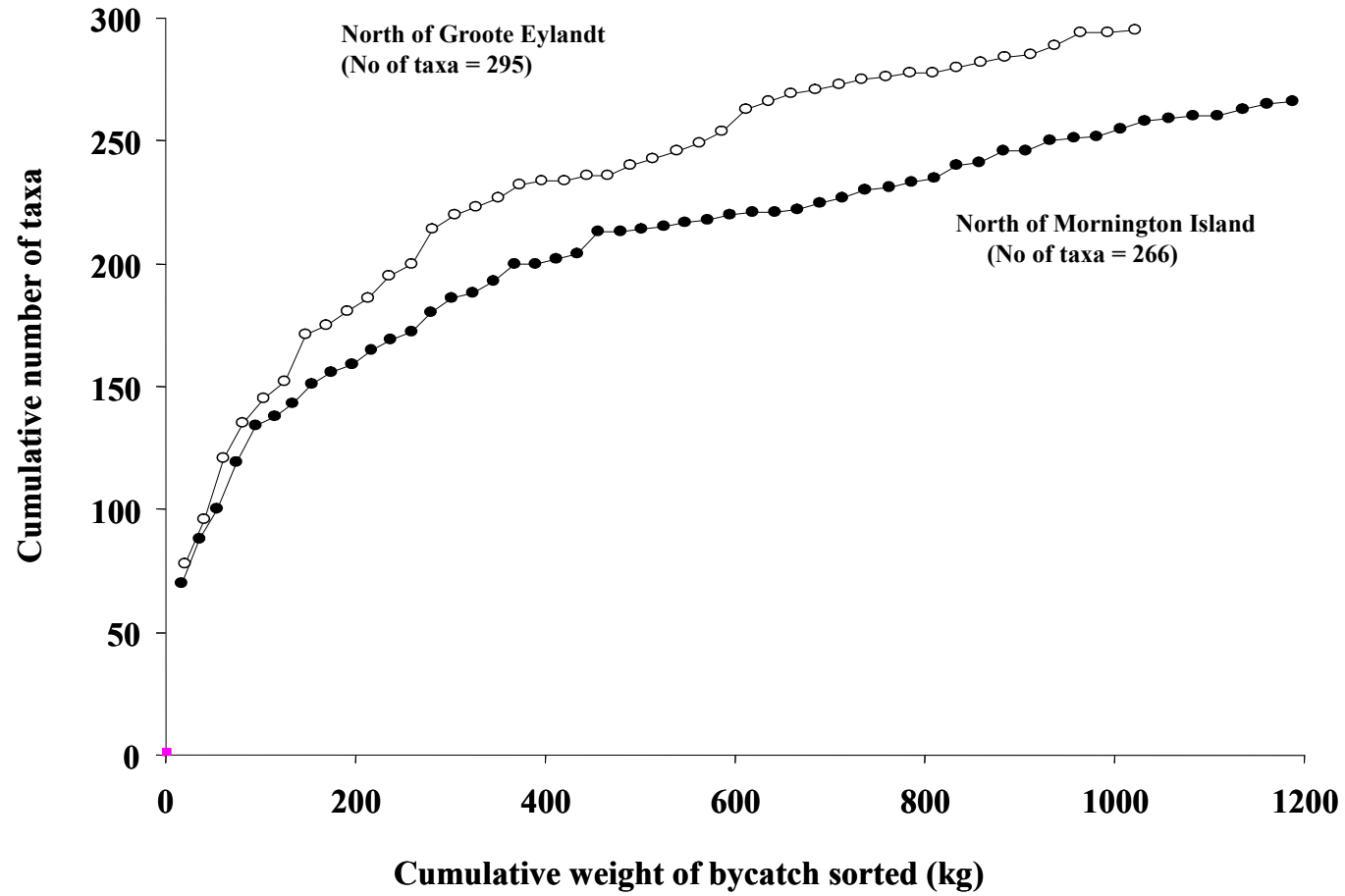


Figure 6. Species-area curves generated from bycatch samples collected by scientific observers on commercial trawlers in two regions of Australia's Northern Prawn Fishery, North of Mornington Island (52 trawls sampled) and North of Groote Eylandt (43 trawls sampled).

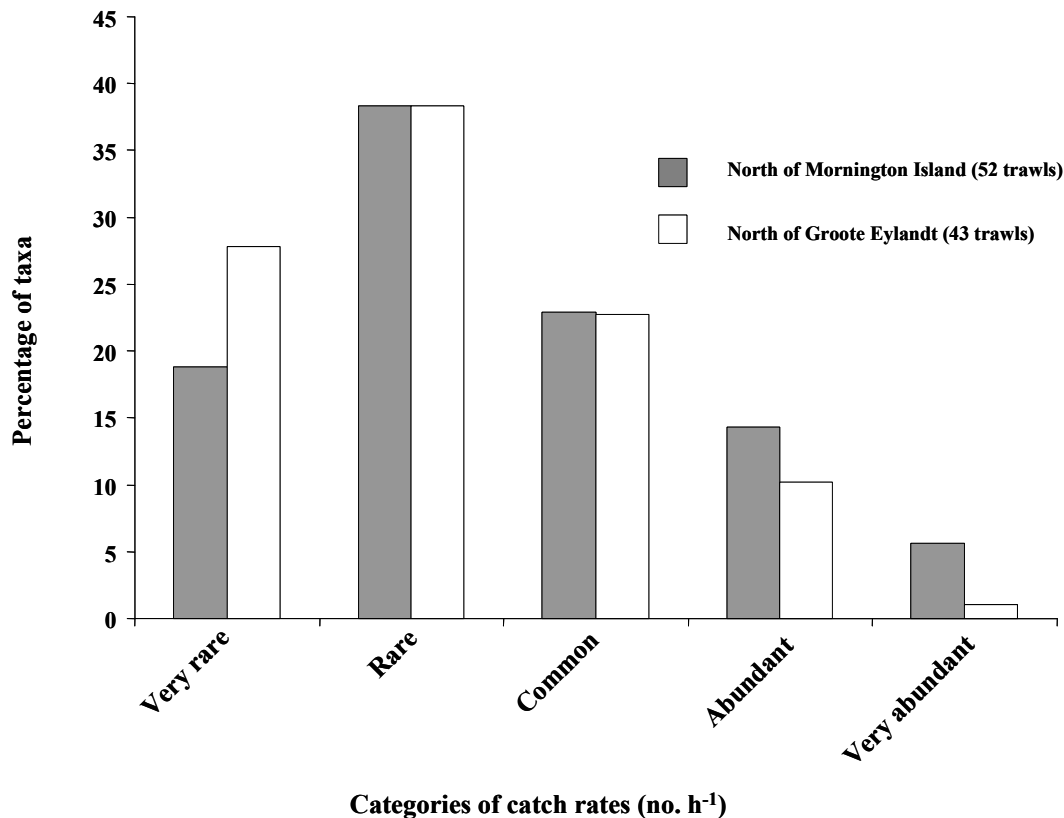


Figure 7. Histograms showing the percentage distribution of prawn trawl bycatch taxa in five categories of relative abundance from two regions of Australia's Northern Prawn Fishery, North of Mornington Island and North of Groote Eylandt

However, postponing assessments of declines, possibly for as long as five years, incorporates risk and assumes a resilience of these rarely caught species during this time. Because so little is known about the general biology and life history of many bycatch species, deciding what constitutes a significant decline for individual species is a challenge to researchers and fishery managers alike. Some bycatch species will have life cycles that are completed many times over a three or five year period of assessment. Although changes in catch rates can indicate changes in the detectability of bycatch species, the link to population change is unproven. Consequently, determining the significance of declines or increases in populations of the diverse suite of bycatch species impacted by this fishery will be a long and involved process. It will require the application of more robust quantitative risk assessments (currently being developed for this fishery) in place of previous methods that only assess relative (not absolute) risk (e.g. Stobutzki *et al.* 2001). These processes are critical to understanding the impact of fishing on bycatch species, and to defining the levels of impact that individual species can sustain.

In the NPF, with over six hundred bycatch species, and even using power of 90% every year by chance alone, some 30 or more species will show declines that require some level of closer inspection. Acceptance of a lower onus of proof (power = 70%) implies that a higher number of species with real declines will remain undetected in the annual assessment. However, with the annual increase in power available from the following year's survey, the declines in these species are more likely to be detected next year. Under these circumstances, adopting a lower onus of proof seems a reasonable and cost-effective strategy.

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If monitoring of numerous and diverse tropical bycatch species is to be undertaken using annual surveys, then strict controls of sampling methods are essential. The untargeted nature of the bycatch, combined with the unknown diel and lunar cycles of detectability, and the large on-deck errors when sampling catches (King 1995, Heales *et al.* 2000, Heales *et al.* 2003a and b), demand that regimented sampling methods be adopted, irrespective of whether the survey is fishery-dependent or independent. If fishery-independent surveys are used, then strict attention must be paid to sampling sites at the same time of night, lunar phase, season, proportion of catch sampled, as well as same trawl speed, gear configuration and rigging; in order to eliminate as many variables as possible.

However, the sustainability of rarely caught species that cannot be monitored cost-effectively, may ultimately depend on the adoption of different bycatch management strategies. For example, Marine Protected Areas can offer long-term habitat refuges for such species (Halpern, 2003). Similarly, restricting the trawl grounds to areas that are currently impacted at medium or high levels may provide refuges in the low effort areas, and is likely to increase the chances of these species' ecological sustainability. Either strategy, Marine Protected Areas or restricting trawl effort to trawl grounds already moderately or heavily impacted, should allow macrobenthos to re-establish in the low effort areas where recruitment is not limited and substrates are suitable. Research on developing less destructive trawl gear could also result in less damage to benthos (Brewer *et al.* 1996).

Conclusions

Using catch rates to monitor annual declines in bycatch populations can be extremely costly because modest-sized surveys have a low power to detect change. A strategy of postponing assessments over some years until sufficient power has been accumulated to detect change, combined with an acceptance of a reduction in detection power, offers fishery managers a more affordable method of monitoring most bycatch populations. The results of this study should encourage other trawl fisheries with large and diverse bycatch to consider the benefits of adopting this strategy.

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Table 4. The minimum number of trawls (n) required to detect declines in catch rates of five abundance categories of bycatch taxa of 20%, 50% or 99.9%, after one, three or five years of annual surveys in two different regions of the Northern Prawn Fishery. Assumed power $(1 - \beta) = 90\%$, α is 0.05. Freq = no. of taxa. Shaded boxes indicate the abundance categories where the annual surveys have sufficient power to detect given declines. The level of dispersion (ϕ) and hence the model implied is shown for each abundance category.

North of Mornington Island (52 trawls)				Declines after one year			Declines after three years			Declines after five years				
Abundance category	Freq	Mean no. per trawl	ϕ	20%	50%	99.9%	20%	50%	99.9%	20%	50%	99.9%		
				<i>n</i>	<i>n</i>	<i>n</i>	<i>n</i>	<i>n</i>	<i>n</i>	<i>n</i>	<i>n</i>	<i>n</i>		
<i>V. Rare</i>	50	0.02	∞	24933	3492	541	4444	915	359	2081	576	323		
<i>Rare</i>	102	0.09	∞	5652	792	123	1008	208	82	472	131	74		
<i>Common</i>	61	0.68	∞	717	101	17	129	27	11	61	18	10		
<i>Abundant</i>	38	7.77	2.08	283	36	4	47	9	3	21	5	3		
<i>V. Abundant</i>	15	49.94	2.88	168	21	2	28	5	2	12	3	2		
Total taxa (%) with detectable declines					20%	43%		20%	43%	43%		20%	43%	43%

North of Groote Eylandt (43 trawls)

<i>V. Rare</i>	82	0.03	∞	15536	2176	337	2770	571	224	1297	359	202			
<i>Rare</i>	113	0.16	∞	3067	430	67	547	113	45	257	72	41			
<i>Common</i>	67	1.83	∞	269	38	7	49	11	5	23	7	4			
<i>Abundant</i>	30	17.40	2.01	255	32	3	42	7	3	19	4	2			
<i>V. Abundant</i>	3	73.09	17.00	34	5	1	6	2	1	3	1	1			
Total taxa (%) with detectable declines					1%	34%	34%		11%	34%	34%		34%	34%	72%

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Table 5. The minimum number of trawls (n) required to detect declines in catch rates of five abundance categories of bycatch taxa of 20%, 50% or 99.9%, after one, three or five years of annual surveys in two different regions of the Northern Prawn Fishery. Assumed power $(1 - \beta) = 70\%$, α is 0.05. Freq = no.of taxa. Shaded boxes indicate the abundance categories where the annual surveys have sufficient power to detect given declines. The level of dispersion (ϕ) and hence the model implied is shown for each abundance category.

North of Mornington Island (52 trawls)				Declines after one year			Declines after three years			Declines after five years				
Abundance category	Freq	Mean no. per trawl	ϕ	20%	50%	99.9%	20%	50%	99.9%	20%	50%	99.9%		
				n	n	n	n	n	n	n	n	n		
<i>V. Rare</i>	50	0.02	∞	14646	2052	318	2611	538	211	1223	339	190		
<i>Rare</i>	102	0.09	∞	3321	466	73	593	123	49	278	78	44		
<i>Common</i>	61	0.68	∞	422	60	10	76	16	7	36	11	6		
<i>Abundant</i>	38	7.77	2.08	167	22	3	28	5	2	13	4	2		
<i>V. Abundan</i>	15	49.94	2.88	99	13	2	17	3	1	8	2	1		
Total taxa (%) with detectable declines					20%	43%		20%	43%	81%		43%	43%	81%

North of Groote Eylandt (43 trawls)															
<i>V. Rare</i>	82	0.03	∞	9126	1279	199	1627	336	132	762	211	119			
<i>Rare</i>	113	0.16	∞	1802	253	40	322	67	27	151	43	24			
<i>Common</i>	67	1.83	∞	158	23	4	29	7	3	14	5	3			
<i>Abundant</i>	30	17.40	2.01	150	19	2	25	5	2	11	3	2			
<i>V. Abundant</i>	3	73.09	17.0	21	3	1	4	2	1	2	1	1			
Total taxa (%) with detectable declines					1%	34%	72%		34%	34%	72%		34%	72%	72%

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Appendix 1: Asymptotic sample size calculations for counts of species

We derive the expression for the sample size, n assuming two different types of densities: Poisson and negative binomial. This method is based on the assumption that asymptotically, $2(l(y; \tilde{\theta}_a) - l(y; \tilde{\theta}_0))$ is distributed as $\chi_C^2(x|\eta)$, where η represents the non-centrality parameter and C corresponds to the degrees of freedom. An approximation for η can be derived as $\eta = 2 \times E_{\tilde{\theta}} \{l(y; \tilde{\theta}_a) - l(y; \tilde{\theta}_0)\}$ where $\tilde{\theta}_a$ and $\tilde{\theta}_0$ represent the alternative and null hypothesis values of θ respectively, for large n and θ is the parameter of interest.

Poisson Density

The log-likelihood for a single observation i assuming a Poisson density for the distribution of counts y is $l(y_i; \mu_i) = y_i \log(\mu_i) - \log(y_i!) - \mu_i$.

The expected value of the log-likelihood of μ_a can be written as $E_{\mu_a} [l(y; \mu_0)] = \mu_a \log(\mu_0) - K - \mu_0$ where K represents a constant and μ_a and μ_0 represent the mean count corresponding to the alternative and null hypothesis respectively. The non-centrality parameter can therefore be expressed as $\eta(\alpha, \beta) = 2\{\mu_a \log\left(\frac{\mu_a}{\mu_0}\right) - \mu_a + \mu_0\}n_p$, where $\eta(\alpha, \beta)$ is approximated using the percentage points of a standard Normal variate.

Substituting the expression for $\mu_0 = \lambda$ and $\mu_a = \lambda\gamma^i$ into the expression for $\eta(\alpha, \beta)$ yields $\eta(\alpha, \beta) = 2\{\lambda\gamma^i \log \gamma^i - \lambda\gamma^i + \lambda\}n_p$ from which, we can derive an expression for n_p as

$$n_p = \eta(\alpha, \beta) / \frac{1}{T+1} \sum_{i=0}^T 2\{\lambda\gamma^i \log \gamma^i - \lambda\gamma^i + \lambda\},$$

where T represents the number of years that monitoring takes place.

Negative Binomial Density

The log-likelihood for a single observation i assuming a negative binomial density for the distribution of counts y is

$$l(y_i; \mu_i, \phi) = \log(\Gamma(\phi + y_i)) - \log(\Gamma(\phi)) - \log(\Gamma(y_i + 1)) + \phi \log \phi - \phi \log(\mu_i + \phi) + y_i \log \mu_i - y_i \log(\mu_i + \phi)$$

The expected value of the log-likelihood of μ_a can be written as

$$E_{\mu_a} [l(y; \mu_0, \phi)] = K - \log \Gamma(\phi) + \phi \log \phi - \phi \log(\mu_0 + \phi) + \mu_a \log \mu_0 - \mu_a \log(\mu_0 + \phi),$$

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where K is a constant, ϕ represents the overdispersion parameter and μ_a and μ_0 are as defined previously. The non-centrality parameter can therefore be expressed as

$$\eta(\alpha, \beta) = 2 \left\{ \phi \log \left(\frac{\mu_0 + \phi}{\mu_a + \phi} \right) + \mu_a \log \left(\frac{\mu_a}{\mu_0} \right) + \mu_a \log \left(\frac{\mu_0 + \phi}{\mu_a + \phi} \right) \right\} n_{NB}$$

Substituting the expression for $\mu_0 = \lambda$ and $\mu_a = \lambda \gamma^i$ into the expression for $\eta(\alpha, \beta)$ yields

$\eta = 2 \left\{ \lambda \gamma^i \log(\lambda^i) + \log \left(\frac{\lambda + \phi}{\lambda \gamma^i + \phi} \right) (\phi + \lambda \gamma^i) \right\} n_{NB}$ from which we can derive an expression for n_{NB} as

$$n_{NB} = \eta(\alpha, \beta) / \frac{1}{T+1} \sum_{i=0}^T 2 \left\{ \lambda \gamma^i \log(\gamma^i) + \log \left(\frac{\lambda + \phi}{\lambda \gamma^i + \phi} \right) (\phi + \lambda \gamma^i) \right\}$$

where T represents the number of years that monitoring takes place.

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5.2.3 Detecting declines in Threatened, Endangered, Protected and 'at-risk' bycatch species

P. Kuhnert, D. Heales and D. Brewer

Abstract

The ability of both fishery-dependent and fishery-independent survey methods to capture the variability in catch rates was compared for a range of bycatch species in Australia's Northern Prawn Fishery. Higher variability in catch rates leads directly to higher levels of sampling effort needed to detect declines in catches from year to year. Both methods have similar abilities to detect declines. However, fishery-independent surveys use random designs with temporally and spatially controlled sampling; whereas commercial vessels use more focussed and locally intensive trawling patterns. The similarity in sampling power between these two methods may not be the same when comparing longer term, time series data sets. Nets with Turtle Excluder Devices installed greatly reduce catch rates and increase variability for larger, often Threatened, Endangered, Protected and 'at-risk' species (e.g. sea turtles, sharks, rays and sea snakes). Consequently, only fishery-dependent surveys can provide high enough numbers of trawls to deliver the sampling power needed to detect declines in the more rarely-caught species recommended for monitoring. The levels of fishery-dependent sampling effort required to detect declines in catch rates of Threatened, Endangered, Protected and 'at-risk' bycatch species were also determined. The monitoring effort required to detect declines is greater for (i) species with low mean catch rates and higher variability, (ii) when shorter time frames are required and (iii) when detecting smaller levels of decline. Sea turtles, in particular, require large numbers of trawls (between 24,077 and 124,121,721) to detect declines from year to year where TEDs are in use. However, some of the more common sea snakes, for example, require less than 100 trawls to detect declines over longer time frames. Although the total fleet capacity is 33,600 trawls, data on most of the species groups will be collected using methods that currently use only part of this sampling effort, such as crew-member observers and scientific observers, but can be varied depending on the effort required. Consequently, selecting appropriate monitoring effort levels for each species will depend on several factors, including a trade-off between sampling effort levels, acceptable rates of decline and the timeframe required. This should also be influenced by life history characteristics of each species group, including whether they are long or short lived and whether they can tolerate lower or higher levels of decline.

Introduction

Between 6.8 million tonnes (Kelleher, 2004) and 20 million tonnes (FAO, 1999) of bycatch are caught each year in fisheries throughout the world, of which approximately one third is the result of prawn trawling (Alverson et al., 1994). Due to the increasing requirement for fisheries to assess their impacts on these species, monitoring changes in catch rates of non-target species in trawl fisheries is a current high priority area, particularly for listed Threatened, Endangered and Protected (TEP) species such as sea turtles.

The Northern Prawn Fishery (NPF) impacts a range of TEP bycatch species including sea turtles, sea snakes, pipefish (syngnathids), as well as other potentially vulnerable sawfish, sharks, rays, small fish and invertebrates. Most bycatch species when returned to the water are either dead or dying (Wassenberg and Hill, 1989; Hill and Wassenberg, 1990, 2000; Alverson, 1997; FAO, 2002). The NPF has committed to establishing a long-term bycatch monitoring program (BMP)

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as part of its strategy to demonstrate sustainability for all impacted species and communities. In order to introduce a cost-effective and acceptable program, a comparison of bycatch monitoring methods has been undertaken. However, this program requires estimates of the effort required to detect change in the different species being targeted.

The study in Section 5.2.2 investigated declines in catch rates of hundreds of small bycatch species (both teleosts and invertebrates) grouped by abundance, by using an underlying model suitable for count data (negative binomial and Poisson models) (Section 5.2.2). Their results also focussed on scenarios using catch records from fishery-dependent surveys conducted in the Gulf of Carpentaria.

The research described here compliments this earlier work and considers a broader range of statistical models as a framework for calculating the necessary sample sizes to detect declines in catch rates for a broader range of bycatch species. These include important TEP and other species identified as 'at risk', including sea turtles, sea snakes and sawfish. The models also can account for the exclusion of larger species by Turtle Excluder Devices (TEDs). The compulsory use of TEDs was introduced in the NPF in 2000.

The first stage compared the ability of two different types of survey methods, fishery-independent and fishery-dependent, to capture the variability in catch rates (without TEDs) for a range of bycatch species. Fishery-independent surveys commonly exert a much higher degree of control over survey structure; compared to the usually more localised and intensive search patterns used by commercial vessels (fishery-dependent).

The second stage focussed on the specific problem of detecting a range of declines (between 10% and 90%) in bycatch species to be included in the NPF's long-term BMP (e.g. sea turtles, sea snakes and sawfish and other species identified as 'at risk'). For some species that were efficiently excluded by TEDs (e.g. sea turtles – 99% exclusion, Brewer et al., 2006), it became obvious that only the large fishery-dependent data sets would have any chance of detecting declines in catch rates. These models are based on the observed mean catch rates for the species of interest in a region, their variance and probability of capture when TEDs are deployed.

The results from both this study and the previous study (Section 5.2.2) are critical to understanding the levels of sampling effort that are required to detect a range of declines in important bycatch species, and to the development of a feasible long-term monitoring program.

Methods

Fishery-dependent versus fishery-independent surveys

In order to develop the most cost-effective monitoring program for bycatch species, the ability of two different survey methods to capture the variability in catch rates for a range of bycatch species was examined. Fishery-dependent surveys – e.g. commercial fishing vessels with a scientific observers on board – are driven by purely commercial incentives and exhibit much more localised and intensive patterns of trawling. In direct contrast, fishery-independent surveys commonly exert a high level of control, both spatially and temporally, and usually aim to stratify survey sites between a range of strata including low and high fishing effort, region and depth. It is important when comparing the two different survey methods to match the datasets, temporally and spatially, in order to maximise the validity of the comparison.

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High coefficients of variation in catch rates are directly linked to higher levels of sampling effort required to detect set levels of change. The coefficients of variation in catch rates of bycatch species from the two methods were compared in order to assess the differences in sampling effort (and ultimately cost) for the survey methods being evaluated for the long-term bycatch monitoring program.

This study focused on comparing the sampling effort provided by each survey method, to detect declines between 10% and 90% for a range of bycatch species. Although detecting both declines and increases in catch rates can be useful, the emphasis on declines is used here as it provides a precautionary approach for the monitoring program and the fishery since more effort is required to detect a statistically significant decline than an increase.

Fishery-dependent survey data was collected by scientific observers from 143 trawls on commercial vessels in two regions (Nth Groote and Vanderlins) in the NPF between September and October over two consecutive years (1996 and 1997). The vessels were twin-rigged and towed either 12 or 14 fathom headrope length nets at around 3.2 knots, for an average trawl duration of 3.1h. Sampled nets were not fitted with TEDs.

Fishery-independent data was collected by scientists from 522 trawls from the CSIRO Research vessel "Southern Surveyor" in two regions (Nth Groote and Vanderlins) in the NPF between September and October over two consecutive years (1997 and 1998). The vessel was rigged with a single 12 fathom headrope, which was towed at around 3 knots for an average trawl duration of 0.5h. These nets were also not fitted with TEDs.

For both fishery-dependent and fishery-independent surveys, large bycatch species (e.g. sea turtles, sawfish, and other elasmobranchs) were identified and counted on board and released alive where possible. Small bycatch species were subsampled (usually at least 10% of the catch), frozen at sea, and freighted to the laboratory, where individual taxa were, identified, counted, weighed and data entered into a database.

Mean catch rates and corresponding variances were estimated for bycatch species caught by both types of survey methods.

Sampling power of Threatened, Endangered, Protected and 'at-risk' species

The results from the fishery-dependent and fishery-independent comparison shows that (i) there are few differences between fishery-dependent and fishery-independent surveys, and (ii) only the fishery-dependent surveys have the power to detect declines in many rarely caught species, particularly those that are effectively excluded by TEDs (see below). In order to define the limits of sampling power provided by the fishery for species to be included in a monitoring program, a dataset was used that contained larger sample sizes and a broader range of the species of interest; especially the species forming the focus of the NPF bycatch monitoring program.

In 2001, five scientific observers collected bycatch data throughout the entire spring tiger prawn season on commercial vessels in the Gulf of Carpentaria to assess the impacts of the recent introduction of TEDs and BRDs on NPF catches. A total of 1632 trawls were sampled for a range of bycatch species including sea turtles, sawfish, seas snakes, and total bycatch. Large bycatch species (e.g. sea turtles, sawfish, and other elasmobranchs) were identified and counted on board,

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released alive where possible, and the data recorded. The vessels were twin-rigged and usually towed one net fitted with a TED and the other without in a paired comparison (see Brewer et al., 2006 for gear description). The data set resulting from this program was analysed to estimate the effort required to detect declines in species selected for the NPF bycatch monitoring program.

Sample Size Calculations

Various approaches exist for identifying the number of samples required in a survey to meet the logistical, economic and statistical requirements. These include simulation approaches to generate data to assess the consequences of taking samples of different sizes under a given effect size, level of significance and power (Tyre et al., 2001; Field et al., 2004, 2005), bootstrapping to calculate power given a pilot dataset (Manly, 1992), and asymptotic methods that can also be used to calculate power (Cox and Hinkley, 1974; Self et al., 1992; Heales et al., 2006). Asymptotic methods have the advantage over other methods by simply and quickly calculating the sample size n , through an expression that is derived by comparing the likelihood ratio of the null and alternative hypotheses for a defined generalised linear model. They are also far less computational than the bootstrap and simulation methods. However, sample size estimates need careful interpretation since estimates are approximated and represent a lower bound on the number of trawls that should be taken in order to detect the type of change specified.

Densities for count data with zero inflation

We adopted an asymptotic approach to determine the number of standard trawls required to detect declines in the catch rates of bycatch. This approach to sample size calculation is computationally convenient and effective as it finds the sample size necessary to give the likelihood ratio test of “no change in decline” (H_0) for a given level of power, significance and variability in catch rate defined in a scenario of interest.

The alternative hypothesis (H_a) that was considered for bycatch species was that the mean count μ declines by a fixed proportion, (γ) per annum, where i is used to index the year, ranging from year 0 to year T .

$$H_0 : \mu_0 = \lambda$$

$$H_a : \mu_a = \lambda\gamma^i \quad i = 0, 1, \dots, T$$

The fixed proportion, γ is a function of the effect size (δ), ($\gamma = 1 - \delta/100$) where δ is expressed as a percentage. So if the mean catch rate declines by a fixed proportion of $\gamma = 0.1$ per year, an overall decline or “effect size” of 90% over the monitoring period, T would be expected. In the above expression μ_0 and μ_a are both equal to the same mean count, λ when $i = 0$.

A zero-inflated model for monitoring declines in bycatch species was used because the introduction of TEDs from 2000 onwards had the capacity to prevent larger bycatch species such as sharks, rays and sawfish from being caught in the trawl net. The premise here is that if a species is excluded from the catch, monitoring of that species becomes limited. This situation is particularly relevant to the TED and BRD dataset.

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The zero-inflated model represents a mixture of a point mass at zero with probability $1-p$ (representing structural zeros) and a density, either Poisson or negative binomial, with probability, p that accommodates the random zeros and remaining count data. Here, structural zeros represent zeros caused by a known process such as the inclusion of TEDs while random zeros arise from random, unknown processes.

Zero-inflated models have been used in practice to model a range of environmental problems (Lambert, 1992; Welsh et al., 1996; Kuhnert et al., 2006; Martin et al., 2005). However, apart from the work by Tyre et al. (2001) and Field et al. (2004, 2005), they have not been used to determine sample size requirements for detecting declines in catch rates.

The densities for the zero inflated Poisson and zero-inflated negative binomial models are shown in Equations 1 and 2 respectively.

$$f(y; \mu, p) = \begin{cases} 1-p + p \exp(-\mu) & \text{if } y = 0 \\ p \mu^y \exp(-\mu) / y! & \text{if } y > 0 \end{cases} \quad (1)$$

$$f(y; \mu, \phi, p) = \begin{cases} 1-p + p \left(\frac{\phi}{\mu + \phi} \right)^\phi & \text{if } y = 0 \\ p \frac{\Gamma(\phi + y)}{\Gamma(\phi)\Gamma(y+1)} \left(\frac{\phi}{\mu + \phi} \right)^\phi \left(\frac{\mu}{\mu + \phi} \right)^y & \text{if } y > 0 \end{cases} \quad (2)$$

In the above equations, y represents a vector of counts, μ represents the mean catch rate and ϕ represents the over-dispersion parameter for the zero-inflated model that is incorporated in the negative binomial density.

Both expressions incorporate p , the probability that the species is included into a trawl net when it encounters a TED. When $p = 1$ in each of these expressions, the densities collapse down to the Poisson and negative binomial densities, standard densities used for modelling count data. These two densities were the focus of Section 5.2.2, where sample size calculations were evaluated for different rates of decline and catch rates based on broad species groupings.

Expressions for sample size calculations

Expressions for calculating the sample sizes required for a particular fishing scenario using the zero-inflated densities shown in Equations 1 and 2 are outlined in Appendix 1. They are also displayed below in Equation 3 and are easily programmed into R (Ihaka and Gentleman, 1995). Note, the non-centrality parameter, $\eta(\alpha, \beta)$ can be written in terms of the percentage points of a standard Normal variate: $(z_{1-\alpha/2} + z_{1-\beta})^2$ (Rochon, 1989) for both expressions.

$$n_{ZIP} = \eta(\alpha, \beta) / \frac{1}{T+1} \sum_{i=0}^T 2 \left\{ 1-p + p \exp(-\lambda \gamma^i) [\log(1-p + p \exp(-\lambda \gamma^i)) - \log(1-p + p \exp(-\lambda))] + p \lambda \gamma^i \log \gamma^i + p \lambda (1 - \exp(-\lambda \gamma^i))(1 - \gamma^i) \right\} \quad (3a)$$

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$$n_{ZINb} = \eta(\alpha, \beta) / \frac{1}{T+1} \sum_{i=0}^T 2 \left\{ \left[1 - p + p \left(\frac{\phi}{\lambda \gamma^i + \phi} \right)^\phi \right]^\times \right. \\ \left. \left[\log \left(1 - p + p \left(\frac{\phi}{\lambda \gamma^i + \phi} \right)^\phi \right) - \log \left(1 - p + p \left(\frac{\phi}{\lambda + \phi} \right)^\phi \right) \right] + \right. \\ \left. p \lambda \gamma^i \log(\gamma^i) + \log \left(\frac{\lambda + \phi}{\lambda \gamma^i + \phi} \right) \left[p \lambda \gamma^i + p \phi \left(1 - \left(\frac{\phi}{\lambda \gamma^i + \phi} \right)^\phi \right) \right] \right\} \quad (3b)$$

Sample size calculations

Catch rates estimated from the TED and BRD fishery-dependent surveys (Brewer et al., 2006) were used to construct a series of power calculations using only nets without TEDs to avoid confounding and a biased estimate of the catch rate.

The current study aims to examine the effort required to detect declines in catch rates of NPF bycatch species with 80% power and a 5% level of significance. Monitoring frameworks of one year, five years and ten years were chosen to describe different effort scenarios, based on different time frames for monitoring. The ability of TEDs to exclude some of the larger species of bycatch from the trawl nets was also taken into account. The species to be included in the monitoring program and their inclusion probabilities when using TEDs are listed in Table 6.

Results

Fishery-dependent versus fishery-independent surveys

Figure 8 compares the coefficient of variation (log-scale) for catch rates computed for a standard trawl from fishery-independent and dependent surveys (from Groote Eylandt region only). The dotted line in Figure 8 represents the cases where both the fishery-independent and fishery-dependent scenarios capture similar mean-variance relationships for the same species in a standard trawl. Species occurring away from the diagonal indicate scenarios where the surveys capture the mean-variance relationship differently. For example, the fishnet lizard fish (*Synodus sageneus*) appears to be captured well by the fishery-independent survey but not by the fishery-dependent survey as the variability around the mean catch rate is lower. The variation around the mean catch rate for a selection of sea turtles, rays and seas snakes are captured well by the fishery-dependent survey but not for the fishery-independent survey. There are also a collection of the rarer species (top right of the plot) whose variability is not captured well by either method. A similar plot can be produced for the Vanderlins region.

Figure 9 summarises the variability (or coefficient of variation) with respect to the mean catch rate of bycatch recorded in a standard trawl. Species that are rarely caught tend to be highly variable. This group is represented by some species of fish (*S. sageneus*, *S. indicus*, *S. jaculum* and *Harpadon translucens*), sea turtles (*Chelonia mydas* and *Eretmochelys imbricata*) and one species of sawfish (*Pristis zijsron*) (Figure 9). Variability also seems to be mostly consistent across the two regions.

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The study is equivocal about the ability of either fishery-independent or fishery-dependent surveys to capture the variability in catch rates. This may not be the case for comparing longer term, time series data sets. However, it is clear that high variability is directly linked to the need for high levels of effort to detect declines, and only the fishery-dependent surveys are capable of providing this power. The bycatch species recommended for monitoring in the NPFs BMP include TEP and ‘at-risk’ species, most of which are relatively rare. Consequently, it is likely that the higher sampling capability of fishery-dependent methods will need to be used to monitor these species and the results of power calculations to describe the effort levels needed for these methods are described below.

Sampling power for detecting declines in Threatened, Endangered, Protected and ‘at-risk’ species

Figures 10-13 display the results of power calculations constructed from mean-variance relationships for fishery-dependent data for the TEP and ‘at-risk’ bycatch species recommended for monitoring in the NPFs BMP. These show the number of standard trawls required to monitor a range of declines over one, five and ten year time frames, and for trawls with and without TEDs. These results describe a wide range of possible monitoring effort levels for the species of interest.

The monitoring effort required to detect declines is greater (i) for species with smaller mean catch rates and higher variance, (ii) when shorter time frames are required and (iii) and when detecting smaller levels of decline. Sea turtles, in particular, require large numbers of trawls (between 24,077 and 124,121,721) to detect declines from year to year in this fishery where TEDs are in use (Figure 12a). However, some of the more common sea snakes, for example, require less than 100 trawls annually to detect declines over longer time frames (Figure 13).

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Table 6: The mean catch rates and variances of the TEP and identified ‘at risk’ bycatch species caught in the Northern Prawn Fishery to be included in the long-term bycatch monitoring program. p_i is the probability of inclusion of species in trawl nets due to TEDs.

Species Group	Species Name	Common Name	Mean Catch Rate	Variance	Inclusion Probability (p_i)
Sea turtles	<i>Caretta caretta</i>	Loggerhead Turtle	0.00612	0.00305	0.01
	<i>Natator depressus</i>	Flatback Turtle	0.02080	0.01152	0.01
	<i>Lepidochelys olivacea</i>	Pacific Ridley Turtle	0.03792	0.01861	0.01
	<i>Chelonia mydas</i>	Green Turtle	0.00245	0.00245	0.01
	<i>Eretmochelys imbricata</i>	Hawksbill Turtle	0.00122	0.00061	0.01
Sea snakes	<i>Acalyptophis peronii</i>	Horned Seasnake	0.00839	0.00910	0.95
	<i>Aipysurus eydouxii</i>	Stagger-banded Seasnake	0.02516	0.03390	0.95
	<i>Aipysurus laevis</i>	Golden Seasnake	0.02984	0.03832	0.95
	<i>Astrotia stokesii</i>	Stokes Seasnake	0.03842	0.04437	0.95
	<i>Disteira kingii</i>	Spectacled Seasnake	0.00156	0.00156	0.95
	<i>Disteira major</i>	Olive-headed Seasnake	0.08348	0.11359	0.95
	<i>Hydrophis elegans</i>	Elegant Seasnake	0.11176	0.14416	0.95
	<i>Hydrophis mcdowelli</i>	Small-headed Seasnake	0.00059	0.00058	0.95
	<i>Hydrophis ornatus</i>	Ornate Seasnake	0.01736	0.02018	0.95
	<i>Hydrophis pacificus</i>	Large-headed Seasnake	0.01619	0.01749	0.95
	<i>Lapemis hardwickii</i>	Spine-bellied Seasnake	0.06280	0.13027	0.95
Sawfish	<i>Pristidae (undifferentiated)</i>	Sawfishes	0.00546	0.00273	0.95
	<i>Anoxypristis cuspidata</i>	Narrow Sawfish	0.03415	0.01680	0.27
Elasmobranchs	<i>Urogymnus asperrimus</i>	Porcupine Ray	0.00273	0.00014	0.5
	<i>Taeniura meyeri</i>	Blotched Fantail Ray	0.00273	0.00014	0.5
	<i>Orectolobus ornatus</i>	Banded Wobbegong	0.00137	0.00068	0.6
Fish	<i>Scorpaenopsis venosa</i>	Raggy Scorpionfish	0.00273	0.00273	0.92
	<i>Dendochirus brachypterus</i>	Dwarf Lionfish	0.00546	0.00273	0.92

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Figure 14 summarises the effort required to detect declines of 20% and 50% for all of the bycatch groups recommended for inclusion in the bycatch monitoring program. It indicates that, for the five-year time frame, only three of the five sea turtle species cannot be assessed by the total fleet effort (about 33,600 trawls per year). Most of the other species groups have the potential to be assessed based on the total fleet capacity. However, the power of the fishery-dependent monitoring program to assess declines in their catches depends on the monitoring method used for each group and the proportion of the fleet participating in each part of the program (described in Sections 5.2.1 and 5.2.6).

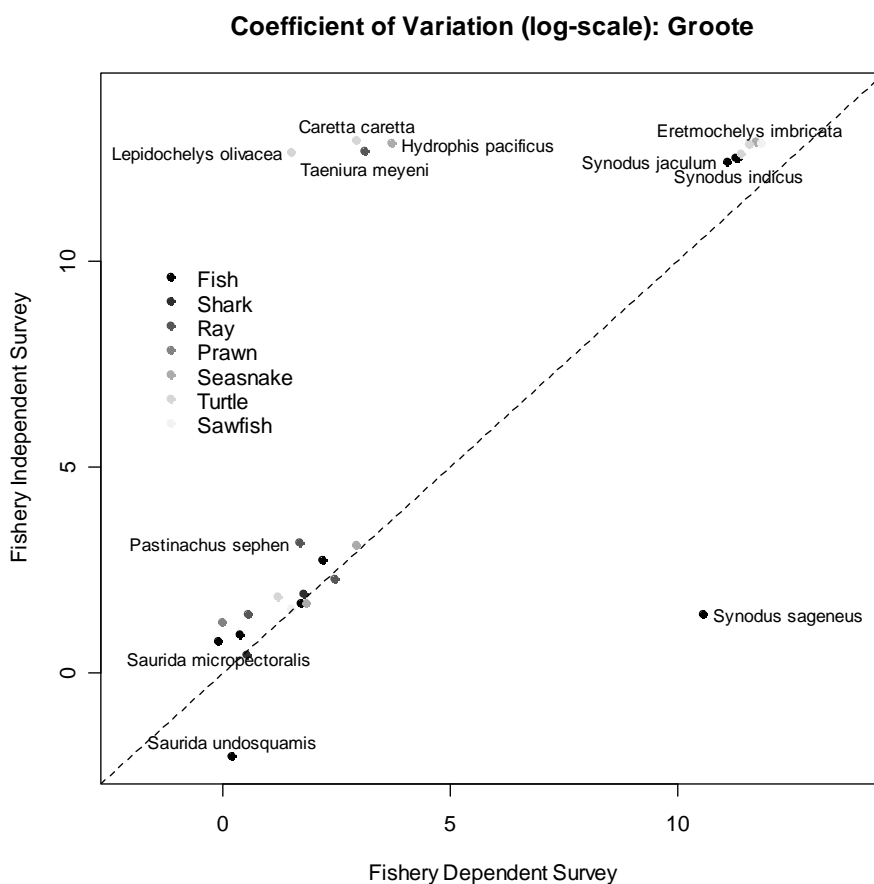


Figure 8. Coefficient of variation in catch rates of bycatch species (log-scale) for the fishery-independent and fishery-dependent studies for each species group in the Groote Eylandt region.

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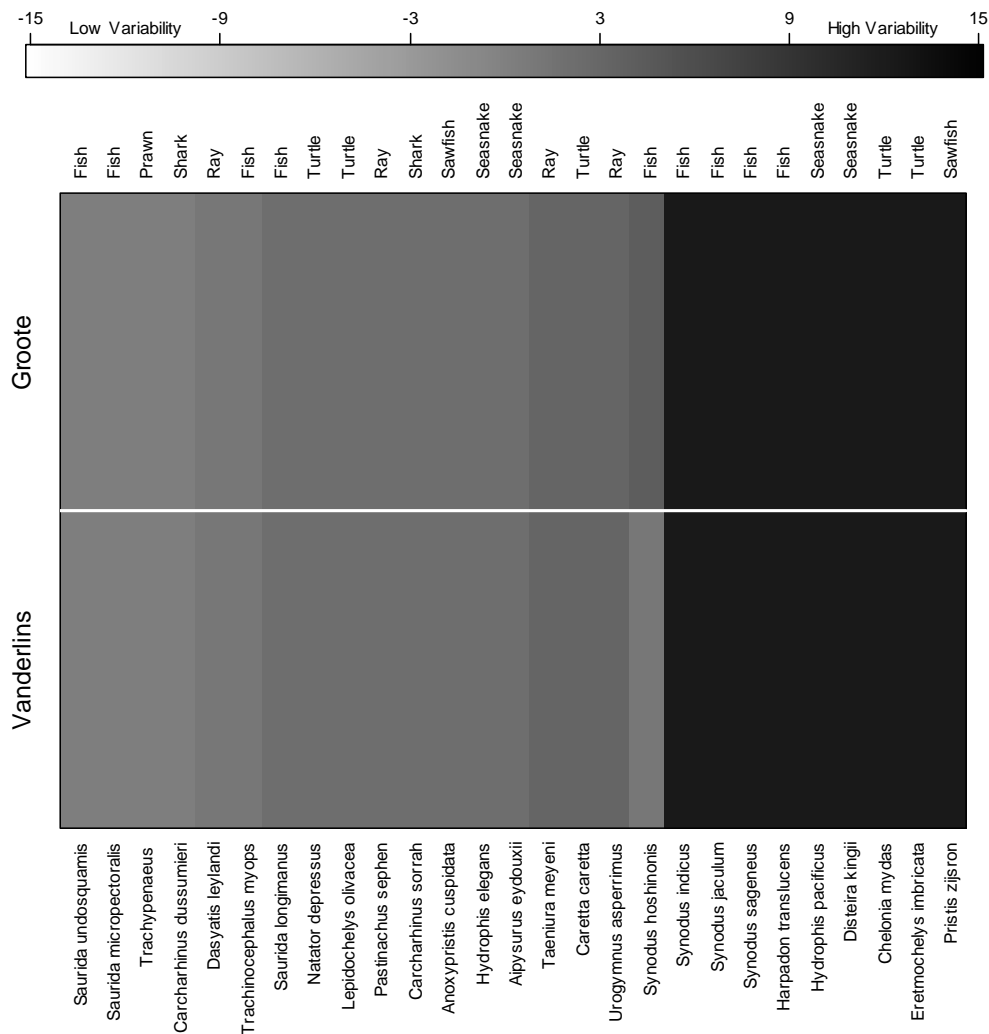


Figure 9. Coefficient of variation (log-scale) shown for each bycatch species recorded in the fishery-dependent surveys across the Groote Eylandt and Vanderlins regions. Species are ordered by their overall level of coefficient of variation across the two regions. Light grey areas represent scenarios of low variability, while black areas represent bycatch species having high variability with respect to the mean catch rate.

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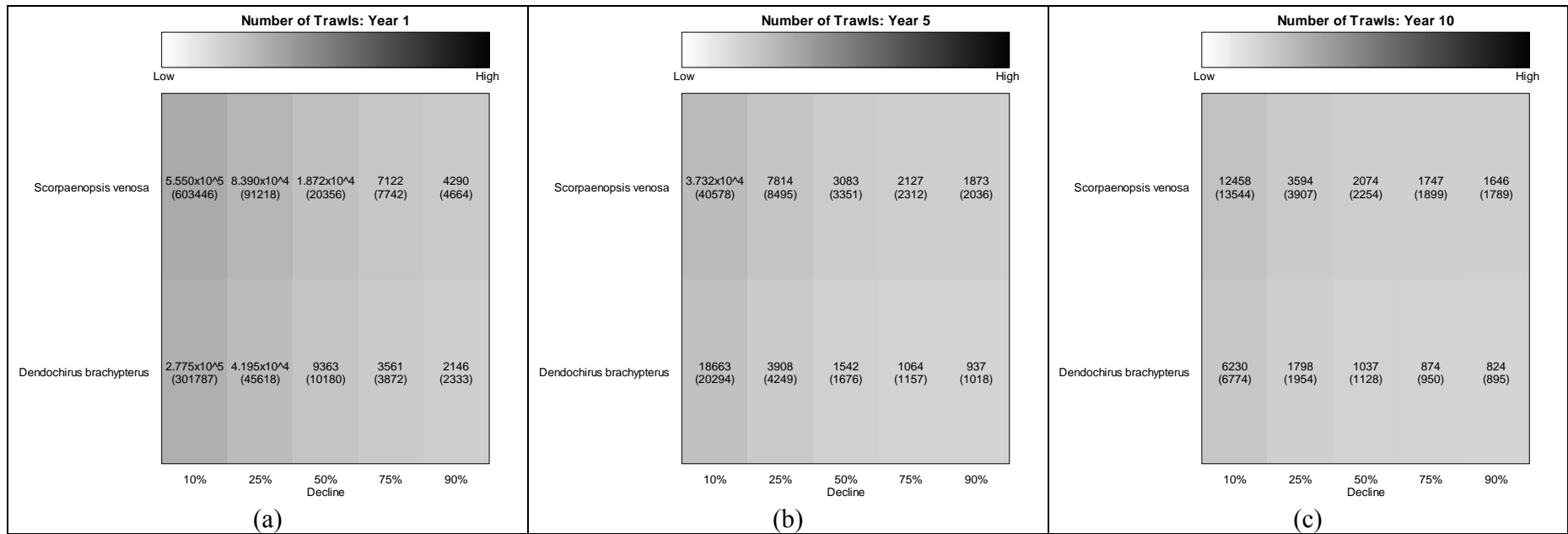


Figure 10. Number of “standard” trawls required to monitor two fish species based on 80% power and 5 % level of significance for a range of declines varying between 10% and 90% for a monitoring framework of (a) one year, (b) five years and (c) ten years. Values in brackets are where TEDs and BRDs are installed in nets.

Objective 1: Design of monitoring program

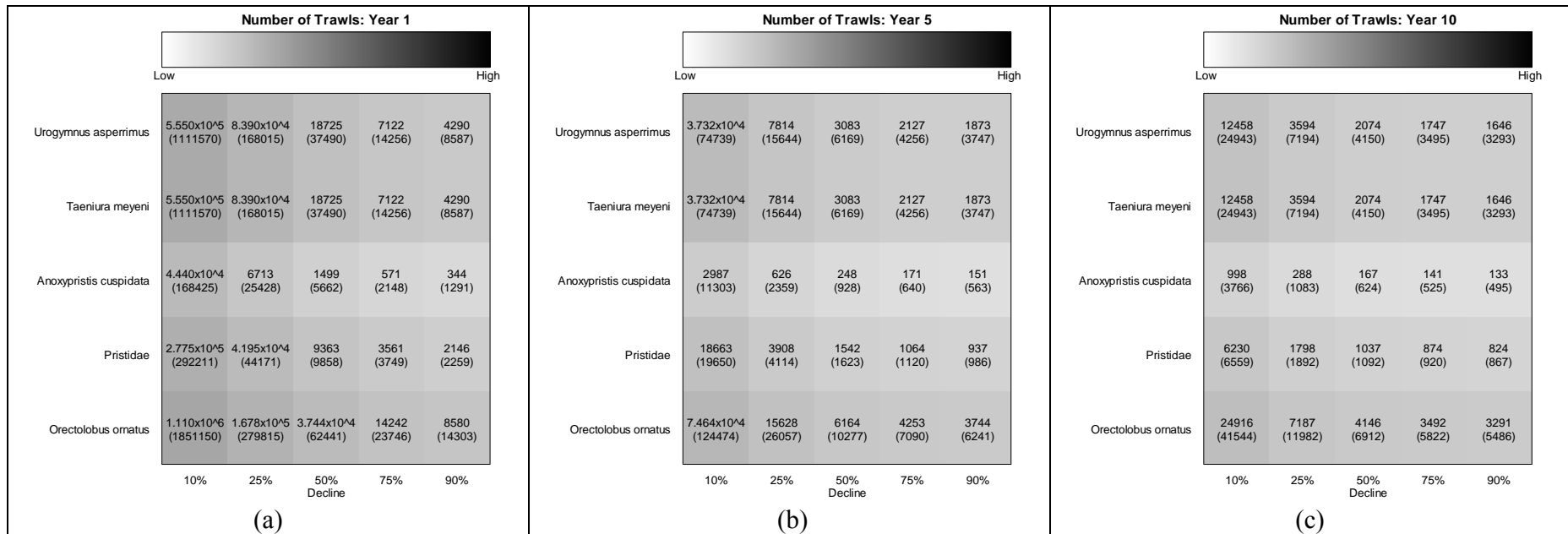


Figure 11. Number of “standard” trawls required to monitor a range of elasmobranchs and sawfish based on 80% power and 5% level of significance for a range of declines varying between 10% and 90% for a monitoring framework of (a) one year, (b) five years and (c) ten years. Values in brackets are where TEDs and BRDs are installed in nets.

Objective 1: Design of monitoring program

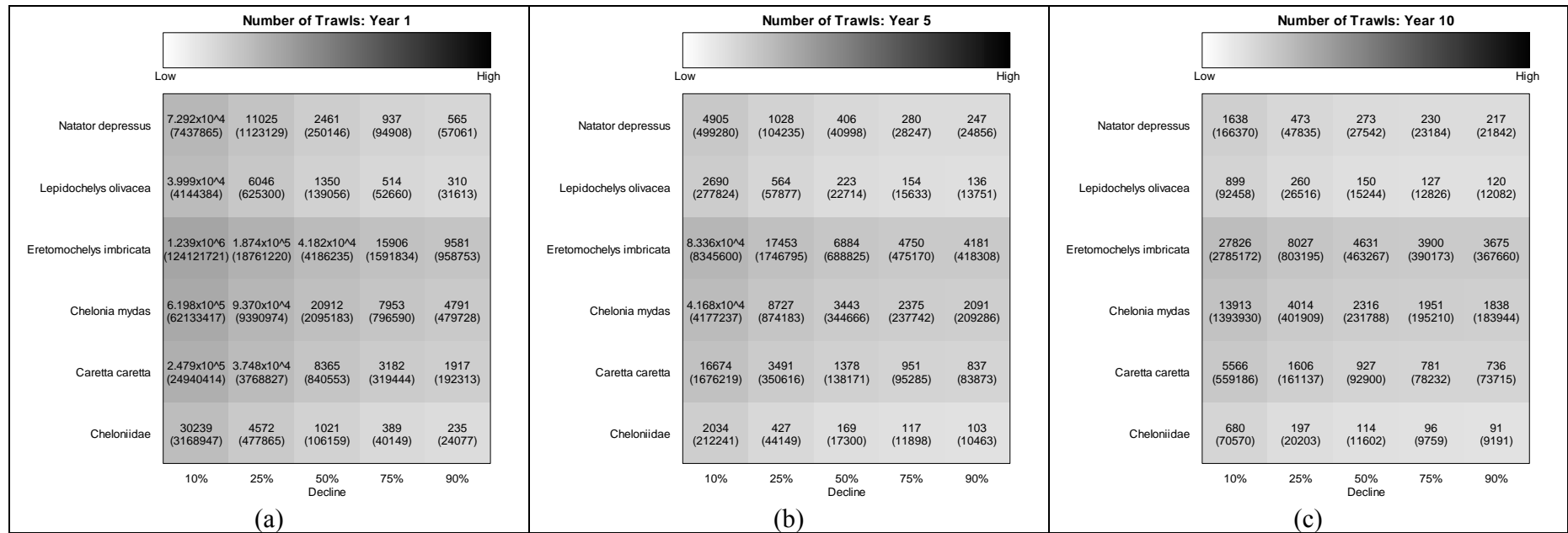


Figure 12. Number of “standard” trawls required to monitor sea turtles based on 80% power and 5 % level of significance for a range of declines varying between 10% and 90% for a monitoring framework of (a) one year, (b) five years and (c) ten years. Values in brackets are where TEDs and BRDs are installed in nets.

Objective 1: Design of monitoring program

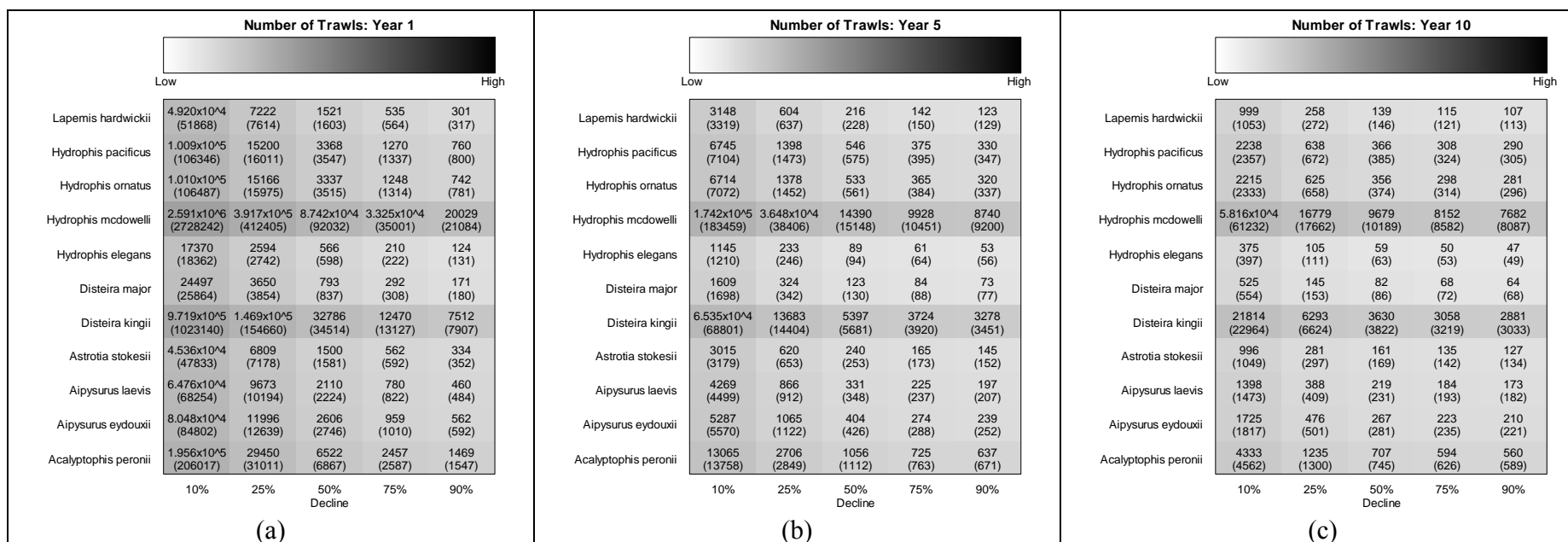


Figure 13. Number of “standard” trawls required to monitor sea snakes based on 80% power and 5 % level of significance for a range of declines varying between 10% and 90% for a monitoring framework of (a) one year, (b) five years and (c) ten years. Values in brackets are where TEDs and BRDs are installed in nets.

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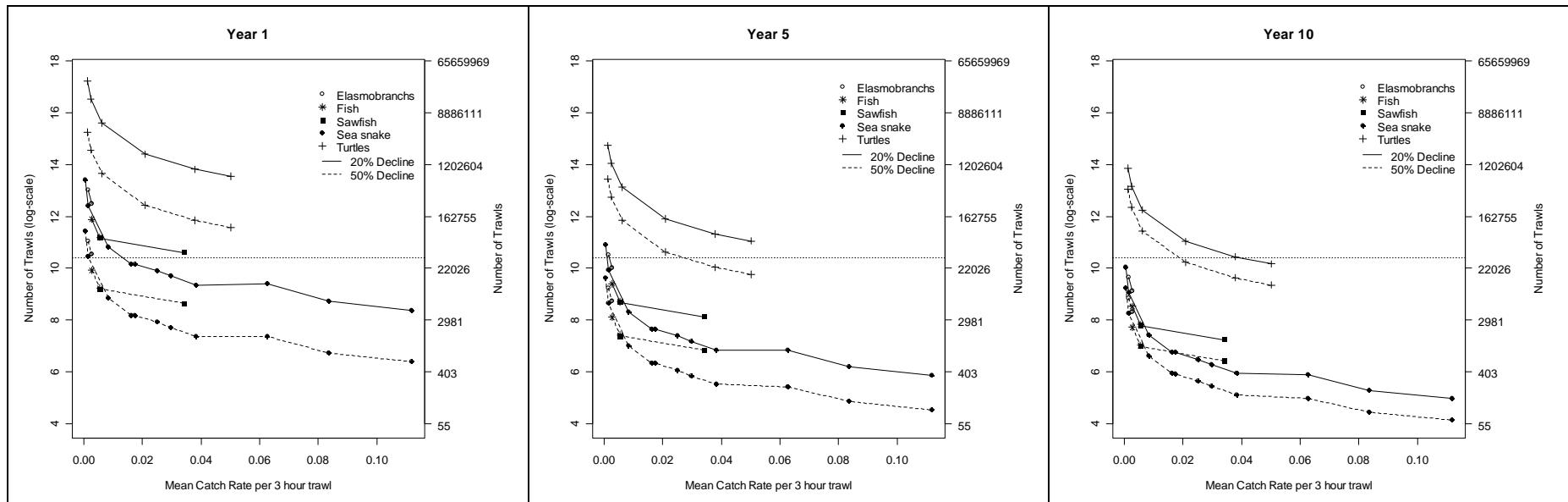


Figure 14. Plots showing the effort required to monitor 20% (solid line) and 50% (dashed line) declines for a range of mean catch rates and for monitoring timelines of (a) one year, (b) five years and (c) ten years. The number of trawls (log-scale and actual) is shown on the y-axes. Species groups are represented by the symbols shown in the legend. The dotted line represents the approximate maximum, annual sampling capacity of the fleet (33,600 trawls, 2006 data).

Discussion

The high diversity and dominance of bycatch species in Northern Prawn Fishery (NPF) trawl catches (Stobutzki et al., 2001) provides a significant challenge to designing an effective, all-encompassing bycatch monitoring program. This has been simplified by the risk assessment allowing the monitoring to focus on a small subset of the many hundreds of bycatch species impacted (Section 5.4). However, this selected group of species is still diverse and requires a broad-ranging analysis to guide the effort levels required to adequately assess changes in each species in the program.

The design of a monitoring program that is capable of detecting declines in catch rates of bycatch species relies heavily on estimating the levels of sampling effort required. Many bycatch species in the NPF are rarely caught, have highly variable catch rates and require high levels of sampling effort to detect even relatively large declines in catch rates (Section 5.2.2). The compulsory use of TEDs substantially increases the between-trawl variability in catches for large species with high exclusion rates (e.g. sea turtles, large sharks and rays; Brewer et al., 2006). This greatly affects the monitoring power for these species, and reduces the ability to detect declines.

There is little difference in the relative abilities of fishery-dependent and fishery-independent surveys to capture the variability around the mean for a range of common and rarer species. However, the variability of some rarer species was captured more effectively by the fishery-dependent surveys, and not well enough by fishery-independent surveys to understand and monitor declines in these species. These differences between the two methods appear to be mainly related to the amount of time spent trawling and the intensity of the survey. Consequently, fishery-dependent surveys are likely to be the only method capable of providing the high levels of sampling effort required to detect declines in catch rates for the mostly rare TEP species such as sea turtles, sea snakes and sawfish.

The ability of the NPF BMP to detect declines in a range of different bycatch species depends on the method being used and its capacity to provide adequate sampling power (see Sections 5.2.1 and 5.2.6). The results presented in Figures 10-13 provide the effort levels for designing the long-term BMP. The effort levels (and hence, cost) used in the BMP should be selected in conjunction with the limitations of the methods available to the BMP. These vary with the species group in question and are described in detail in Section 5.2.6. For example, fishery logbooks in 2006 collected data from about 33,600 trawls in 2006. However, they do not collect any individual species data for most of the species being targeted by the BMP. Data on many of the species groups will be collected by crew-member observers and scientific observers, who have a lower sampling potential than logbooks. The number of observers can be varied depending on the level of effort required in the program (Section 5.2.6). Consequently, selecting appropriate monitoring effort levels for each species will depend on several factors, including a trade-off between sampling effort levels, acceptable rates of decline and timeframe required.

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Appendix 1: Asymptotic sample size calculations for counts of species

The expression for the sample size, n , was derived assuming a zero-inflated Poisson density and a zero-inflated negative binomial density for situations where the variance in catch rates exceeds the mean (over dispersion). As highlighted in Section 5.2.2, the method that our sample size calculations are based on, assumes that asymptotically, $2(l(y; \tilde{\theta}_a) - l(y; \tilde{\theta}_0))$ is distributed as $\chi_C^2(x | \eta)$, where η represents the non-centrality parameter and C corresponds to the degrees of freedom. An approximation for η can be derived as $\eta = 2 \times E_{\tilde{\theta}} \{l(y; \tilde{\theta}_a) - l(y; \tilde{\theta}_0)\}$ where $\tilde{\theta}_a$ and $\tilde{\theta}_0$ represent the alternative and null hypothesis values of θ respectively, for large n .

Zero-Inflated Poisson (ZIP) Density

The log-likelihood for a single observation i assuming a ZIP density for the distribution of counts y is

$$l(y_i; \mu_i, p) = \begin{cases} \log(1 - p + p \exp(-\mu_i)) & \text{if } y_i = 0 \\ \log p + y_i \log \mu_i - \mu_i - \log(y_i!) & \text{if } y_i > 0 \end{cases}$$

The expected value of the log-likelihood of can be written as

$$E_{\mu_a} [l(y; \mu_0, p)] = (1 - p + p \exp(-\mu_a)) \log(1 - p + p \exp(-\mu_0)) + p(1 - \exp(-\mu_a)) \log p + \mu_a \log(\mu_0) - \mu_0 p(1 - \exp(-\mu_a)) - K$$

where K represents a constant, p represents the mixing probability (or probability of inclusion) and μ_a and μ_0 represent the mean evaluated for the alternative hypothesis and null hypothesis respectively. The non-centrality parameter can therefore be expressed as

$$\eta(\alpha, \beta) = 2\{(1 - p + p \exp(-\mu_a))[\log(1 - p + p \exp(-\mu_a)) - \log(1 - p + p \exp(-\mu_0))] + p\mu_a \log \frac{\mu_a}{\mu_0} + p(1 - \exp(-\mu_a))(\mu_0 - \mu_a)\}n_{ZIP}$$

Substituting the expression for $\mu_0 = \lambda$ and $\mu_a = \lambda\gamma^i$ into the expression for η yields:

$$\eta(\alpha, \beta) = 2\{(1 - p + p \exp(-\lambda\gamma^i))[\log(1 - p + p \exp(-\lambda\gamma^i)) - \log(1 - p + p \exp(-\lambda))] + p\lambda\gamma^i \log \gamma^i + p\lambda(1 - \exp(-\lambda\gamma^i))(1 - \gamma^i)\}n_{ZIP}$$

from which is derived an expression for n_{ZIP} as

$$n_{ZIP} = \eta(\alpha, \beta) / \frac{1}{T+1} \sum_{i=0}^T 2\{(1 - p + p \exp(-\lambda\gamma^i))[\log(1 - p + p \exp(-\lambda\gamma^i)) - \log(1 - p + p \exp(-\lambda))] + p\lambda\gamma^i \log \gamma^i + p\lambda(1 - \exp(-\lambda\gamma^i))(1 - \gamma^i)\}$$

Note that when $p = 1$, this expression defaults to the expression obtained for the Poisson distribution used in Section 5.2.2.

Zero-Inflated Negative Binomial (ZINb) Density

The log-likelihood for a single observation i assuming a ZINb density for the distribution of counts, y is

$$l(y_i; \mu_i, \phi, p) = \begin{cases} \log \left(1 - p + p \left(\frac{\phi}{\mu_i + \phi} \right)^\phi \right) & \text{if } y = 0 \\ \log p + \log \Gamma(\phi + y_i) - \log \Gamma(\phi) - \log \Gamma(y_i + 1) + \phi \log \phi - \phi \log(\mu_i + \phi) + y_i \log \mu_i - y_i \log(\mu_i + \phi) & \text{if } y > 0 \end{cases}$$

The expected value of the log-likelihood of μ_a can be written as

$$E_{\mu_a} [l(y; \mu_0, \phi)] = (1 - p + p \left(\frac{\phi}{\mu_a + \phi} \right)^\phi) \log \left(1 - p + p \left(\frac{\phi}{\mu_0 + \phi} \right)^\phi \right) + p \mu_a \left(\log \frac{\mu_0}{\mu_0 + \phi} \right) + \left(1 - \left(\frac{\phi}{\mu_a + \phi} \right)^\phi \right) \left(p \log p + Kp - p \log \Gamma(\phi) + p \phi \log \phi - p \phi \log(\mu_0 + \phi) \right) +$$

where K represents a constant, ϕ represents the overdispersion parameter, p represents the mixing probability and μ_a and μ_0 are as defined for the ZIP density. The non-centrality parameter can therefore be expressed as

$$\eta(\alpha, \beta) = 2 \left\{ (1 - p + p \left(\frac{\phi}{\mu_a + \phi} \right)^\phi) \left[\log \left(1 - p + p \left(\frac{\phi}{\mu_a + \phi} \right)^\phi \right) - \log \left(1 - p + p \left(\frac{\phi}{\mu_0 + \phi} \right)^\phi \right) \right] + p \left(1 - \left(\frac{\phi}{\mu_a + \phi} \right)^\phi \right) \left[\phi \log(\mu_0 + \phi) - \phi \log(\mu_a + \phi) \right] + p \mu_a \log \left(\frac{\mu_a}{\mu_0} \right) + p \mu_a \log \left(\frac{\mu_0 + \phi}{\mu_a + \phi} \right) \right\} n_{ZINb}$$

Substituting the expression for $\mu_0 = \lambda$ and $\mu_a = \lambda \gamma^i$ into the expression for $\eta(\alpha, \beta)$ yields:

$$\eta = 2 \left\{ (1 - p + p \left(\frac{\phi}{\lambda \gamma^i + \phi} \right)^\phi) \left[\log \left(1 - p + p \left(\frac{\phi}{\lambda \gamma^i + \phi} \right)^\phi \right) - \log \left(1 - p + p \left(\frac{\phi}{\lambda + \phi} \right)^\phi \right) \right] + p \lambda \gamma^i \log \gamma^i + \log \left(\frac{\lambda + \phi}{\lambda \gamma^i + \phi} \right) \left[p \lambda \gamma^i + p \phi \left(1 - \left(\frac{\phi}{\lambda \gamma^i + \phi} \right)^\phi \right) \right] \right\} n_{ZINb}$$

from which is derived an expression for n_{ZINb} as

$$n_{ZINb} = \eta(\alpha, \beta) / \frac{1}{T+1} \sum_{i=0}^T 2 \left\{ (1 - p + p \left(\frac{\phi}{\lambda \gamma^i + \phi} \right)^\phi) \left[\log \left(1 - p + p \left(\frac{\phi}{\lambda \gamma^i + \phi} \right)^\phi \right) - \log \left(1 - p + p \left(\frac{\phi}{\lambda + \phi} \right)^\phi \right) \right] + p \lambda \gamma^i \log \gamma^i + \log \left(\frac{\lambda + \phi}{\lambda \gamma^i + \phi} \right) \left[p \lambda \gamma^i + p \phi \left(1 - \left(\frac{\phi}{\lambda \gamma^i + \phi} \right)^\phi \right) \right] \right\}$$

Note that when $p = 1$, this expression defaults to the expression obtained for the negative binomial distribution as discussed in Section 5.2.2.

5.2.4 Dealing with sustainability of rarely-caught species: Do alternate management strategies have a role in cost-effective bycatch monitoring?

D. Heales and D. Brewer

Abstract

Commonwealth legislation governing Australian fishery managers (see EPBC, NPF BAP) requires fisheries to demonstrate that populations of species impacted remain sustainable. Many of the small teleost and invertebrate bycatch species impacted by the Northern Prawn Fishery are rarely caught by prawn trawls. Conventional trawler-based monitoring techniques aim to detect declines in catch rates, but will be unable to deliver cost-effective monitoring for a large number of rarely-caught species (Heales *et al.* in press, Section 5.2.3). Consequently, there is a need to review a range of alternative non-trawling methods suitable for assessing, demonstrating or promoting the long-term sustainability for these species. Alternate management strategies can include any method of solving fishery problems by using unconventional approaches.

As an example of an alternate management strategy, we describe a scenario for permanently closing low effort areas as a way to promote the sustainability of bycatch species by providing refuge areas that are mostly adjacent to existing medium and high effort trawl areas. Such closures may also promote rebuilding of populations of sessile benthic fauna (and their commensal communities) that were cleared during the early days of the fishery in many regions including Weipa, North Mornington Island, Bountiful and Vanderlin fishing grounds. If fishing effort was restricted to those grids (6nm by 6nm) where more than 5 fishing days were recorded in any one year of the last five years, the total number of grids fished annually would decline by 56%. Intuitively, spatial closure strategies of this type potentially provide:

- increased insurance for long-term sustainability of species and habitats
- insurance against further expansion of the fishery into previously undisturbed habitats (e.g. other sessile benthic fauna and seagrass habitats such as the area west of Mornington Island)

In an operational sense, strategies of this type present options and compromises that may be acceptable, via negotiation, to all fishery stakeholders. Further and more detailed examination of a range of alternative management strategies (e.g. effort reductions, spatial management, restocking of selected 'at-risk' species and electric fishing) may be needed in order for the fishery to fulfill its objectives under existing Commonwealth legislation.

Background

Demonstrating the long-term sustainability of bycatch populations is problematic for fishery managers worldwide. There are many examples of global codes of conduct (FAO 1999) and national legislation aimed at reducing impacts on bycatch species (See Section 5.2.1 for review). Under Australian legislation, in particularly the EPBC Act 1999, the sustainability of all species impacted by Commonwealth and export fisheries needs to be demonstrated for Threatened Endangered and Protected species, as well as the less recognised bycatch species, and habitats. In this section (and the report in general), we have interpreted the relevant sections of the EPBC Act, to mean that all species are to be treated equally, in the need to conserve them and in the demonstration of their long-term sustainability.

Objective 1: Design of monitoring program

There are few instances in other prawn trawl fisheries worldwide where bycatch is monitored at the level of the individual species in order to assess their long-term sustainability. This is relatively easy in fisheries interacting with a small number of bycatch species, since they are easily counted and weighed. For example, in the Alaskan trawl fisheries, bycatch limits for some individual species are assessed on a daily basis, made possible by a high level of observer coverage. Spatial closures are subsequently imposed when limits are reached by the fleet (McElderry *et al.*, 1999). Some prawn trawl fisheries monitor changes in selected bycatch species that are commercial species in other fisheries. For example, in the Gulf of Mexico legislation imposes catch limits on juvenile red snapper bycatch to reduce conflict with recreational and commercial snapper fisheries. However, the remainder of the bycatch is not reported or monitored (Diamond, 2005). Consequently, there are no suitable monitoring frameworks for the Northern Prawn Fishery (NPF) to adopt to successfully monitor individual species comprising highly diverse bycatch assemblages.

The NPF managing body, the Northern Prawn Management Advisory Committee (NORMAC), has consistently been pro-active in addressing both short- and long-term bycatch sustainability issues in the fishery. For example, the compulsory use of Turtle Excluder Devices (TEDs) and Bycatch Reduction Devices (BRDs) introduced by NORMAC in 2000 has greatly reduced the fishery's impact on sea turtles (99% reduction), large sharks and rays (86-94% reduction) (Brewer *et al.*, 2006). Public scrutiny of bycatch sustainability issues in Australia (Aslin and Byron, 2003) has provided further pressure on fishery managers to promote precautionary management to ensure ecological sustainability in the absence of quantitative data. In the future, Industry, conservation organisations and the general public need to remain receptive to the use of all the available means for ensuring the sustainability of impacted bycatch species.

The bycatch of the NPF is large and diverse with 478 teleost and hundreds more invertebrate taxa impacted by the fishery (Stobutzki *et al.*, 2001a; Brewer *et al.*, 2006). The small bycatch component (i.e. mainly small fishes and invertebrates) of the average commercial trawl in the tiger prawn fishery contains around 100 species, of which about two thirds comprise teleosts, and the remaining one third invertebrates (Heales *et al.*, 2001). Compared to the relatively small number of target species in the NPF, there is limited distributional data for bycatch species and a poor understanding of their life-histories and movements. A new quantitative risk assessment method (SAFE) has recently been developed in order to focus the future monitoring program on species at true risk, rather than highest relative risk (see Stobutzki *et al.*, 2001b) of becoming unsustainable due to fishing (See Section 5.4.1). However, despite this progress a large number of species cannot be adequately monitored using conventional fishing techniques, since their rarity requires a large number of samples (and high cost) in order to detect statistically significant changes in catch rates (Section 5.2.2.1).

Neither fishery-dependent or independent platforms employing prawn trawling can provide the necessary sampling power to detect changes in the catch rates of rarely caught species. Scenarios in section 5.2.2.1 showed that after five years of fishery-dependent sampling using modest-sized annual surveys, it is unaffordable for the fishery to detect even a 50% decline in the catch rates of between 82 to 152 species (depending on region). A similar result is likely for fishery-independent sampling (Section 5.2.2.2), where rarely caught teleost species require large numbers of samples, well above an affordable size survey. Furthermore the high costs of monitoring will preclude monitoring programs in each of the 16 statistical regions of the NPF.

Objective 1: Design of monitoring program

Alternative management strategies can provide acceptable solutions to difficult problems. For example, one method of reducing diel and lunar variability in surveys of NPF Penaeid prawn target species may be to use electric trawls to sample high and consistent proportions of prawns in the path of the net. This strategy could also reduce impacts on bycatch. Another example would be to assess NPF impacted sea turtle populations by counting nesting females on shore, rather than attempting to detect population changes based on the low numbers of sea turtles currently caught in the NPF, due to the successful introduction of TEDs in 2000.

We examined the hypothetical example of permanently closing low effort areas in the NPF to ensure long term sustainability of a large number of rarely caught species impacted throughout the NPF. The example chosen also highlights the comparison of permanently closing areas, rather than using rotational closures. There are numerous anecdotal accounts of clearing sessile benthic fauna (large sponges in particular) from potential trawl grounds in the early days of the NPF (Dell pers comm.) and in other Australian fisheries. For example, removal of large sponges by fish trawlers in the North West Shelf fishery in Australia, directly led to a replacement of valuable Lutjanid and Lethrinid species by non-commercial Synodontids and Nemipterids (Sainsbury, 1988). The removal of large sponges in the early fishery days of the NPF may also have led to similar changes in community structure as documented by Sainsbury (1998).

A hypothetical example

Permanent closure of low effort areas

In our hypothetical example, we used NPF logbook effort data to identify any grid where fishing was recorded from 1999 to 2003 (Figure 15). We then reduced the number of grids by retaining just the medium and high effort grids where > 5 days of fishing effort were recorded in any one of the five years (Figure 16). This process reduced the number of fished grids by around 56%.

Discussion

For fisheries with large and diverse bycatch like the Northern Prawn Fishery (NPF), there are no simple answers to the legislative requirements that fishery managers ensure the sustainability of all impacted species. Annual bycatch surveys, either fishery-dependent or independent, will be unable to cost-effectively monitor to achieve this goal for a wide range of rarely-caught species (Section 5.2.2.1; 5.2.2.2). However, the permanent closure of low effort areas may be a reasonable compromise between the high cost of annual monitoring in all regions, and alternative assumptions that the long-term sustainability for all bycatch species will automatically follow reductions in fishing effort.

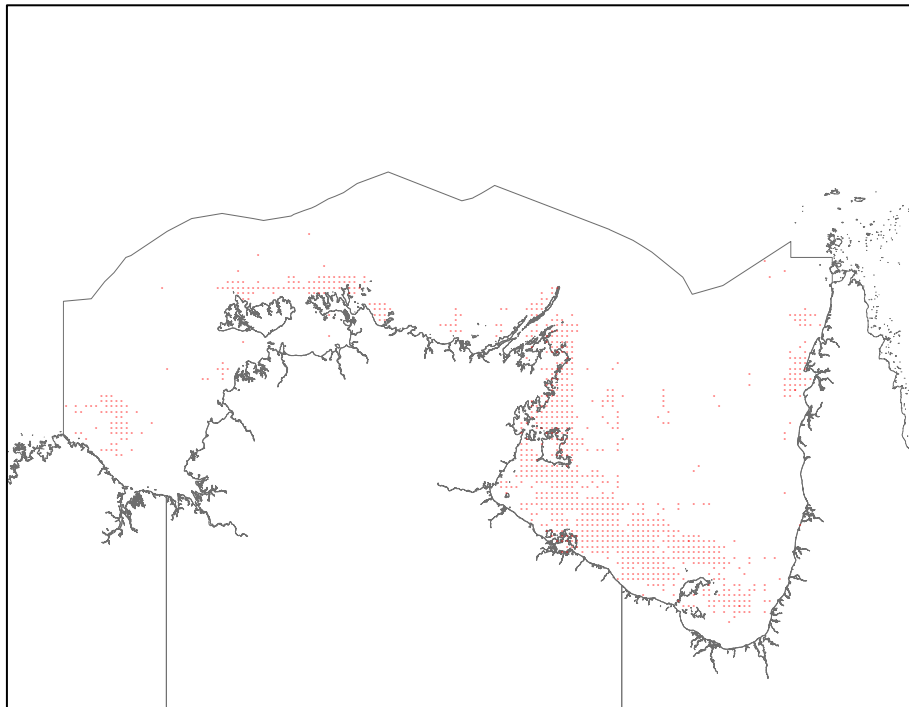


Figure 15. Map of the Northern Prawn Fishery showing the grids (6 * 6nm) that were fished in any year between 1999 and 2003 inclusive.

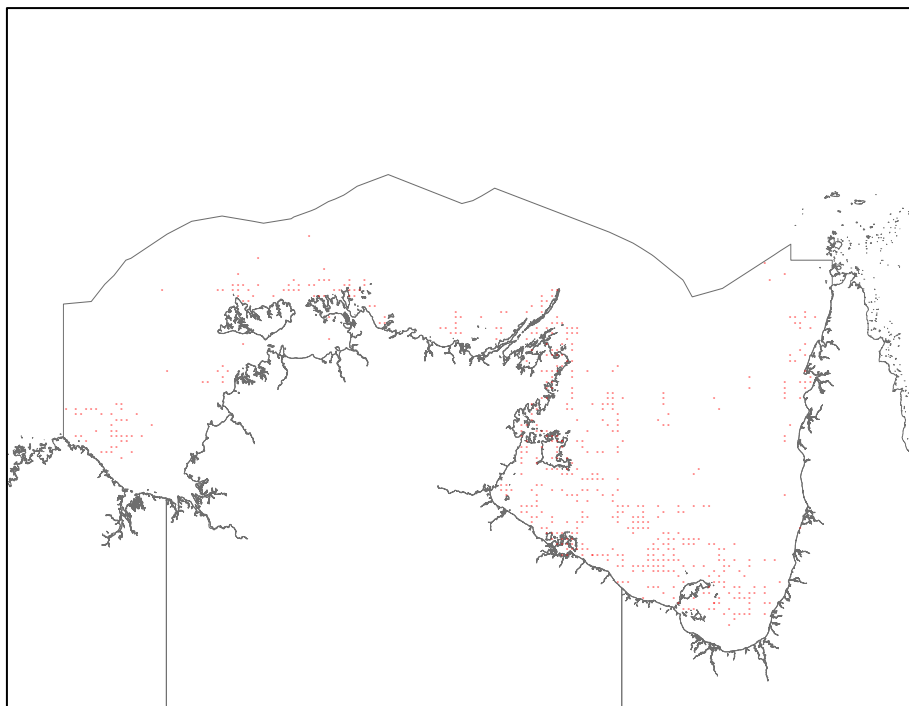


Figure 16. Map of the Northern Prawn Fishery showing the grids (6 * 6nm) that were fished for more than 5 days in any one year between 1999 and 2003 inclusive. Around 56% of grids were deemed “low effort” under these criteria.

Objective 1: Design of monitoring program

A desktop study that examines the sensitivity of the risk assessment process to changes in fished distributions, could potentially provide a clearer picture of the benefits or otherwise of such a strategy. Using the hypothetical example above, the percent of fished grids is reduced by around 56%. The current suite of quantitative risk assessments developed during this project (Section 5.4.1) rely on species distributional data as one of the important factors in determining whether a species is at risk. Repeating the risk assessments using the reduced set of fished grids would provide an interesting test of a scenario that intuitively should reduce sustainability concerns for many species.

It is beyond the scope of this Section to do more than highlight the need for innovative strategies to solve monitoring feasibility issues for rarely caught bycatch species. A wide-ranging consultative process is needed when considering any alternative management strategy. This process will require the full commitment from all fishery stakeholders, particularly the fishers and conservationists. A wide range of issues need to be addressed, including interactions with the marine protected areas planning process in Northern Australia, the potential need for future monitoring to demonstrate the links between spatial closures, and the subsequent rebuilding populations of bycatch species and in particular, sessile benthic fauna and their associated communities.

Conclusions

Alternative management strategies appear to be the only way to cost-effectively demonstrate sustainability for species too rare to detect changes with monitoring. We recommend that a committee of industry, scientific members and other stakeholders, such as the bycatch sub-committee, examine the range and usefulness of alternate management strategies for this purpose.

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5.2.5 Providing annual estimates of total bycatch: the issues and the reality

D. Heales

Project Objective

“To design, trial and implement an integrated long-term bycatch monitoring program; that addresses (i) total amount of bycatch, (ii) protected species and (iii) high risk species in the most cost-effective manner possible using the NPF as an example”.

Abstract

Estimates of total bycatch are indicators of fishery impact on the total bycatch community and can be made by all methods available to the bycatch monitoring program in the Northern Prawn Fishery (NPF). Daily estimates of total bycatch from individual vessels have been collected during this project by Crew Member Observers (CMOs) and scientific observers, and this process will continue in the monitoring program currently being implemented. Models using these data based on fleet effort patterns are needed to assess fishery impacts on bycatch in the future. Research proposals for the construction of these models are currently under consideration. The recent response by AFMA to a Commonwealth ministerial directive (December, 2005) requests that fisheries reduce their discards by half by 2008. This has placed further pressure on the need for total bycatch estimates for the Northern Prawn Fishery (NPF).

However, in the absence of species composition data, interpreting trends in total bycatch may not be informative due to the conflicting interpretations possible about both perceived declines and increases. Furthermore, the long-term monitoring program recommended in this Report (Section 6) is focused on determining the sustainability of individual species in the bycatch through quantitative risk assessments, supported by targeted monitoring of species identified as ‘at risk’. Most bycatch species are rarely caught in trawls and contribute little to the total bycatch weight in the average trawl. Consequently, monitoring trends in total bycatch are unlikely to contribute to solving sustainability issues for individual species, whilst the potential exists for many rarely-caught species to incur unsustainable mortality, which may go undetected.

Introduction

Annual estimates of fishery impacts on bycatch are provided by many world trawl fisheries (Kenelly et al., 1997; Ortiz et al., 2000; Ye et al., 2000; Diamond, 2003; Borges et al., 2005a, 2005b). Total bycatch estimates are useful for providing broad estimates of total fishery discards, and at a fishery level, to compare ratios of target species to discards (Alverson *et al.*, 1994). These estimates may have more importance for fisheries with a small number of bycatch species, where the estimated impact may be allocated with some confidence to a species or family. The EPBC Act 1999 implies that all species are to be treated equally, in the need to conserve them and in demonstrating their long-term sustainability, and we have used this interpretation throughout this report. In Australia's Northern Prawn Fishery (NPF) where the bycatch consists of hundreds of mostly small bycatch species (<30 cm length) (Stobutzki et al., 2001), total bycatch estimates will be unable to provide direction in resolving sustainability issues at the level of individual bycatch species.

The project steering committee originally agreed to take a low priority approach to the total bycatch issue considering the doubts about its utility as an indicator of sustainability. It was

Objective 1: Design of monitoring program

also agreed that the project objective should be separated into two objectives 1) measuring the fisheries impact by estimating total bycatch and 2) interpreting trends in total bycatch. To achieve the first objective, we commenced a program of using the platforms provided by the CMOs and scientific observer to collect data needed for annually estimating total bycatch.

The Ministerial Direction to AFMA in December 2005 further raised the priority of providing annual total bycatch estimates by requiring a halving in discards for all Commonwealth managed Fisheries. AFMA responded to the Ministerial directive by promising that the NPF would reduce bycatch by 50% by the year 2008, based on the total bycatch baseline in 2005. The only estimate of total bycatch of the NPF was 33,000 tonnes (Pender et al., 1992), when the fleet size and effort patterns were markedly different from that of 2005. Consequently, there is no current baseline estimate of total bycatch upon which reductions can be gauged. This data gap placed increased pressure on activating a modelling process using recent data, and the necessary parallel collection of bycatch estimates to populate the models.

The combined pressure has led to the development of a funding proposal (as yet unfunded) to build the required models and provide a baseline estimate as soon as possible, and to predict total bycatch in future years under a range of different effort patterns including the proposed Individual Total Quota systems, and other possible changes to gear configurations.

In this section we report on the progress of the data collection necessary for providing future annual total bycatch estimates for the NPF. We also report on the usefulness of interpreting trends in annual total bycatch as an indicator of both fishery and individual species sustainability.

Methods

Sub-objective 1: Measuring the fisheries impact by estimating total bycatch

Data collection

We defined the minimal sampling unit as being the total bycatch caught by all nets (except 'Try-nets') during one full (24 hr) days fishing, irrespective of regional and daylight closures. We requested both CMOs and scientific observers to record catch estimates separately for every trawl of the fishing day where possible.

The methodology used to measure total bycatch ranges from estimates by eye, to estimates made by counting leveled baskets of bycatch having a known average weight, or by physically weighing each basket.

In practical terms, estimates of the total bycatch per trawl are made based on estimating visually the weight of the codend (based on its size), of one of the two nets (twin-rigged vessels) just before the bag is spilled. The catch of prawns and other byproduct is deducted from the total and the resulting weight, in kilograms, is then doubled (i.e. in twin-rigged vessels) to account for the two nets. This assumes that, in the long-term, TED affected catches (due to temporary blockages) are equally likely in either net. The accuracy and precision of this method is heavily reliant on the past experience of the person in visually estimating the weight in codends.

We also requested both CMOs and scientific observers to calibrate their estimates, where possible, over a range of catch sizes. We expected large errors would be incorporated into the data due to the visual estimation method, particularly when estimating large catches (e.g.

Objective 1: Design of monitoring program

between 500-1000 kg). The process of comparing visual estimates with lug-basket counts is complicated because most of the twin-rigged NPF vessels have seawater hoppers into which both codends are usually spilled in quick succession. Calibrations require that the catch from one of the two nets be first visually estimated for weight (size) and kept separate from the catch from the other net. Furthermore, in order to lug basket the entire catch from one of the two codends, the sorting conveyor needs to be disconnected to allow the catch to be collected in lug baskets. This process takes time away from the usual (paid) duties of the CMOs and requires a reasonably fit CMO to undertake it. For scientific observers this task is still complicated but they have the time to complete it without compromising their other duties. It becomes much harder when the catch in the codend to be weighed via lug baskets weighs around 1000kg, or approx 25-30 baskets full.

We also considered the usefulness of requesting estimates of total bycatch from the fishery logbooks, either compulsory or voluntary.

Sub-objective 2: Interpreting trends in total bycatch

Given the implications of the EPBC Act to conserve all species (e.g. all bycatch species) equally, we examined whether interpreting annual trends in total bycatch could provide insight into sustainability issues at a community level as well as at individual species level. We canvassed expert opinion as to the range of interpretations surrounding the detection of both declines and increases in annual total bycatch. In order to detect the ability of total bycatch to reflect species sustainability, we examined the contribution that individual species make to the total bycatch weight in the average trawl. We analysed a set of 24 trawl catches (from both commercial and research vessels in the Gulf of Carpentaria) that had been completely sorted. For each trawl, we ranked the contributions of each species to the total catch weight and calculated the percentage contribution for each species.

Results

Sub-objective 1

CMOs provided 52 estimates of total bycatch for the full fishing day and a further 457 estimates of total bycatch from either one or more trawls in the fishing day. Scientific observers provided 16 estimates of bycatch from fullfishing days and a further 51 estimates of either one or more trawls in the fishing day. A total of 54 calibration comparisons were made during the study with a highest weight of 450 kg.

Figure 17 indicates that CMOs usually make reasonably good estimates of the total bycatch weight, although there is a tendency for most to be overestimated.

Sub-objective 2

Expert opinion assessments found that interpreting trends in annual bycatch was confounded by many factors. A downward trend can indicate two completely opposite outcomes for bycatch species; either the dominant species are declining (poor outcome) or bycatch reduction devices are working well, and therefore less bycatch is caught (good outcome). Similarly an upward trend is indicative of catching more bycatch (poor outcome) or alternatively, bycatch populations are on the increase (good outcome). Consequently the value of annual estimates of total bycatch is useful mainly for political value. Other factors

Objective 1: Design of monitoring program

that have the potential to effect trends in annual bycatch estimates are changes in species distributions due to other factors such as climate change or changes in spatial or temporal fishing effort patterns.

In the average trawl in the Gulf of Carpentaria, we found that around 75% of species contributed as little as 10% of total bycatch weight, and around 85% of species contributed around 20% (Figure 18).

Discussion

The primary objective of estimating annual total bycatch is presently being achieved through an on-going process of data collection by CMOs and scientific observers. The first annual estimate will be available by June 2008 and reported on in the 2008 calendar year, and continue indefinitely in the long-term monitoring program. The accuracy and precision of estimates is expected to vary widely and quantifying estimation errors will continue by comparing catches weighed in lug baskets with the initial visual estimates. We expect errors to be high initially, especially in estimating the weights of the larger codends. But increasing the number of calibrations will improve the estimates by scientific observers. Whether CMOs can improve their accuracy is unknown in the absence of more comprehensive calibration data.

The question of total bycatch estimates being provided in logbooks was canvassed at a NORMAC meeting in July, 2006. The unofficial viewpoint of industry was that the estimates should not be compulsory in logbooks. Based on that view it is likely that a request for voluntary estimates via the logbook system will be pursued in the near future. However, we recognise that a further change in culture is needed in order to increase the useful collection of data by this method.

The development of total bycatch models is critical to providing a baseline against which the projected 50% reduction of bycatch by 2008 (Ministerial Direction 2005) may be measured. There have been few estimates of total bycatch from the NPF (see Pender *et al.*, 1992), and a project proposal seeking funding to develop suitable models has been submitted.

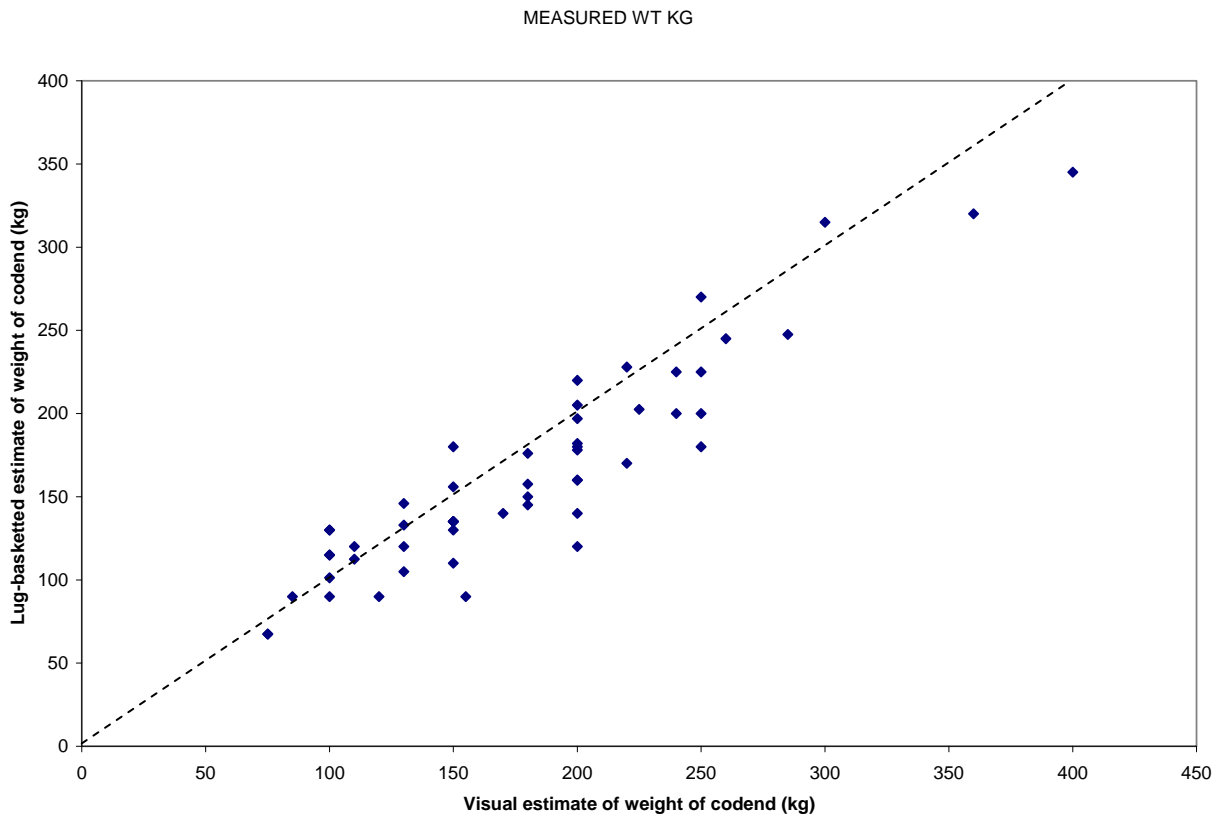


Figure 17. Comparison of visually estimated codend weights with estimate of codend weights via lug baskets. Data were collected from Crew Member Observers 2003-2004. Dashed line indicates perfect correlations between estimated and weighed bycatch estimates

The value of annual estimates of total bycatch, in the absence of parallel species composition data, is unclear because a lower annual total bycatch estimate can be an indicator of both favourable and poor outcomes, with a similar equivocal outcome for a higher estimate. Furthermore, a large number of species are caught rarely and contribute little to the total bycatch weight, a common occurrence in prawn trawl fisheries with numerous and diverse bycatch is (Stobutzki et al., 2001; Heales et al., 2003). This disproportionate contribution highlights that total bycatch trends are insensitive to sustainability issues for individual species because large numbers of mostly rarely species could conceivably become extinct, yet be undetected in catches. Sustainability issues at this level are best addressed using quantitative ecological risk assessments, as described in Section 5.4, and targeted monitoring be undertaken on individual species identified as ‘at risk’.

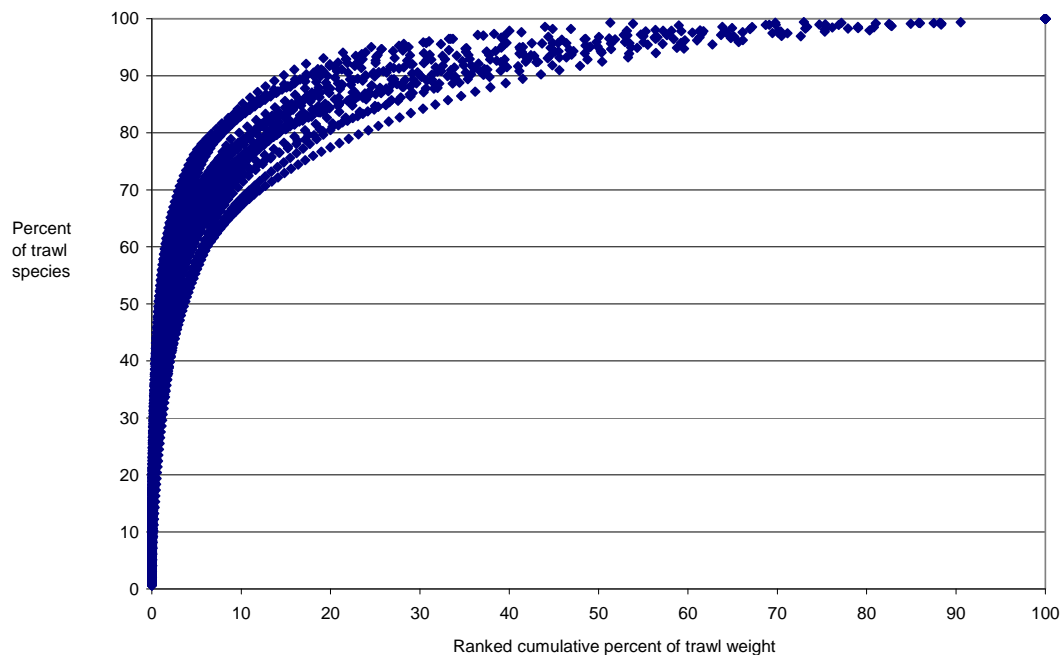


Figure 18. The contribution of individual bycatch taxa to the total weight of bycatch in an average NPF codend. The weight contributions by individual taxa are ranked from lowest to highest within each of 24 completely sorted trawl catches, then summed cumulatively as a percent.

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Ye, Y., Alsaffar, A. H., Mohammed, H.M.A., 2000. Bycatch and discards of the Kuwait shrimp fishery. *Fish. Res.* 45, 9-19.

5.2.6 Bycatch monitoring program - Implementation, effort and cost scenarios

D. Brewer, D. Heales and S. Griffiths

Section 5.3 describes a collaborative model between AFMA, CSIRO and industry for implementing a long-term bycatch monitoring program (BMP) for the NPF. This model includes annual data collection from both the ‘banana’ (common and red-legged banana prawn) and ‘tiger’ (tiger, endeavour and red-legged banana prawns) sub-fisheries, using a combination of methods, and annual delivery of a bycatch sustainability report. The format and recommended BMP are detailed below.

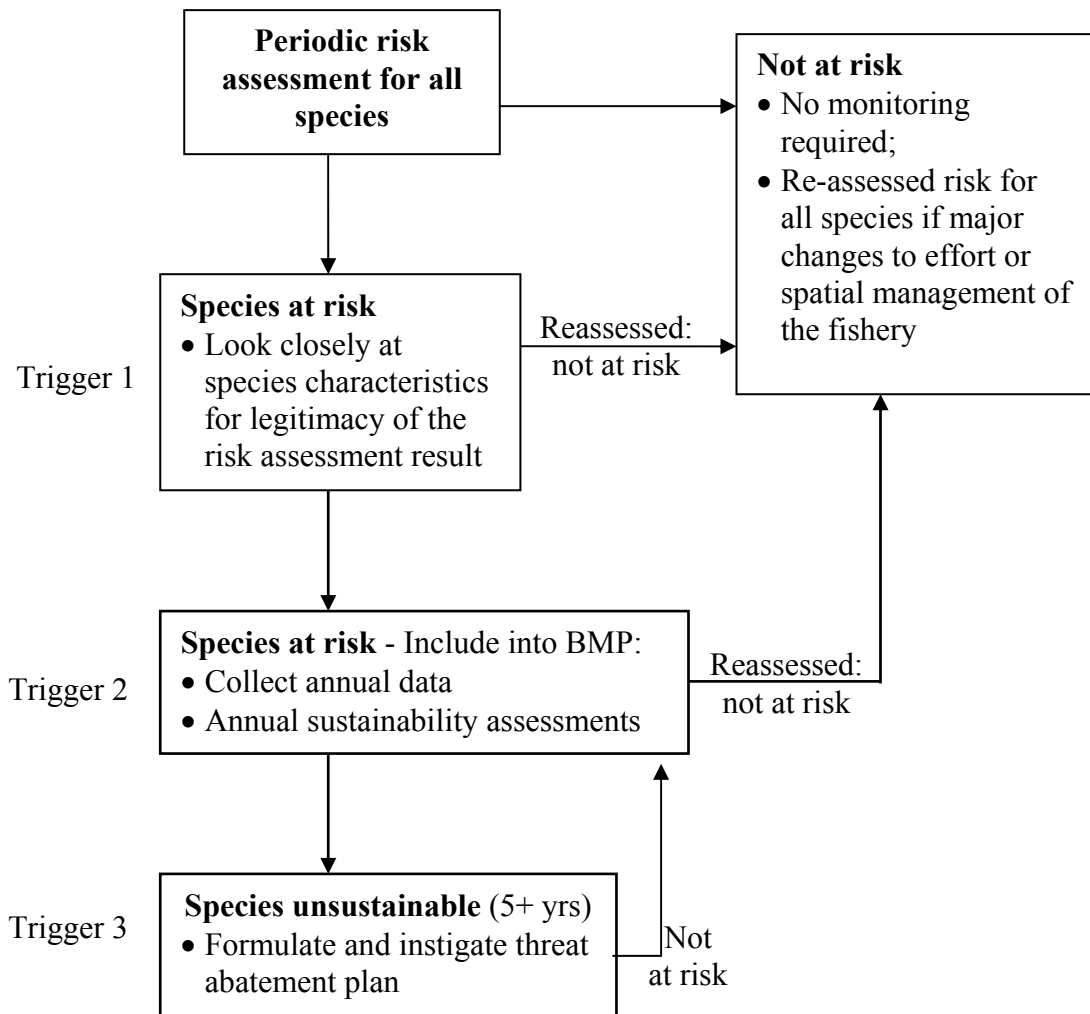
Decision process for monitoring and assessing bycatch sustainability

Figure 19 shows the decision process recommended for assessing and demonstrating sustainability of NPF bycatch using a quantitative risk assessment in conjunction with a BMP. Species at risk from NPF fishing activity are included in the BMP along with listed Threatened, Endangered and Protected (TEP) species. The BMP will then use a medium and long-term data set to collect additional information to determine whether (i) there is ongoing risk (= remain in the monitoring program); (ii) the risk is not real or removed (= cease monitoring); or (iii) the impact is unsustainable. In the latter case the fishery should formulate and instigate a specific threat abatement plan to remove this risk (e.g. use a specific bycatch reduction device to reduce its catch and improve survival). The risk assessment should be repeated periodically (e.g. five-yearly) to incorporate any new data and provide up-to-date assessment of risk. The remaining many hundreds of species need only be re-assessed if there are major changes to effort or spatial management of the fishery.

Recommended bycatch monitoring program

Northern Prawn Fishery bycatch contains a wide variety of species groups, many of which are rare (Stobutzki et al., 2001; Heales et al., 2003). The species of concern in the fishery also include a high proportion of rare species including sea turtles, sea snakes, syngnathids, sawfish, sharks, rays, small fish and invertebrates (Table 7, Section 5.4). These require relatively high samples sizes to detect change in catch rates over time in order to assess the sustainability of populations (Sections 5.2.2 and 5.2.3). The information collected in this study provides the basis for the design of a cost-effective bycatch monitoring program that can take advantage of the high sample collection ability and lower cost of fishery-dependent methods with the higher accuracy and acceptance of fishery-independent methods (Table 8).

Figure 19. Decision diagram showing the process recommended for assessing and demonstrating sustainability of NPF bycatch using a quantitative risk assessment and Bycatch Monitoring Program (BMP).



Objective 1: Design of monitoring program

Table 7: List of species included in the Bycatch Monitoring Program due to their being listed under Australia's EPBC Act as Threatened, Endangered and Protected (TEP), or considered 'at-risk' by the new 'SAFE' risk assessment (Section 5.4). See photos in Figure 20.

Scientific name	Common name
TEP species	
Cheloniidae	Sea turtles
1. <i>Caretta caretta</i>	Loggerhead turtle
2. <i>Chelonia mydas</i>	Green turtle
3. <i>Eretmochelys imbricata</i>	Hawksbill turtle
4. <i>Lepidochelys olivacea</i>	Olive ridley turtle
5. <i>Natator depressus</i>	Flatback turtle
6. <i>Dermochelys coriacea</i>	Leathery turtle
Hydrophiidae	Sea snakes
1. <i>Acalyptophis peronii</i>	Horned seasnake
2. <i>Aipysurus eydouxii</i>	Stagger-banded seasnake
3. <i>Aipysurus laevis</i>	Golden seasnake
4. <i>Astrotia stokesii</i>	Stokes' seasnake
5. <i>Disteira kingii</i>	Spectacled seasnake
6. <i>Disteira major</i>	Olive-headed seasnake
7. <i>Hydrophis elegans</i>	Elegant seasnake
8. <i>Hydrophis mcdowelli</i>	Small-headed seasnake
9. <i>Hydrophis ornatus</i>	Ornate seasnake
10. <i>Hydrophis pacificus</i>	Large-headed seasnake
11. <i>Lapemis hardwickii</i>	Spine-bellied seasnake
Syngnathidae	Pipefishes
Pristidae	Sawfishes
1. <i>Anoxypristis cuspidata</i>	Narrow sawfish
2. <i>Pristis clavata</i>	Dwarf sawfish
3. <i>Pristis zijsron</i>	Green sawfish
'At risk' species	
Other elasmobranchs	Other sharks and rays
1. <i>Orectolobus ornatus</i>	Banded wobbegong
2. <i>Taeniura meyeni</i>	Blotched fantail ray
3. <i>Urogymnus asperrimus</i>	Porcupine ray
Teleostei	Teleost fishes
1. <i>Dendrochirus brachypterus</i>	Dwarf lionfish
2. <i>Scorpaenopsis venosa</i>	Raggy scorpionfish

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Table 8 describes a combination of sampling methods that can provide a cost-effective bycatch data collection program. It relies on using the sampling power of the fishing fleet to collect adequate sample sizes for a range of TEP and ‘at-risk’ species (Table 7, Figure 20. Species included in the NPF Bycatch Monitoring Program, and as listed in Table 7.). It also provides a high level of broader stakeholder acceptance by including annual training for crew member observers (CMOs) and validation of data for all species groups using a combination of methods including scientific observers and fishery-independent surveys.

Table 8. Recommended cost-effective Bycatch Monitoring Program for the Northern Prawn Fishery; indicating which sampling method(s) collect the primary (1= majority of data), secondary (2) and minor (m) data sets. V indicates where the data will be used for validation of other methods. The cost ranking indicates the relative cost to collect bycatch data from 1 (cheapest) to 4 (most expensive).

	Logbooks	CMO	Scientific observers	Fishery - independent surveys
Cost ranking	1	2	3	4
Bycatch group				
Sea turtles	1	2V	2V	m
Sea snakes	-	1	2V	mV
Syngnathids	-	1*	2V	m
Sawfish	1*	1V	2V	mV
Other elasmobranchs	-	1	2V	m
Total bycatch estimates	1*	1V	2V	2V
Subsample collection	-	-	1	1

* = expected to successfully collect these data in the near future

BMP sampling capability and effort scenarios

Figure 14 present the effort levels required to detect a 20% or 50% decline over five and ten year time frames for all species to be included in the BMP (based on 80% power and 5% significance – Section 5.2.3). The total number of trawls that the entire fleet is capable of monitoring is roughly about 33,600 trawls (70 vessels x 120 days x 4 trawls), based roughly on 2006 effort levels. This represents the logbook data collection capability.

CMOs and scientific observers can combine to collect acceptable and validated data for sea snakes, syngnathids, sawfish, and selected elasmobranchs and fish. However, the level of effort will depend on the number of each engaged by the BMP. Furthermore, our research has shown that it is only feasible for CMOs to collect data from one of the two nets, and about 80% of trawls during the season (Section 5.2.1, Table 9). Scientific observers will collect this data from about 90%-100% of trawls, depending on the species. By using the lower, conservative end of these effort data, we can represent the number of nets from which we can reasonably expect to collect bycatch data. Consequently, we expect one CMO could provide data from 384 nets per year and one scientific observer, data from 864 nets per year. These sampling effort levels are shown in Table 9 for a range of CMO and scientific observer recruitment scenarios.

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The BMP sampling capability and scenarios for each of the key bycatch groups are described below.

Sea turtles

Sea turtle species data can be usefully collected by logbooks although require validation by CMO and, to a lesser extent, scientific observer data (Table 8). Due to the low numbers caught by the fishery since the introduction of TEDs in 2000 (Brewer et al., 2006), and the cosmopolitan distribution of most sea turtle species, the assessment of sustainability should be primarily made from other data-rich monitoring programs (e.g. counts of nesting females). These are conducted by national and international agencies (see review by Limpus and Chatto, 2004) and the data from the NPF BMP should be used in conjunction with these other sources.

Logbook data can detect declines for two species of sea turtles (*Natator depressus*, and *Lepidochelys olivacea*) but not for the remaining three species (Table 7, Figure 20 a–f). The flatback sea turtle, *Natator depressus*, is endemic to the northern Australia region and this puts a strong onus on the NPF to assess changes in catches of this species.

Sea snakes

Sea snake species (Figure 20 g–q) can be adequately assessed using data collected by CMOs and validated by scientific observers. Although these species are difficult to identify and some are rare, the sampling power of these methods will provide the collection of adequate data for most to enable a sustainability assessment over time (Table 8).

Detecting declines in sea snake species using CMOs and scientific observers is dependent on the observer effort level used in the industry. For example, ten CMOs and one scientific observer (2,350 trawls) can detect a 50% decline over five years for nine of the eleven species. However, it would require 15 CMOs and three scientific observers (4,180 trawls) to detect declines in ten of the species, and more than double that sampling effort to detect declines in all eleven species.

Sea snakes are one of the few potentially vulnerable bycatch species groups that are not impacted by other fisheries in the NPF managed area. This puts a strong onus on the NPF to demonstrate sustainability for these species. A current CSIRO project (FRDC Project 2005/051) is building on recent studies of sea snakes (Milton et al., 2001; Stobutzki et al., 2001; Brewer et al., 2006) to produce quantitative population and risk assessments to assess the status of sea snake populations in the NPF.

Syngnathids

Syngnathids are difficult to sample as they are rare in catches, small, and are difficult to find amongst the small bycatch (Figure 20 r). Although none of the fishery-dependent methods trialled the collection of syngnathid data during the 2003 and 2004 seasons, it has since been included in the CMO bycatch data collection program. With validation from increased observer coverage there is a reasonable expectation that the sustainability of this group can be adequately assessed as the NPF bycatch monitoring program continues and is fine-tuned in subsequent years (Table 8). However, scientific observers and fishery-independent data collection indicate that large numbers of trawls will be required to detect population declines in these species. The BMP will collect more information on these species in the next few years to provide a better understanding of the species involved, their catch rates, and whether their population status can feasibly be assessed in the program.

Sawfish

Sawfish are not listed in the TEP species list or evaluated to be ‘at risk’ by the SAFE ecological risk assessment. However, they are a highly vulnerable group due to their life history characteristics (Simpfendorfer, 2000), and are caught by several different fisheries in northern Australia. They have been removed through fishing in most other regions of the world, including neighbouring countries to Australia. A cumulative ecological risk assessment is needed for these species to assess the full extent of impact from all fisheries throughout their range. In the meantime, a precautionary approach is essential to ensure the viability of sawfish populations in Australian waters.

Sawfish are easily identified by both fishers and scientists, but two of the three species caught by the fishery – Green sawfish (*Pristis zijsron*) and Dwarf sawfish (*Pristis clavata*) (Table 7, Figure 20 s-u) – are rarely encountered and require large sample sizes to detect population change over time. Sawfish will be included in fishery logbooks to enhance the level of data collection on this group and add to the data collection by other methods (Table 8). The CMO and scientific observer data, in particular, will be used to validate sawfish data from logbooks.

Detecting population declines in the most commonly caught sawfish (*Anoxypristis cuspidata*) is feasible using 10 CMOs. However, the rarer species require large numbers of trawls. Their inclusion in fishery logbooks from 2007 has the potential to enhance the program’s ability to detect population declines in these rarer species and will be assessed in more detail as more data is collected during the BMP.

‘At risk’ elasmobranchs

Other elasmobranchs identified by the SAFE method as being ‘at risk’ (Section 5.4.4) are relatively rare and will have most data collected by CMOs with validation by scientific observers if possible (Figure 20 v-w, Table 8). These species can also be validated using photographs and/or actual specimens returned to the laboratory for identification.

Power calculations on the three elasmobranchs that are ‘at risk’ suggest that the BMPs ability to detect population declines is highly dependent on the observer effort level used in the industry. Effort levels required in these species vary from 4,150 trawls (10 CMOs and one scientific observer) to detect a 50% decline in *Urogymnus asperimus* and *Taeniura meyeni* over ten years, to 15,644 trawls (data collection by half of the fleet) to detect a 25% decline in the same species over five years and a 50% decline for *Orectolobus ornatus* over the same period (Table 7). As with other species groups, a management decision will be required to decide on a combination of the level of decline and the time frame required in which to assess trends in catches for these species.

‘At risk’ teleosts

Teleosts identified by the SAFE method as being ‘at risk’ (Section 5.4.5) are small, and need to be sorted from the catch. They are relatively rare and will have most data collected by CMOs with validation by scientific observers if possible (Table 8). These species should be validated by sending specimens to the laboratory for identification.

Declines in the two ‘at-risk’ teleost fish species – *Dendrochirus brachypterus* and *Scorpaenopsis venosa* (Table 7, Figure 20 y-z) – are also dependent on the observer effort level used. A 25% decline in both species can be detected by 15 CMOs and five scientific

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observers. However, if a 50% decline is satisfactory over either a five or 10 year time frame, then 10 CMOs and one scientific observer should be adequate.

Bycatch communities

Assessing the composition and structure of species communities is a key indicator for ecosystem-based fisheries management (Rochet and Trenkel, 2003; Fulton et al., 2004a, 2004b; Hall and Mainprize, 2004). There is also evidence that a smaller suite of species can act as indicators of community health for the wider community (e.g. Zacharias and Roff, 2001). The bycatch taken by the NPF comprises many hundreds of species from a wide variety of taxonomic groups (Stobutzki et al., 2001). Their dominance in trawl catches and high diversity warrant ongoing assessment to determine whether trawl impacts change the structure of these demersal communities (see Sainsbury et al., 1987), but without necessarily putting individual species at risk. Changing the structure of communities, such as the demersal fish species caught as NPF bycatch is not desirable (see Longhurst, 2006) and an ongoing assessment of community composition is necessary to understand the long-term fishery impact on these demersal communities. This is especially true given that these communities may have already been altered during the fishery's history and present-day ecological risk assessments are based on recent data and not from a hindcasted pre-fished state before inception of the fishery (Section 5.4). The monitoring of a relatively small number of individual species is also unable to provide information to make this assessment.

Assessment of the NPF bycatch assemblage – comprised of hundreds of small fish and invertebrates – can only be made on data collected by scientific observers or fishery-independent surveys. The other methods collected an unacceptably high proportion of inaccurate data (Section 5.2.1). Both methods collected reliable and accurate subsamples, although the fishery-independent sampling used a more robust sample design for assessing changes from year to year.

Fishery-independent bycatch surveys have an advantage over fishery-dependent platforms in that they can control and minimise spatial, diel and lunar influences from year to year, providing more precise assessments (King, 1999). These surveys could piggyback on the current NPF prawn monitoring mid-year surveys and provide certainty in data collection from year to year. They are also the only method that can provide an option for collecting control data (samples outside the high effort areas) to interpret whether any changes detected in species composition and structure are due to fishing impacts or some other source (e.g. climate change – see McFarlane et al., 2000).

Consideration should also be given to making periodic surveys without TEDs installed to assess the catch rates of a wide range of large species that are now excluded by TEDs. This has the power to demonstrate recovery of many vulnerable species including sharks and rays, which may have been depleted prior to the introduction of TEDs in 2000. Furthermore, the number of trawls required to adequately sample many of these species will be much lower, based on their increased catch rates by having TEDs removed.

Total bycatch discards

Assessing the total discarded weight of bycatch is also unable to provide an indication of any change in the bycatch assemblage structure. It is conceivable that many rarely caught species could unknowingly be driven to extinction if only total discards were monitored (Section 5.2.5). However, total bycatch discard estimates from catches will allow annual assessments of the total discards from the fishery (Section 5.2.5). This is a widely used indicator of fishery impact and can be collected by all methods available to the bycatch monitoring program. However, CMO's validated by scientific observers will be the main methods for these

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assessments in the immediate years. This data collection will be requested in logbooks (initially on a voluntary basis) from 2007 onwards to improve the fishery's total discarding assessments (Table 8). This should include awareness and data collection protocols and information. The logbook data will also require validation by CMO and scientific observers before its inclusion in models being developed to estimate total discards.

Effort and cost scenarios for the NPF BMP

Table 9 summarises the main cost options for the recommended bycatch monitoring program. It describes the AFMA costs for running the BMP, including placing CMO and scientific observers into the industry, and the CSIRO costs for processing bycatch sub-samples, conducting fishery-independent surveys, analysing bycatch data and producing an annual sustainability report for bycatch species. Five different AFMA costings are presented (based on different levels of observer sampling effort mainly for collecting data on TEP and 'at-risk' species) and seven CSIRO costings (based on different scenarios for collecting species composition data, including a basic program with no species composition component).

For some of the rarer TEP or 'at-risk' bycatch species there is insufficient sampling capability in the industry (maximum of about 33,600 trawls per year) to detect changes in their populations based on catch information. Or in the case of syngnathids, their taxonomy is too poorly known at this stage to include in this sampling effort scenario. In both these cases alternative management strategies should be considered in order to enhance the long-term sustainability of these species (see Section 5.2.4) and provide protection for demersal communities impacted by this fishery.

The recommended comprehensive BMP can deliver sustainability assessments for two species of sea turtles, all eleven species of sea snakes, one species of sawfish, all three species of 'at-risk' elasmobranch, both species of 'at-risk' teleost fish, species composition and structure and total bycatch discards. Invertebrate risk assessments will be complete in the near future and unseen impacts on habitats and benthic communities should also be addressed. However, the current scenario would cost an estimated \$885,000 in order to include fishery-independent surveys, with control regions included, in both the Gulf of Carpentaria (GoC) and Joseph Bonaparte Gulf (JBG).

Reduced cost scenarios can be selected from Table 9. For example, the scenario described above of 15 CMOs and three scientific observers will allow the detection of declines for two species of sea turtles, ten of eleven species of sea snakes, one species of sawfish, two of three species of at-risk elasmobranchs, and both species of at-risk teleost fish. This scenario costs about \$425,000 per year, but some of the selected TEP and at-risk species and includes no assessment of community composition and structure.

The costs of the basic program including data analyses, observer training and deployment and annual sustainability reports. The addition of the community assessment scenarios also include laboratory processing of samples and vessel charter. They range from the lowest cost option using scientific observers (no vessel charter), to other (Fishery Independent Survey) options that include the GoC region, JBG region and collection of control data at both regions. The GoC region has no chartered vessel component as it would piggyback on the current mid-year prawn monitoring survey. However, extra charter days are required for both the JBG region and inclusion of control data collection

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Responses to trends in bycatch data

Figure 19 describes a framework for the general approach to decision making. It describes how 'at-risk' species are included in the BMP and species either not initially 'at risk', or later demonstrated to be fished sustainably are not included or removed from the BMP, respectively. Species assessed as in irreversible decline (e.g. beyond a pre-defined limit reference point) will require the urgent formulation and instigation of a threat abatement plan.

The collection of bycatch data during the BMP will provide detailed information on trends in catches for each species included in the program. Changes in catches are surrogates for changes in population levels and management responses to these results are required.

These responses may include a series of biological reference points, such as u_{msm} and u_{crash} proposed in Section 5.4, triggers and management actions, although these have not been fully developed and validated for bycatch species where limited biological information is available. To this end, the BMP will participate in national Ecological Risk Assessment forums for guidance and consistency across fisheries on these issues.

Processes such as periodic risk assessments (SAFE) and sustainability assessments may flag changes in the needs of the monitoring program. It should be recognised that the ongoing program be able to incorporate changes such as the removal or addition of species potentially 'at risk' in the monitoring program and use of upgraded versions of the risk assessment technique. All BMP processes should be reviewed periodically and adapted to maximise the cost effectiveness of the program and the needs of the fishery management.

Other requirements of the Bycatch Monitoring Program

Assessing bioregions separately

The assessment of sustainability should be made separately in all bioregions encompassed by the NPF, as seen in the method used for the SAFE risk assessment model (Section 5.4). Studies of the fish (Blaber et al., 1994), epibenthic communities (Long et al., 1995) and sediments (Somers and Long, 1994) indicate that there at least are two main bioregions in the GoC – approximately defined as eastern and a western regions. This study (Section 5.5.2) and Ramm et al. (1990) described a third major bioregion in the JBG. Where possible these three bioregions should be represented individually in any assessments of individual species or community assessments of ongoing sustainability.

The effort and costs scenarios described above treat the NPF managed area as a single bioregion. Detecting declines or change in each bioregion separately would increase the effort and cost required by the program, and the repercussions of not taking this approach should be discussed in future management forums.

Collection of Abiotic data

Data collection for the BMP should include the collection of abiotic data on most sampling platforms to assist in the interpretation of detected changes. Both scientific observers and CMO could be provided with, and trained in the use of, data loggers that can be attached to trawl boards or nets and interrogated and recharged periodically throughout the fishing season.

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Table 9. Summary of annual effort and cost (x1000) of detecting declines in catches of key species (sea turtles, sea snakes, syngnathids, sawfish, and selected elasmobranchs and fish) relying on CMO and scientific observer (SO) coverage, based on a 5-year minimum time frame and power of 0.8. The number of trawls assumes the CMO and scientific observer reliability as reflected in the methods trial in 2003/04. FIS = Fishery-Independent Surveys; GoC = Gulf of Carpentaria; JBG = Joseph Bonaparte Gulf.

	10 CMO+ 1 SO (2,350 trawls)	10 CMO+ 3 SO (3,200 trawls)	15 CMO+ 3 SO (4,180 trawls)	10 CMO+ 5 SO (4,080 trawls)	15 CMO+ 5 SO (5,040 trawls)
TEP and at-risk species only	\$270	\$416	\$425 k	\$562	\$582
+ GoC SO species composition	\$332	\$478	\$487 k	\$624	\$644
+ GoC FIS	\$385	\$531	\$540 k	\$677	\$697
+ GoC FIS & controls	\$438	\$584	\$594 k	\$730	\$750
+ GoC and JBG species composition	\$418	\$564	\$573 k	\$710	\$730
+ GoC and JBG FIS	\$506	\$652	\$661 k	\$798	\$818
+ GoC and JBG FIS & controls	\$573	\$719	\$728 k	\$862	\$885

Figure 20. Species included in the NPF Bycatch Monitoring Program, and as listed in Table 7.

Threatened Endangered and Protected species

Sea turtles

a. *Caretta caretta*



b. *Chelonia mydas*



c. *Eretmochelys imbricata*



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d. *Lepidochelys olivacea*



e. *Natator depressus*



f. *Dermochelys coriacea*



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Sea snakes

g. *Acalyptophis peronii*



h. *Aipysurus eydouxii*



i. *Aipysurus laevis*

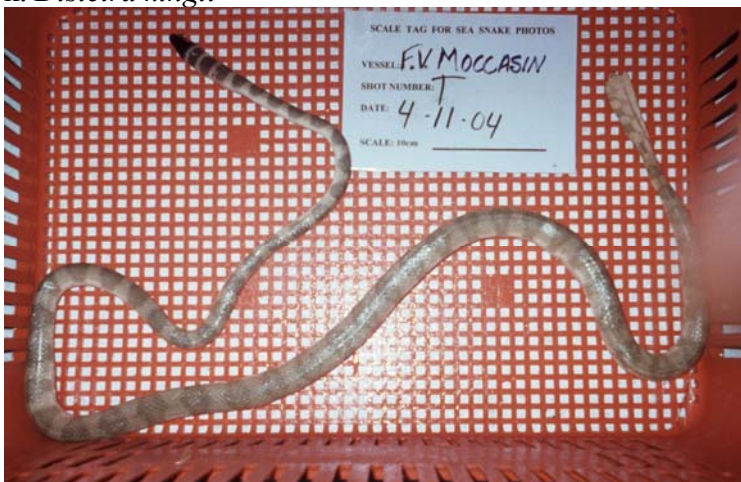


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j. *Astrotia stokesii*



k. *Disteira kingii*



l. *Disteira major*

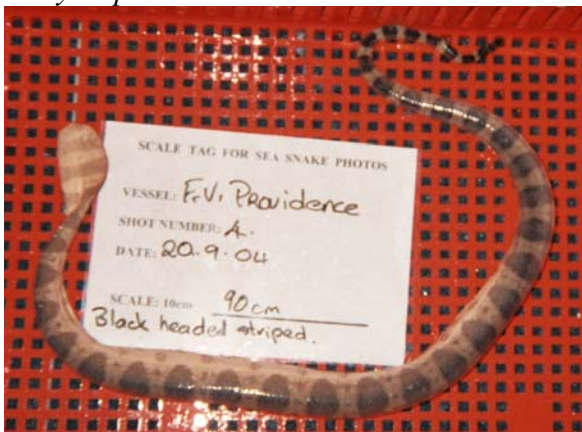


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m. *Hydrophis elegans*



n. *Hydrophis mcdowelli*



o. *Hydrophis ornatus*



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p. *Hydrophis pacificus*

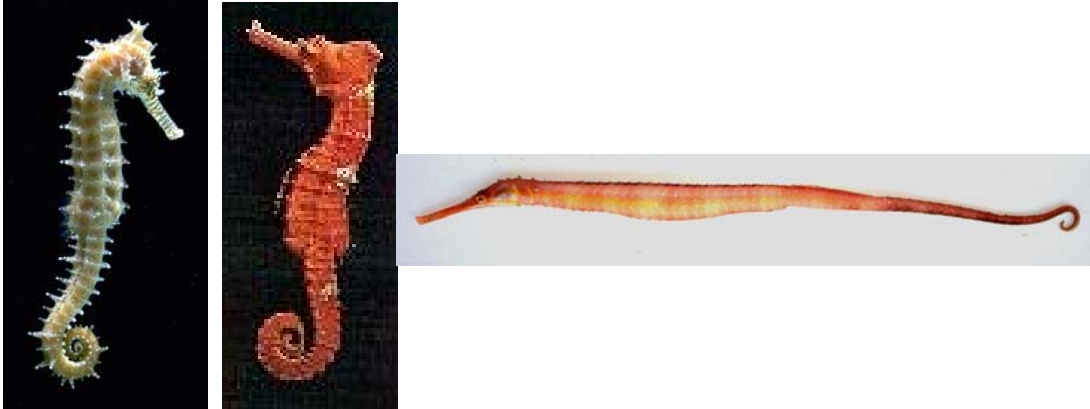


q. *Lapemis hardwickii*



Syngnathids

r. Syngnathidae sp.



Sawfish

s. *Anoxypristis cuspidata*



t. *Pristis clavata*



u. *Pristis zijsron*



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At risk species

Other elasmobranchs

v. *Orectolobus ornatus*



w. *Taeniura meyeri*



x. *Urogymnus asperimus*



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Teleosts

y. *Dendrochirus brachypterus*



z. *Scorpaenopsis venosa*



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5.3 Objective 2 – To transfer ownership, momentum and responsibility of ongoing monitoring to NORMAC and AFMA

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Abstract

Responsibility and ownership for the long-term monitoring program was gradually transferred to AFMA throughout the collaborative AFMA and CSIRO project. In order to maintain the momentum of the long-term monitoring program, a series of detailed protocols are being developed to ensure that corporate knowledge gained during the project on the best ways to manage the long-term monitoring objectives are clearly defined. Protocols that are critical to the continued successful running of both Crew Member Observer and scientific observer programs will be easily accessible to staff, particularly new staff, in order to ensure ownership and continuity of the program. Protocols defining the different roles for both AFMA and CSIRO clearly spell out individual departmental responsibilities in organising, maintaining and reporting on this long-term monitoring program.

Transfer of responsibility and ownership to NORMAC will occur with the presentation (in early December 2006) of the projected monitoring budget for the 2007 financial year. The response of NORMAC to the projected budget and work plan will provide future direction and funding for bycatch work in the NPF.

There is a need to ensure long-term funding for this program otherwise there may be reduced participation and enthusiasm of involved staff and a lower level of involvement by Industry in the Crew Members Observer program. AFMA will endeavour to secure continued funding for the training of Crew Member Observers and will allocate 40% of a full time employee to run this part of the program. This position will ensure regular contact with the participants to encourage them and to maintain interest and ensure long-term commitment. AFMA will also allocate funding for scientific observer coverage during both fishing seasons in the NPF.

5.3.1 Introduction

The development and implementation of the long-term bycatch monitoring program will play an important role in AFMA achieving its objectives under the NPF Bycatch Action Plan (BAP), the AFMA bycatch guidelines (a result of the 2005 Ministerial Direction) and its obligations under the strategic assessment of the Fishery by the Department of Environment and Heritage (DEH).

Strategy Three of the BAP is to monitor bycatch in the Fishery including protected and other identified potentially vulnerable species, and general bycatch. The data collected by logbooks, Crew Member Observers (CMOs), scientific observers and fishery-independent surveys will be used to address this strategy.

The strategic assessment of the Fishery in 2003 recommended that AFMA ensure that ongoing monitoring of selected, 'high risk' sea snakes, Pristidae (sawfishes) and Dasyatidae (rays) is undertaken. In addition, the development of the new suite of quantitative Ecological Risk Assessments has identified other bycatch species that may be 'at risk' from trawling, and

Objective 2: Transfer of monitoring program to NORMAC & AFMA

included them in the monitoring program. These species will be addressed through the planned data collections.

5.3.2 The project model of collaboration

From its inception, the project required that AFMA, CSIRO and Industry work collaboratively. Staff from both AFMA and CSIRO conducted pre-season port visits in April and July/August from 2003 - 2005 to the main ports for the Fishery (Brisbane, Cairns, and Darwin). The main objective of these visits was to recruit crew members to (a) participate in the CMO program and (b) to collect bycatch samples under the Requested Industry Collection scheme, and (c) recruit vessels to take scientific observers.

During the mid-season breaks in 2003, 2004 and 2006, AFMA and CSIRO organised training workshops for recruited CMOs. No training for CMOs was conducted in 2005 as per the project timetable. In 2003 the main aim of the workshop was to brief CMOs on the project and expected outcomes, train them in duties required for the coming seasons, and acquaint them with AFMA and CSIRO project staff.

The 2004 and 2006 workshops had similar aims, but also included reporting project results back to the CMOs, as well as seeking feedback from CMOs who had returned for their second or third workshop. The workshop logistics were organised by AFMA, with input from CSIRO. Staff from both departments made presentations to the group and were involved in the training components. The training included:

- Rationale for data and sample collection.
- Identification of the key species (sea turtles, sea snakes, and selected elasmobranchs)
- Safe handling of sharks, rays, sea snakes, and sea turtles
- Data recording requirements

5.3.3 Handover of duties to AFMA

The handover of the projects CMO and scientific observer components began halfway through the project and continued to the completion of the project. In the 2005/06 financial year AFMA incorporated \$30,000 into the Fishery budget to continue the recruitment and training of CMOs and for scientific observer coverage. In 2006, 50% of an AFMA full time employee (FTE) has been dedicated to organising CMO recruitment and training, assisting at training workshops and assisting CSIRO with the deployment of scientific observers.

AFMA staff also conducted pre-season port visits to brief skippers before leaving port, as well as some post-season debriefings. Their port duties also included liaising with CMOs and providing them with equipment and incentives (including fish identification books, t-shirts, bags, hats and vouchers).

5.3.4 Continuation and progression of BMP

Bycatch monitoring and bycatch reduction in general are high priorities for the NPF and in the 2006/07 financial year, AFMA has incorporated \$70,930 into the Fishery budget for CMO recruitment and training, and for scientific observer coverage.

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The Bycatch Monitoring Project steering committee agreed to a 'slow start' to the long-term bycatch monitoring program, due mainly to the financial implications that the increased observer coverage will have on the fishery. The scientific observer coverage has been increased from 62 days in 2005, to 90 days in 2006. Furthermore, the collection of bycatch data from the fishery-independent surveys has also been postponed until 2007, pending the approval of project recommendations (See Section 6) and the subsequent approval of the 2007 AFMA NPF Bycatch Monitoring Program budget.

5.3.5 General role clarification

Bycatch monitoring in the NPF will continue to be a collaborative approach between AFMA, CSIRO, and Industry, with AFMA taking the lead role. AFMA will be responsible for the deployment of scientific observers and recruitment of CMOs. Both AFMA and CSIRO staff will participate in the training of scientific observers and CMOs. .

AFMA will be responsible for the organisation and funding of annual mid-season CMO training workshops including the agenda, venue, presenters, activities and rewards, with input from CSIRO.

The training of scientific observers will initially be undertaken with the AFMA observer section in Canberra for 1 to 2 days. If necessary (e.g. for new NPF observers), this will be followed by a further training day with CSIRO staff at the Cleveland laboratories.

AFMA will be responsible for collating and entering the data from the scientific observers and CMOs into the AFMA database and will carry out preliminary analysis. The data will also be sent to CSIRO for a more detailed analysis of the sustainability of bycatch species and communities, and the delivery of reports to AFMA and NORMAC.

5.3.6 Importance of changing the AFMA and Industry culture and processes

To ensure consistent and successful long-term bycatch monitoring, it is important that AFMA develop and maintain documentation of the program for future staff. AFMA has a fairly high staff turnover. During this project, when new staff have taken responsibility for project objectives, there has been a lack of protocols that were critical for maintaining the CMO program, or for defining the duties of scientific observers. A set of documented processes has been developed and will be finalised by June 2007 to ensure consistency of these programs in future years.

The need to change the culture of industry in regards to bycatch is important and has already been achieved to some extent by this project. Many industry members now recognise the need to monitor and reduce bycatch. Through an improved understanding and participation in the project, some NPF skippers have taken it upon themselves to develop new ways of reducing bycatch through the development and more effective positioning of BRDs.

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5.3.7 Long-term Collaborative model between AFMA, CSIRO and Industry

a. Part of an overall AFMA/CSIRO collaboration

Currently CSIRO undertake fishery-independent prawn monitoring surveys prior to both fishing seasons. These are funded by the industry levy and AFMA. It is proposed that, in the future, the late winter survey trips will also include one to two additional staff to collect bycatch data (including TEP and high-risk species). The bycatch component of the prawn surveys may be under a separate research project and contract with AFMA. In essence, this additional work should ‘piggy-back’ on the existing monitoring surveys (See Section 6.2).

b. Unique collaboration model for bycatch monitoring

The collaboration between AFMA, CSIRO and Industry on this program is unique with no other AFMA funded projects having all parties playing a significant role in the programs development and execution. It will be imperative for all parties to work closely together to ensure the bycatch monitoring continues both efficiently and effectively.

To ensure the continued effectiveness of the monitoring program, it is recommended that there be six-monthly face-to-face meetings of AFMA and CSIRO staff to discuss progress, including data collection, other information needs, and any limitations staff are finding with the program. This meeting should include personnel from AFMA’s observer and data section to ensure collaboration between the department sections, as well as between the departments.

c. Data security

All data collected by both CMOs and scientific observers will be treated with the same confidentiality applied to logbook data, ie. commercial in confidence. The normal restrictions regarding the presentation and reporting of results also applies in that only grouped data of more than five vessels is allowed. Only AFMA and CSIRO staff will have access to the bycatch database. Requests from other organisations (eg DEH) for data, other than that reported on in the annual reporting process, will be forwarded to the NORMAC REC for vetting if necessary.

d. Role clarification

- i. Scientific observer management (AFMA) – scientific observers will be organised by the observer section with assistance from the NPF management section.
- ii. CMOs – will be organised by the NPF management section with assistance from scientific observers, mainly during training and field interactions.
- iii. Permits (AFMA) – and other permits (e.g. DEH) required for the collection of any specimens will be organised by AFMA’s NPF management section for CMOs, and by the Observer section for scientific observers. Both sections will co-ordinate and communicate when such permits are necessary. If CMOs or scientific observers are required to collect any specimens for other projects, (e.g. CSIRO research or requests from Universities) then any necessary permits must be arranged by the requesting organisation.
- iv. Data collection – the collection of bycatch data will be the combined responsibility of all program partners. Crew members participating in the CMO program will collect data on a range of selected species groups. Complementary data collection will be completed by staff on the prawn/bycatch monitoring surveys and by AFMA scientific observers (see Section

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- 6.2). Collection protocols will be developed by AFMA in consultation with CSIRO. These will be annually reviewed and updated by AFMA NPF management and observer section staff, and CSIRO.
- v. Data entry – the information collected by scientific observers will be forwarded to the AFMA observer section for entry into a Microsoft Access database. In early 2007 this will change and observer data will be entered into an Oracle database. CMO data will be entered by AFMA's data entry section into an Access database. Raw data sheets will be photocopied and sent to CSIRO by NPF management staff once the data has been entered. The current version of the entire database (includes all years), will be forwarded to CSIRO for analysis and storage.
 - vi. Data analyses – following data entry at AFMA, the information will be analysed by both AFMA and CSIRO staff for reporting. Data analyses techniques will be described in detail in the first 'Bycatch Sustainability Report' to AFMA and NORMAC in June 2008.
 - vii. Joint reporting – following data analyses CSIRO and AFMA will both be responsible for reporting to NORMAC and other stakeholders as required. This will be co-ordinated by AFMA and will consist of the following components:
 - i. Overall Bycatch Sustainability Report co-ordination - AFMA
 - ii. Bycatch Sustainability Report - CSIRO
 - iii. CMO recruitment and training and scientific observer coverage (including any issues identified by CMOs, scientific observers and the departments) - AFMA
 - iv. CMO and scientific observer catch summaries will be incorporated into the NPF data summary - AFMA
 - v. CMO review. AFMA and CSIRO will develop a list of questions to be rotated between CMOs each year to gather feedback from them on the program and their participation

Reporting to Industry will be required mid-year, to NORMAC, at their June meeting and at the CMO training workshop, also mid-year. The reporting will be on the previous banana and tiger seasons e.g. reporting in June 2007 will be for the 2006 banana and tiger prawn seasons. Reporting will include presentations given by both AFMA and CSIRO to the CMOs at the training workshop and a paper to NORMAC.

- viii. Feedback to stakeholders – Feedback will be given to stakeholders through correspondence, newsletters, pre and post-season port visits, articles in AFMA's magazine *Fishing Future* and the AFMA update on the departments' website. The feedback provided will be jointly developed by AFMA and CSIRO.

5.3.8 Contracts and funding

a. Definition of funding model with industry

1. Project budgeting - Funding for the CMO and scientific observer programs each year has been provided via the normal AFMA budgeting process. To date, this has occurred without the use of rigorous scientific analyses on the level of coverage required. This advice is necessary for the correct budgeting for these activities.
2. Future budgeting process - The normal process is that the AFMA manager budgets the appropriate amount of funds for the required coverage of CMO and scientific observers.

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CMOs are catered for under the line item of ‘travel and subsistence’ while the scientific observers are budgeted in the ‘observer’ line item.

3. The funding approval process - Approval for funding is negotiated through interactions with NORMAC, where draft budgets are reviewed by the NORMAC cost committee, before eventual approval by NORMAC. Draft annual budgets will be presented to NORMAC at the pre-season meeting (usually in April).

b. Funding model with CSIRO

1. Interaction and synergies with prawn monitoring project – It is intended that the annual mid-year prawn monitoring survey incorporate bycatch data collection as well. The funding for this increased workload will be sought through a separate research proposal presented to the NORMAC REC. The additional staff required for the survey will either be a CSIRO scientist, AFMA management officer or observer. Additional funds will be required for sea-going staff, processing bycatch subsamples, data analysis and reporting.
2. Timing and frequency of contract – It is preferred that the budget and subsequent contract development is multi-year. This will provide continuity for the work as well as surety that the work will be completed. A detailed work program will be developed and agreed to by NORMAC, similar to the fishery-independent survey and RAG model.

5.3.9 Risk management

An integral factor in the success of any long-term biodiversity monitoring program is the dedication and training of those involved and long-term funding for the program.

a. Risk of not maintaining consistency

A long-term, wide-area monitoring system requires several generations of staff being involved over time, and those involved with sampling will be widely dispersed and many will not receive direct benefits from the system (Watson & Novelly, 2004). If this is not well managed, it can lead to a lack of commitment by staff and participants.

As discussed previously, AFMA has a high level of staff turnover and to alleviate the risk of inconsistency in the program (due to new staff not being fully aware of the needs of the long-term monitoring program), AFMA has developed protocols for the program to ensure those taking over the monitoring program fully understand their responsibilities (see Section 5.3.10 and Attachment 1).

b. Risk of changes in funding levels

Imperative to the long-term funding of such a program is that institutional commitment needs to recognise explicitly that a long-term monitoring program will become core business, maintained into the future, unless a well-considered decision is made to discontinue the program following formal review. Further to this, long-term contracts will need to be negotiated to ensure the commitment is formalised, rather than just being an agreement between individuals or small groups of researchers with common interests (Watson & Novelly, 2004).

Watson and Novelly also state that *it can be argued that if there is not even a strong expectation of continued funding, the monitoring system should not be implemented in the first place.* Due to

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the economic hardships faced by many fisheries, it can be difficult for managers and researchers to secure funds for fisheries research and it can be particularly difficult to gain commitment to long-term funding if it is not seen as a priority for the fishery. AFMA is aware of the difficulties faced by researchers to acquire appropriate funding and will try to ensure ongoing funding for the long-term monitoring program.

c. Risk of losing Industry support

Watson and Novelly (2004) state that in order to maintain support for the monitoring program it is important to regularly report to stakeholders on progress. This approach has been successfully used in the NPF for many years (Dichmont et al., in press) and should continue to create demand for the system's outputs as significant change and improvement occur. Annual reports in consistent formats will also provide important archival material in subsequent years and ensure corporate memory.

To ensure successful continuation of the Bycatch Monitoring Program and collaboration between AFMA, CSIRO and Industry, AFMA will minimise the risk of losing industry support by providing ongoing information on results of the monitoring to owners, operators, and other interested parties.

5.3.10 Documentation of protocols (for new program staff)

a. Details of running CMO programs

It was found that due to staff shortages within AFMA in 2005 and new staff members, recruitment and the level of contact with CMOs reduced and this had a direct impact on the level of participation and the amount of data collected. Time constraints (e.g. due to sudden withdrawal of CMOs from the program) resulted in the recruitment of CMOs being undertaken in an ad-hoc manner. On occasions, this situation created slight disharmony between the Department and skippers, fleet managers and/or owners due to a lack of communication. Some skippers and fleet managers were unaware of their crew's involvement in the CMO program. Due to work commitments for the crew, and/or a lack of understanding of the program, they would not allow crew to continue CMO duties. To alleviate this possible problem in the future AFMA has set protocols to be followed for the recruitment of CMOs (see Attachment 1).

b. Details of running scientific observer programs

Prior to AFMA finalising the NPF budget, NPF Management, the AFMA Observer section, and CSIRO staff will meet to discuss minimum observer coverage required during the fishing seasons in each upcoming financial year. Once the budget has been finalised, the sections will meet again to discuss, develop and refine the scientific observer duties before each season (as duties may vary depending on the season and current research needs). The AFMA Observer section will be responsible for the recruitment, training and deployment of the scientific observers with assistance from NPF Management and CSIRO.

During pre-season port visits NPF Management will discuss with interested skippers and operators the possibility of them taking a scientific observer onboard. This information will then be given to the Observer section to assist in the deployment of the scientific observers. It will

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also help to ensure that scientific observers will be deployed to vessels whose skipper and crew are more likely to be helpful and supportive.

c. Fishery-independent surveys

The organisation and execution of the prawn monitoring surveys will be CSIRO's responsibility. This will involve close co-ordination with the NPF prawn monitoring project and a separate contract with AFMA within an overarching bycatch monitoring program.

Detailed protocols for staffing requirements, sample collection, sorting, analyses and reporting will be developed by CSIRO once a sustained level of funding has been agreed to by NORMAC.

References

- Dichmont, C.J., Loneragan, N.R., Brewer, D.T., Poiner, I.R., in press. Partnerships towards sustainable use of Australia's Northern Prawn Fishery. In: *Successes in Marine Resource Management*. McClanahan, T.R. and Castilla, J.C. (eds). Pew Foundation.
- Watson, I., Novelly, P., 2004. Making the biodiversity monitoring system sustainable: Design issues for large-scale monitoring systems. *Aust. Ecol.* 29, 16-30.

Attachment 1: Crew Member Observer Recruitment Process

Recruitment of Crew Member Observers (CMOs) was done by AFMA and CSIRO staff during pre-season port visits, through a Crew Awareness Program in 2004, and through information distributed to the NPF owners and operators. Recruitment of CMOs via these methods has been quite successful in attracting interest in the program. Despite fluctuations in participation rates, the program produces quality data, particularly on interactions with ‘high-risk’ and Threatened, Endangered and Protected (TEP) species. Through the project, it became obvious that long-term CMOs are often a good source of knowledge regarding potential new recruits.

To continue industry support for the CMO program, AFMA will liaise with skippers, fleet managers and/or owners to ensure they are kept informed of upcoming CMO recruitment drives. The process for recruitment will be as follows:

- Fleet managers and SFR holders will receive written notification of recruitment drives well before the start and end of each season. Letters will ask for nominations for the program, information on crew members currently involved and provide advice to crew that might be interested in joining the program.
- Nominated CMOs are to receive written or verbal (e.g. phone call to AFMA) – permission from their skippers before joining the program. AFMA will encourage interested skippers to attend annual training workshops, in conjunction with their nominated CMO.
- CMOs nominating to join the program will be assessed on years of experience in the fishery, level of interest, and their perceived ability to carry out the required functions. Newly nominated CMOs with verbal or written references from current CMOs and/or industry will be given preference.
- CMOs and fleet managers will be notified of the progress of planned training workshops
- NORMAC and SFR holders will receive regular updates on the CMO program. This will include information on the number of CMOs participating, the vessels and companies involved in the program, and the duties the CMOs are carrying out.

Fleet managers are to be reminded that recommended CMOs should be assessed on crew members’ natural enthusiasm for bycatch issues and not solely on performance of regular fishing duties. The CMO workshop should not be used as a reward for crew as the CMO program requires dedication and commitment due to the extra duties undertaken throughout each season.

CMO workshops

CMO workshops will be held annually approximately 3 weeks prior to the start of the tiger prawn season (currently the first week of July). Previous CMO workshops have often been organised with little notice due to changes in season start/finish dates. In order to ensure better organisation future workshops will be assumed to be held during the first week of July. The specific tasks involved in the organisation of the workshop can be found in the table below.

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CMO workshop tasks and timeframes

<i>TIMEFRAME</i>	<i>TASKS</i>
January-March	<ul style="list-style-type: none">• NORMAC, fleet managers and SFR holders asked to nominate CMOs and advised of the timing of the workshop.• Draft budget formulated (quotes for facilities, airfares etc) and the number of attendees approved by NORMAC and CSIRO.
March-April	<ul style="list-style-type: none">• Crew members and skippers informed of program via CAP, port visits and information sheets.
April	<ul style="list-style-type: none">• List of CMOs collated and considered by AFMA, CSIRO, fleet managers and industry.• New CMOs approved by AFMA and CSIRO.• Workshop venue determined and approved.
May	<ul style="list-style-type: none">• Attendees notified of workshop venue and dates.• Data sheets developed by CSIRO and sent to AFMA for printing.• Equipment for CMOs ordered.• Prizes for CMOs determined and arranged.• CSIRO attendees and presentations organised.• AFMA presentations organised.• Workshop schedule finalised.
June	<ul style="list-style-type: none">• Bookings for attendees confirmed.• Flights and transfers booked.• Room lists and schedule submitted to venue and CSIRO.
July	<ul style="list-style-type: none">• Workshop held.

CMO Workshop Budget and Venue

The venue for past workshops has been Seaworld Nara resort. Seaworld was originally chosen as a venue to encourage participation and reduce travel expenses as the Gold Coast is a relatively central location for most attendees. In general CMOs have been happy with the venue, though a number of other options have been discussed.

The total cost for the 2006 CMO Workshop was \$24,313.59 (see table below). The workshop was attended by 11 CMOs, 7 CSIRO staff and 3 AFMA staff. The 2006 workshop was not attended by any skippers. Future workshops will require skippers and more CMOs to attend therefore costs will increase in 2007.

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Costs for 2006 CMO Workshop

<i>Item</i>	<i>No.</i>	<i>Total cost</i>	<i>Average cost p.p.</i>
Airfares	13	\$10824.51	\$832.65
Accommodation (3 nights)	10 rooms for 18 people	\$4650	\$258.33
Meals and drinks	18	\$5780.87	\$321.15
Venue hire 2 days	For max. 20 people	\$570	\$28.50
CMO prizes (vouchers)	5	\$125	\$12.50
CMO plaques	5	\$272	\$27.20
CMO bags	20	\$682.5	\$22.75
Field sampling equipment Waterproof paper for datasheets	2200 sheets for 10 CMOs	\$609.84	\$61
Stationary	10 CMOs	\$393.87	\$39.40
Disposable cameras	40	\$405	\$40
Total		\$24,313.59	\$1643.48

5.4 Objective 3 – To develop a new, innovative, quantitative method for defining the risk to the sustainability of bycatch species from prawn trawling, and apply the model to the bycatch of the NPF

5.4.1 Introduction

S. Griffiths

Objectives of this section

Original objective: To validate the previous semi-quantitative risk assessment model used in the NPF using new data regarding elasmobranch Turtle Excluder Device exclusion rates.

New objective: Develop a new, innovative, quantitative method for defining the risk to the sustainability of bycatch species from prawn trawling, and apply the model to the bycatch of the NPF.

The concept of risk assessment

Risk can be defined as the probability that a negative impact upon a characteristic of value may arise from a present process or future event. Risk assessment can be defined as the process of estimating the probability that an adverse event with specific consequences will occur in a given period (Lindenmayer and Burgman, 2005). Risk assessment is a discipline that has been employed in a variety of fields for decades including finance (McNamee, 1998), insurance (Ericson and Doyle, 2004), medicine (Hallenbeck, 1993) and engineering (Ayyub, 2003). The approaches used to assess risks range from subjective qualitative assessments to complex quantitative models.

Risk assessment has been increasingly employed in natural resource management worldwide (Suter, 1993). Ecological risk assessment is a logical process for objectively defining the probability of an adverse effect to an organism or collection of organisms when exposed to one or more environmental or anthropogenic stressors (U.S. EPA, 1992, Newman et al. 2001). Ecological risk assessment approaches have largely focused on biosecurity (Kohler, 1992; Pheloung et al. 1999; Hayes, 2002a, 2002b, 2003) and ecotoxicology on populations of single species and ecosystems (Newman, 2001; Clements and Newman, 2002; Pastorok et al. 2002). There is also growing application of risk assessment in conservation biology and fisheries to quantify the certainty of population sustainability exposed to specific stressors (Burgman et al. 1993; Punt and Walker 1998; Nakamaru et al. 2002; Oro et al. 2004; Cheung et al. 2005). However, there have been fewer instances where ecological risk assessment has been applied to entire ecosystems. This is primarily due to the great difficulty in understanding and modelling the intricate relationships between all species and their environment, in order to predict an effect of a stressor on the system, such as fishing.

Ecological Risk Assessment in Fisheries

For fisheries in particular, there is increasing urgency for the development of models capable of assessing their impacts on entire ecosystems, mainly due to the dramatic shift in the traditional

Objective 3: Development of a new quantitative risk assessment

fisheries management paradigm from a single species focus (i.e. target species), to considering a fishery's impact on entire ecosystems (Hall and Mainprize, 2004; Scandol et al., 2005). This shift towards Ecosystem-Based Fisheries Management (EBFM) has arisen in response to increasingly stringent worldwide environmental and fisheries policy or legislation that demand fisheries take greater responsibility for managing the direct and indirect impacts on ecosystem components, other than target species as a result of fishing activities. Several countries, including Australia, began to include objectives in their fisheries management policies that addressed the ecological sustainability of their fisheries since the early 1980s (Scandol et al., 2005). However, Canada was possibly the first country to explicitly adopt EBFM in their Oceans Act of 1997 declaring that, "*Fisheries are to be managed within the broader context of integrated ocean management of the aggregate ocean uses, ecosystem features are to be considered, and a precautionary approach applied*". Since this time, many countries have followed suit, and the FAO has now developed best practice guidelines for fisheries to implement EBFM (FAO, 2003).

In Australia, ecological sustainability is now an explicit objective of environmental legislation and policy, including the *Fisheries Management Act*, Australia's Oceans Policy, and more recently the *Environmental Protection and Biodiversity Conservation Act* (EPBC) 1999. Further to these legislative requirements, perception and scrutiny by the wider community has been an important driver for fisheries to adopt more ecologically sustainable fishing practices and adopt ecosystem-based approaches to fisheries management (Aslin and Byron, 2003).

Despite existing for nearly a decade, EBFM has been successful in few fisheries, since the goals are often not clearly defined or practical to implement (Hall and Mainprize, 2004). Even the term "sustainability" used in many existing fisheries management policies in relation to single species can be ambiguous in its definition (Longhurst, 2006). Owing to the enormous complexity in understanding the relationships between components of the ecosystem, few practical tools available can assess the risk of all species impacted by fisheries to becoming unsustainable. In recent years, there have been significant advances in the quantitative approaches for modelling entire ecosystems and predicting the effects of external stressors in the system, including Atlantis (Fulton et al., 2004) and Ecopath (Christensen and Pauly, 1992). Such ecosystem models have been used with mixed success, but generally the intensity of data and modelling expertise required is generally not available or too expensive for most fisheries. As an alternative, there has been increasing interest in developing ecological risk assessment approaches, which can be used to incorporate the breadth of fishing-related activities to assess ecological sustainability in data-limited situations faced by many fisheries worldwide.

Current approaches to risk assessment in fisheries

Fletcher (2005) and Hobday et al. (2006) developed similar qualitative likelihood-consequence approaches modified from the Australian and New Zealand Standard Risk Analysis (Standards Australia, 2000). They assessed the risk of a broad range of fishing-related activities to the sustainability of specific components of fisheries supporting ecosystem, such as target species, byproduct, bycatch and general ecosystem integrity and functionality. The approach is largely facilitated by stakeholder groups agreeing on the likelihood and possible consequence of a particular risk occurring in the fishery.

The advantage of qualitative approaches is that they are often rapid, cost-effective and require little or no data. Because the method relies heavily on stakeholder involvement, management

Objective 3: Development of a new quantitative risk assessment

strategies arising from assessments are more likely to be accepted by stakeholders.

Unfortunately, such methods are of limited use to directly assess whether a fishery is operating in an ecologically sustainable manner. That is, the populations of every species impacted by a fishery are sustainable (see *EPBC Act* 1999). This Act requires the demonstration of sustainability for individual species and this is often difficult for all species in highly diverse bycatch assemblages. Furthermore, since the scoring of factors is subjective and often not based on quantitative data this approach is best used as a tool to provide a relative risk to broad species groups (e.g. “bycatch” or “red snappers” (Zellar and Snape, 2006)) or prioritise issues for closer quantitative investigation, such as stock assessment.

A number of semi-quantitative attribute-based ecological risk assessment methods have been developed to assess the relative sustainability of individual species impacted by fisheries. These include Susceptibility Recovery Analysis (SRA) (Milton, 2001; Stobutzki et al., 2001), RAPFISH (Pitcher and Preikshot, 2001), Fuzzy Logic Expert Systems (Cheung et al., 2004), Productivity-Sustainability Analysis (PSA) (Hobday et al., 2006), Qualitative Risk Matrices (Astles et al., 2006), and a rapid assessment for evaluation of risk (Walker, 2004). These methods are similar in that they rank each species on a number of criteria relating to their susceptibility to being captured, and their capacity to recover should the population become depleted. For each species, susceptibility criteria (e.g. geographic distribution, water column position and diel migration) and recovery criteria (e.g. reproduction strategy, growth rate and fecundity) are given a rank reflecting the contribution of the attribute to the overall sustainability of the species. The species having the lowest ranks across all criteria are then considered the highest risk species.

These methods have the advantage of being able to assess the relative sustainability of hundreds of species with little quantitative biological or catch data. As a result, such methods have been popular for assessing the sustainability of diverse, rare and low value species impacted by fisheries, such as bycatch in tropical trawl and gillnet fisheries (Stobutzki et al., 2001; Gribble et al., 2004). More recently, the PSA method was applied to 21 Australian fisheries, which vary substantially in size, fishing methods used, and data availability (Hobday et al., 2006). Unfortunately, these semi-quantitative attribute based methods have some severe limitations in particular fisheries (discussed below and in section 5.4.2), which initiated the development of an alternative quantitative approach.

Developing a quantitative risk assessment approach

Objective 3 of the initial project proposal was to validate the risk assessment for high risk species in the Northern Prawn Fishery (NPF) identified by Stobutzki et al. (2000). Our original approach to fulfilling this objective was to focus on the highest risk species and identify and fill major data gaps for criteria using targeted sampling during the project. By improving data quality, the aim was to increase our confidence in being able to determine whether a particular species is high risk or not. However, the project team ran several risk assessment workshops and discussed our intentions using the existing approach with several scientists involved with ecological risk assessment in fisheries. We determined that existing semi-quantitative methods can only determine the relative risk of species to fishing, and that there is no way of determining whether their populations are actually ‘at risk’ in reality without the use of an intensive monitoring program for all species impacted. Such a monitoring program would be expensive, logistically difficult, and potentially unnecessary.

Objective 3: Development of a new quantitative risk assessment

We did however satisfy this original objective of validating the existing risk assessment by updating the existing risk assessment for elasmobranchs with new data relating to exclusion rates due to Turtle Excluder Devices. The results are reported in section 5.4.2 (and Griffiths et al., 2006), which demonstrate that the method has a number of shortcomings that severely limit its use in assessing sustainability in fisheries. In particular, the method is not sensitive to changes in size selectivity in the fishery (see section 5.4.2), which is often brought about by gear modifications as a strategy by fisheries managers to reduce fishing mortality of particular species (King, 1995). As a result, we sought to develop a more powerful risk assessment method that: i) optimises the use of the limited data available for the majority of bycatch species, ii) is conceptually simple, and iii) delivers a quantitative assessment of impact. Consequently, we would have greater confidence that the species nominated for inclusion in the long-term monitoring program were in fact at high risk of depletion due to trawling in the NPF.

The method is a Sustainability Assessment of Fishing Effects (SAFE) and is a quantitative modelling approach using similar “susceptibility” and “recovery” concepts to existing semi-quantitative attribute-based methods. However, our method places greater emphasis on susceptibility elements, particularly spatial distribution of a species, which can be modelled from simple detection-nondetection data (see section 5.4.3), and their catchability in trawls. The SAFE method and its application to NPF bycatch is presented in detail in sections 5.4.4 and 5.4.5 in this section.

The term “sustainability” can have a range of specific meanings in fisheries in a biological and ecosystem context, but can also have ambiguous meanings in fisheries policy, which can incorporate biological, social and economic values (Longhurst, 2006). Therefore, our definition of sustainability is the risk assessment of NPF bycatch is based on the definition of Tesfamichael and Pitcher (2006) as “Capable of being maintained at a certain rate or level for a long time or indefinitely”.

It is also important to note that we undertook our sustainability assessments based on the current state of bycatch populations in the NPF, and not from a hindcasted pristine state before inception of the fishery, which could be very different to what presently exists (see Pitcher, 2001; Saenz-Arroyo et al. 2005). The project steering committee considered this the best approach given the lack of adequate time series data for bycatch in order to reliably reconstruct pristine community structure. This allowed recently collected data to be used as a baseline for future comparisons. A second reason for this approach was that the fishery’s high effort regions and fleet size have reduced by more than half since the fishery’s inception, therefore significantly reducing the fishing mortality on bycatch. In the absence of adequate data, it is unknown what impact the high fishing effort in the NPF had on bycatch species, but given the vast amount of data that now exists, the NPF is now well placed to ensure the remaining populations are sustainable in the long term.

Bycatch taxonomic groups assessed in this project

The risk assessment component in this project assessed the long-term sustainability of individual species within three broad species groups caught as bycatch in the NPF: elasmobranchs (sharks and rays), teleosts (bony fishes), and invertebrates. Sea snakes and sea turtles are two other major species groups that are caught as bycatch, or interacted with, by the NPF. However, these two groups were not assessed in the present study for reasons detailed below.

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Sea turtles

Sea turtles have been a major conservation concern for the NPF for many years. Poiner and Harris (1996) estimated that 5000-6000 sea turtles were captured in the NPF between 1989 and 1990, with up to 18% drowning and 50% sustaining injury. However, the compulsory use of TEDs in the NPF since 2000, a series of effort reductions and improved sea turtle handling practices have dramatically reduced the impact on sea turtles to the extent where less than 20 sea turtles per year are predicted to incur injury from trawling (Brewer et al. 2006). Owing to the conservation status of sea turtles, several long-term monitoring programs are in place around northern Australia (see review by Limpus and Chatto, 2004). Along with good biological and ecological information on most turtle species common in the NPF, long time-series data collected in these programs can be analysed using more sophisticated quantitative population models to provide a more accurate assessment of the long-term sustainability of sea turtle populations than can be achieved using our quantitative risk assessment approach.

Sea snakes

Sea snakes are another species group that are a significant conservation issue for the NPF. All seasnake species are protected in Australia under the EPBC Act, and it is agreed that trawling in the NPF nearly exclusively accounts for the majority of anthropogenic-induced mortality of these species. Milton (2001) assessed the sustainability of sea snakes impacted by the NPF and found that the highest risk species were *Hydrophis pacificus*, *Aipysurus laevis* and *Astrotia stokesii*. This led to these species being listed in the NPF strategic assessment as species of potential concern in the NPF. Sea snakes were not assessed in the present study as they are a group with considerably more biological data than most bycatch species allowing a current study (FRDC 2005/051) to use more sophisticated quantitative single species population modelling approaches to assess their long-term sustainability in the NPF. The results of this project will be delivered in 2007.

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5.4.2 Validating ecological risk assessments for fisheries: assessing the impacts of turtle excluder devices on elasmobranch bycatch populations in an Australian trawl fishery.

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Abstract

Demonstrating ecological sustainability is a challenge for fisheries worldwide, and few methods can quantify fishing impacts on diverse, low value or rare species. We employed a widely used ecological risk assessment method and incorporated new data to assess the change in sustainability of species following the introduction of Turtle Excluder Devices (TED) in Australia's Northern Prawn Fishery. Population recovery ranks changed for 19 of the 56 elasmobranch species post-TEDs, with nine species showing an increase in sustainability. Unexpectedly, ten species showed a decrease in sustainability. This was due to TEDs successfully excluding large animals from the catch, resulting in a lower mean length at capture, which reduced the recovery ranks for two criteria relying on length data. This falsely indicates that TEDs increase the impact on pre-breeding animals, thus reducing the recovery potential of these species. Our results demonstrate that existing attribute-based risk assessment methods may be inadequate for reflecting even the most obvious changes in fishing impacts on bycatch species. Industry and management can benefit greatly from an approach that more accurately estimates absolute risk. We discuss the development and requirements of a new quantitative risk assessment to be developed for the NPF and applicable to fisheries worldwide.

Keywords: extinction, prawn, rays, sharks, sustainability, tropical, fisheries management.

Introduction

The requirement for long-term ecological sustainability is increasingly influencing the management strategies of many world fisheries. It is now well documented that fishing activities can significantly impact the populations of not only target species, but also those caught incidentally as bycatch, and have the potential to disrupt the functionality of an ecosystem (Hall, 1996; Pauly et al., 2001). Although globally recognized for decades, only recently has Ecosystem-Based Fishery Management provided practical solutions through the development of guidelines and more tangible performance measures (FAO, 2003).

As is the case in many other countries, in recent years Australian environmental legislation has become more stringent to help ensure all Australian export fisheries operate in an ecologically sustainable manner, in particular the *Environment Protection and Biodiversity Conservation (EPBC) Act 1999*. Public pressure and market drivers are also requiring Australian fisheries to demonstrate ecological sustainability (e.g. Aslin and Byron, 2003). Consequently, an increasing number of fisheries are aiming to adopt ecosystem-based management strategies.

Demersal trawling is a relatively non-selective fishing method, and the bycatch often comprises a significantly higher proportion of the catch than the target species (Saila, 1983; Andrew and Pepperell, 1992). In Australia's Northern Prawn Fishery (NPF) the bycatch has averaged around 33,000 tonnes per year (Pender et al., 1992), five times the retained catch, and comprises more than 600 species (Stobutzki et al., 2001a). The NPF catches 56 species of elasmobranchs, which

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comprise around 4% of the NPF bycatch by weight (Stobutzki et al., 2001a); many of which may be particularly vulnerable to overfishing, due to their slow growth, low natural mortality rates and low reproductive potential (Stevens, 1997; Walker, 1998; Prince, 2002; Baum et al., 2003). Caution needs to be exercised in managing these species, as the populations of some elasmobranchs in Australia have been projected to go extinct in as little as six years, despite very low fishing mortality (Otway et al., 2004).

In 2000 the NPF introduced the mandatory use of Turtle Excluder Devices (TEDs) and Bycatch Reduction Devices (BRDs), which has resulted in the exclusion of >90% of individuals for nine species of elasmobranchs and smaller exclusion rates for six other species (Brewer et al., 2004). These mainly large animals usually include the large breeding females and their survival would be expected to improve the chances of long term sustainability for these species. Although the impact of the fishery on elasmobranch populations has been dramatically decreased since 2000, their long-term sustainability in the fishery is still unknown. The juveniles of most species and adults of smaller species can pass through the bars of a standard TED and many suffer damage or death in trawl catches.

Demonstrating that populations of elasmobranchs caught by the fishery are sustainable is difficult and expensive using traditional monitoring techniques, due to the high number of species involved, and the relative rarity of many of these species. Additionally, sampling the true population and size composition of larger species which are almost entirely excluded by TEDs is difficult now that TEDs are compulsory in the NPF. For this reason, an alternative strategy is needed to demonstrate the long-term sustainability of elasmobranchs with which the NPF interacts.

Dulvey et al. (2000) reviewed the methods by which extinction risks can be quantified for populations of species impacted by fishing and showed that few methods are useful for assessing large numbers of species, for which biological data are few. An exception is a method developed concurrently by Milton (2001) for sea snakes and Stobutzki et al. (2001b, 2002) for teleosts and elasmobranchs; herein referred to as a Susceptibility-Recovery Analysis (SRA). The method is a simple qualitative risk assessment technique which ranks each species using a number of criteria describing (i) their susceptibility to capture by a specific fishing method and (ii) their capacity to recovery once populations are depleted. The overall susceptibility and recovery ranks for each species can be plotted to determine the species which might be most 'at risk' to being overfished. Because of the method's simplicity and capability of handling hundreds of species with limited data, it has been adopted by several Australian fisheries to assess ecological sustainability including the Queensland offshore and inshore gillnet fisheries (Gribble et al., 2004), the Northern Prawn Fishery (Stobutzki et al., 2001b, 2002), the Western Australian Shark Bay Prawn Fishery (EA, 2002), and the Queensland East Coast Trawl fishery (QDPI, 2004).

The aim of this paper was to i) examine the change in sustainability of individual species after the introduction of TEDs in the NPF by incorporating new data on elasmobranch exclusion and ii) validate the SRA method and assess its sensitivity to changes in catchability resulting from the introduction of TEDs in the NPF.

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Materials and methods

The SRA method used in this paper uses information for each bycatch species on 11 criteria describing: 1) the susceptibility of the species to capture and mortality by fishing, and 2) the capacity of the population to recover following depletion (Milton, 2001; Stobutzki et al., 2001b, 2002). Each criterion was weighted according to its relative importance in contributing to population sustainability. Each species was given a rank on a scale of 1-3 for each criterion. A rank of 1 was assigned to a criterion that would contribute to the species being highly vulnerable to capture, or had a low capacity to recover. A rank of 3 was assigned to a criterion that would contribute to the species having a low vulnerability to capture, or had a high capacity to recover. As a precautionary approach, a rank of 1 was assigned to a particular criterion in cases where no species-specific information was available, nor information on closely related species. A description of the ranking criteria and weighting for each criterion are shown in Table 10. The ranks were summed for all susceptibility and recovery criteria for each species, and the values plotted on a two-dimensional graph, with recovery values on the x axis and susceptibility scores on the y axis. The species having the lowest ranks on both axes were considered the highest risk species. Below we briefly describe the susceptibility and recovery criteria used in the SRA (after Stobutzki et al., 2002).

Susceptibility criteria

Water column position - Prawn trawling in the NPF occurs on or close to the sea floor. As a result, benthic or demersal species are more likely to be captured than benthopelagic or pelagic species.

Survival – An estimate was made of the within-net survival, or the mortality of animals before they are landed on the deck. Survival ranges between 0-100% and was divided into thirds for the ranks.

Range – The geographic distribution of a species within the NPF was determined from the presence or absence of a species in samples collected by fishery-dependent and fishery-independent surveys in 9 fished regions of the NPF. It was assumed that species having a broad geographic range were less ‘at risk’ of depletion than species with a restricted range.

Day and night catchability – Fishing predominately takes place at night in the NPF tiger prawn fishery. As a result, species more susceptible to capture at night (i.e. nocturnal vertical migrations) are more likely to be impacted by trawling.

Diet – The diet of a species may attract them to trawl areas and make them vulnerable to capture. We assumed that species that fed upon commercially important prawns or fed demersally were more susceptible to capture than species which do not feed on prawns or feed in the pelagic zone.

Depth range – Fishing in the NPF are made between 15 m and 40 m. We assumed that species that occur within this depth range are more susceptible than species found in deeper or shallower water.

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Recovery criteria

Probability of breeding – This criterion is an indicator of the potential reproductive capacity of a species' population. We assumed that where the mean length of a species in the catch is greater than the length at sexual maturity, the majority of individuals have had the opportunity to breed before capture and the population is likely to be sustainable. In contrast, where the mean length of a species in the catch is significantly less than the length at sexual maturity, this may be seen as an indicator that large mature fish have been fished down. As a result, the reproductive capacity of the population is reduced and has a lower capacity to recover after depletion. A *t*-test was used to determine whether the mean length at capture was significantly different from the size at maturity for each species.

Maximum size – Maximum size of a species was used as an indicator of relative recovery rate. Larger species generally live longer, thus their populations would recover more slowly after depletion than species with short life spans.

Removal rate – Removal rate is the percentage of the average annual biomass removed by the fishery as bycatch. Species having a higher proportion of their biomass removed as bycatch were assumed to have lower capacity to recover. Removal rate was based on the catch rates from fishery-independent and fishery-dependent surveys in the NPF. The biomass from these surveys was multiplied by the number of fishing days in the 1997 tiger prawn season. The total biomass from each species was estimated for the total NPF area and the biomass taken by the fishery was expressed as a percentage. See Stobutzki et al., (2002) for details of biomass calculations.

Annual fecundity – Annual fecundity was estimated from data in the literature and biological collection during NPF surveys. Annual fecundity of a species was calculated as the number of pups produced per female multiplied by the number of times the females are thought to breed each year. Where the breeding frequency was unknown, it was assumed to be annual.

Mortality index – The capacity of a population to recover is likely to be related to its fishing mortality rate. Because limited biological information exists for most species, an index of mortality was calculated using length-frequency data as:

$$\text{Mortality index} = (L_{max} - L_{ave}) / (L_{ave} - L_{min}),$$

where L_{max} is the maximum recorded length of the species; L_{ave} and L_{min} is the average length and minimum length at capture by the fishery, respectively. The closer L_{ave} is to L_{max} the lower the fishing mortality. As fishing mortality increases, L_{ave} approaches L_{min} .

New data

Brewer et al. (2004) provided new species specific data on maximum, minimum and mean length at capture, and exclusion rates for 35 elasmobranch bycatch species in NPF trawl catches from nets where TEDs were installed. No additional species specific data were available for the remaining 21 species recorded as NPF bycatch, due to low sample sizes. Brewer et al. (2004) measured exclusion rates in nets fitted with various combinations of TEDs and Bycatch Reduction Devices (BRDs). Since BRDs were shown not to significantly reduce elasmobranch bycatch for the majority of species, we used data where only a TED was fitted to the net. We

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included data from both upward and downward-excluding TEDs since no differences were found in elasmobranch exclusion rates between TED types.

The new data from Brewer et al. (2004) allowed us to reassess the rankings for three recovery criteria for each of the 35 species; probability of breeding, removal rate and mortality index to provide an updated risk assessment for elasmobranchs in the NPF. We updated information on the size at first capture and mean length at capture, which allowed recalculation of ranks for the probability of breeding and mortality index criteria. Ranks for the removal rate criterion were revised by reducing the catch for each species estimated by Stobutzki et al. (2002) by the exclusion rate due to TEDs. For example, if the fishery removed 700 t from a total biomass of 1000 t of a species, the removal rate is 70% and a rank of 1 in the SRA. If TEDs reduced the biomass of the species by 80%, meaning the fishery now only removes 140 t, the new removal rate is 14% and a rank of 3.

New unpublished information was also available on the biology and ecology of some elasmobranch bycatch species, which could have been integrated into a number of susceptibility criteria. However, we did not use this data in order to avoid masking the impact of TEDs on the sustainability assessment of elasmobranchs.

Results

We recorded a change in the risk assessment recovery ranks for 19 of the 56 elasmobranch species after incorporating TED exclusion data (Table 11). Of the remaining 37 species, five species showed no change because the updated data did not change ranks for specific recovery criteria, while there were no new data available for the remaining 32 species. Of the 19 species that did show a change in their recovery rank, nine species showed an increase in their recovery rank (i.e. increased sustainability), while 10 species showed a decrease (i.e. decreased sustainability).

Only one of the species considered by Stobutzki et al., (2002) to be at highest risk, *Aetomyleus vespertilio*, showed an increase in recovery rank (= decreased risk) by 38%, from the 52nd (1.33) highest ranked recovery ability to the 31st (1.83) highest. This species was excluded by TEDs by 100% (Table 11). Similarly, a large change in the recovery rank was found for *Atelomyxerus fasciatus*, increasing by 24% from 38th (1.75) highest ranked recovery ability to the 12th (2.17) highest. The change in recovery ranks for these species was enough to change their status from potentially 'at risk' to lowest risk. The recovery rank for *Rhizoprionodon acutus* also increased significantly by 34%, from 19th highest rank species to the highest (1st) ranked recovery ability. However, the risk status of this species did not change since it was already considered to be lowest risk. *Chiloscyllium punctatum* and *Carcharhinus dussumieri* were two other species to show a change in recovery rank by at least 5%, but this did not change their risk status. Interestingly, *Himantura undulate* and *Anoxypristis cuspidata* showed only small increases in their recovery ranks (5% and 4%) despite TEDs excluding these species from catches by 95% and 79%, respectively.

In contrast, *Himantura toshi* which was the equal highest ranked species in terms of recovery capacity (2.58) (Stobutzki et al., 2002), decreased in recovery rank by 22% to become the 21st (2.00) highest in recovery ability. This changed the status of this species from lowest risk to potentially 'at risk'. Five species, *Dasyatis leylandi*, *Nebrius ferrugineus*, *Sphyrna mokarran*,

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Dasyatis kuhlii and *Rhynchobatus australiae* showed a negative change in recovery rank by at least 5%, but this did not change their status of being lowest risk. Surprisingly, *Nebrius ferrugineus* and *Pastinachus sephen* were both excluded by TEDs by greater than 97% (Table 11), yet showed a decrease in recovery rank by 10% and 4%, respectively. Furthermore, *Rhynchobatus australiae* and *Amphotistius annotata* both showed reasonable exclusions from catches due to TEDs (28% and 26%) but again, showed a negative change in recovery rank by 5% and 4%, respectively.

Five species considered to be at medium risk with respect to their recovery capacity, *Himantura uarnak*, *Aetobatus narinari*, *Rhinobatos typus*, *Rhina ancylostoma* and *Stegostoma fasciatum*, showed no change in recovery rank despite being excluded by 100% due to TEDs (Table 11). This was mainly due to the recovery ranks for the removal rate criterion being at the maximum value of 3 (see Stobutzki et al., 2002), thus not allowing for any increase in ranks.

Discussion

The vulnerability of elasmobranch populations to decline due to fishing activities has been shown in many fisheries worldwide, mainly due to their generally slow growth, low natural mortality rate and low reproductive potential (Stevens, 1997; Walker, 1998; Prince, 2002; Baum et al., 2003). The compulsory use of TEDs in the NPF has been highly effective in reducing some categories of unwanted bycatch, including many large species of elasmobranchs (Brewer et al., 2004). However, assessing the effectiveness of such a management strategy on the sustainability of elasmobranch species in the NPF is difficult due to the size and remoteness of the fishery (over 1 million km²), the high number of species impacted by the fishery, limited biological information for the vast majority of species, the variable intensity of impact due to temporal and spatial closures in the fishery and temporal shifts in fishing effort. These also factors make the monitoring of each individual species logistically difficult and expensive.

The SRA model is the only feasible method that can assess the ecological sustainability of large number of species with limited biological information. However, this method is relatively new and until now has not been validated by investigating the effect of including significant new data. The re-implementation of this method described here suggests that the introduction of TEDs had a negligible or negative impact on the sustainability status of 15 elasmobranch species in the NPF, despite large reductions in the number and mean size of animals caught in nets using TEDs. In fact, we found ten species to have a lower recovery rank after the introduction of TEDs, and thus be considered less sustainable. This was mainly attributed to the exclusion of large animals from the catch, which resulted in a lower mean length at capture. Owing to this reduction in mean length at capture, ranks for two recovery criteria that rely on length data (probability of breeding before capture and mortality index) were reduced. This falsely indicates that the fishery is increasing its impact on pre-breeding animals, thus reducing the recovery potential of the species.

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Table 10. Susceptibility and recovery criteria, definition of ranks and relative weighting used to determine the relative sustainability of 56 elasmobranch species caught as bycatch in the NPF after the introduction of Turtle Excluder Devices. Asterisks denote criteria where ranks for individual species were changed using updated data from Brewer et al. (2004). Table modified from Stobutzki et al. (2002).

Criteria	Weight	Rank		
		1	2	3
Susceptibility				
Water column position	3	Demersal or Benthic	N/A	Benthopelagic or pelagic
Survival	3	Probability of survival <33%	Probability of survival 33%-66%	Probability of survival >66%
Range	2	Species range ≤3 fishery regions	Species range 3-6 fishery regions	Species range >6 fishery regions
Day and night catchability	2	Higher catch rate at night	No difference between day and night	Higher catch rate during the day
Diet	2	Known to, or capable of, feeding on commercial prawns or benthic organisms	N/A	Feed on pelagic organisms
Depth range	1	Less than 60 m	N/A	Deeper than 60 m
Recovery				
Probability of breeding*	3	Probability of breeding before capture <50%	Probability of breeding before capture not significantly different from 50%	Probability of breeding before capture >50%
Maximum size	3	Maximum disc width >1755 mm Maximum total length >4781 mm	Maximum disc width 853-1755 mm Maximum total length 1861-4781 mm	Maximum disc width ≤ 853 mm Maximum total length ≤ 1861 mm
Removal rate*	3	Removal rate >66%	Removal rate 33-66%	33% ≤ removal rate
Annual fecundity	1	Annual fecundity ≤5 young per year	Annual fecundity 5-19 young per year	Annual fecundity > 19 young per year
Mortality index*	1	Mortality index >3.47	Mortality index 0.92-3.47	Mortality index <0.92

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Table 11. Species of elasmobranchs included in a risk assessment for the Northern Prawn Fishery, showing the recovery ranks before and after the introduction of TEDs into the fishery, and the percentage change to the overall recovery rank. Species are listed in descending order of percentage change in recovery rank. Recovery ranks before TEDs adopted from Stobutzki et al., (2002).

Family	Species	Recovery rank before TEDs	Recovery rank after TEDs	% change in recovery rank	% exclusion by TEDs	% change in mean length	% change in min length
Myliobatidae	<i>Aetomylaeus vespertilio</i>	1.33	1.83	37.59	100	0	0
Carcharhinidae	<i>Rhizoprionodon acutus</i>	2.00	2.67	33.50	-	11.98	0
Scyliorhinidae	<i>Atelomycterus fasciatus</i>	1.75	2.17	24.00	-	26.66	0
Hemiscylliidae	<i>Chiloscyllium punctatum</i>	2.00	2.17	8.50	-	0.27	0
Carcharhinidae	<i>Carcharhinus dussumieri</i>	2.25	2.42	7.55	-	12.11	0
Dasyatidae	<i>Himantura undulata</i>	1.75	1.83	4.57	95	-27.05	0
Carcharhinidae	<i>Carcharhinus sorrah</i>	2.08	2.17	4.33	-	52.61	0
Pristidae	<i>Anoxypristis cuspidata</i>	2.00	2.08	4.00	79	-1.81	0
Hemiscylliidae	<i>Hemigaleus microstoma</i>	2.58	2.67	3.49	-	3.63	0
Carcharhinidae	<i>Carcharhinus albimarginatus</i>	1.67	1.67	0	-	0	0
Carcharhinidae	<i>Carcharhinus amboinensis</i>	1.50	1.50	0	-	0	0
Carcharhinidae	<i>Carcharhinus brevipinna</i>	1.67	1.67	0	-	0	0
Carcharhinidae	<i>Carcharhinus fitzroyensis</i>	1.83	1.83	0	-	0	0
Carcharhinidae	<i>Carcharhinus leucas</i>	1.50	1.50	0	-	0	0
Carcharhinidae	<i>Carcharhinus limbatus</i>	1.58	1.58	0	-	0	0
Carcharhinidae	<i>Carcharhinus macroti</i>	2.08	2.08	0	-	0	0
Carcharhinidae	<i>Carcharhinus tilstoni</i>	2.00	2.00	0	-	-0.54	0
Carcharhinidae	<i>Dasyatis brevicaudatus</i>	1.08	1.08	0	-	0	0
Carcharhinidae	<i>Galeocerdo cuvier</i>	2.17	2.17	0	-	0	0
Carcharhinidae	<i>Negaprion acutidens</i>	1.92	1.92	0	-	0	0
Carcharhinidae	<i>Prionace glauca</i>	1.58	1.58	0	-	0	0
Dasyatidae	<i>Dasyatis</i> sp. A	1.75	1.75	0	-	0	0
Dasyatidae	<i>Dasyatis thetidis</i>	1.92	1.92	0	-	0	0
Dasyatidae	<i>Himantura fai</i>	1.75	1.75	0	-	0	0
Dasyatidae	<i>Himantura granulata</i>	1.75	1.75	0	-	0	0
Dasyatidae	<i>Himantura jenkinsii</i>	1.67	1.67	0	-	0	0
Dasyatidae	<i>Himantura</i> sp. A	2.00	2.00	0	-	0	0
Dasyatidae	<i>Himantura uarnak</i>	1.92	1.92	0	100	0	0
Dasyatidae	<i>Taeniura meyeri</i>	1.50	1.50	0	-	0	0
Dasyatidae	<i>Urogymnus asperrimus</i>	1.83	1.83	0	-	0	0
Gymnuridae	<i>Gymnura australis</i>	2.58	2.58	0	-	1.48	0
Myliobatidae	<i>Aetobatus narinari</i>	1.58	1.58	0	100	0	0
Myliobatidae	<i>Aetomyleus nichofii</i>	2.08	2.08	0	-	-5.49	0
Narcinidae	<i>Narcine westraliensis</i>	1.50	1.50	0	-	0	0
Orectolobidae	<i>Orectolobus ornatus</i>	1.75	1.75	0	-	0	0
Pristidae	<i>Pristis clavata</i>	1.25	1.25	0	-	0	0

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Table 11 continued

Family	Species	Recovery rank before TEDs	Recovery rank after TEDs	% change in recovery rank	% exclusion by TEDs	% change in mean length	% change in min length
Pristidae	<i>Pristis microdon</i>	1.50	1.50	0	-	0	0
Pristidae	<i>Pristis pectinata</i>	1.17	1.17	0	-	0	0
Pristidae	<i>Pristis zijsron</i>	1.75	1.75	0	-	0	0
Rhinobatidae	<i>Rhinobatos typus</i>	2.17	2.17	0	100	0	0
Rhynchobatidae	<i>Rhina ancylostoma</i>	2.17	2.17	0	100	0	0
Scyliorhinidae	<i>Galeus</i> sp. A	1.50	1.50	0	-	0	0
Sphyrnidae	<i>Eusphyrna blochii</i>	2.50	2.50	0	-	0	0
Sphyrnidae	<i>Sphyrna lewini</i>	2.00	2.00	0	-	19.89	0
Squatinae	<i>Squatina</i> sp. A	1.50	1.50	0	-	0	0
Stegostomatidae	<i>Stegostoma fasciatum</i>	2.17	2.17	0	100	0	0
Carcharhinidae	<i>Rhizoprionodon taylori</i>	2.58	2.50	-3.10	-	0	0
Dasyatididae	<i>Pastinachus sephen</i>	2.08	2.00	-4.00	97	-68.40	-24.44
Hemiscylliidae	<i>Hemipristis elongatus</i>	2.00	1.92	-4.00	-	-16.94	0
Dasyatididae	<i>Amphotistius annotata</i>	1.83	1.75	-4.37	26.3	-21.33	0
Rhynchobatidae	<i>Rhynchobatus australiae</i>	1.83	1.75	-4.55	28.2	-25.35	0
Dasyatididae	<i>Dasyatis kuhlii</i>	2.25	2.08	-7.56	-	-12.20	0
Sphyrnidae	<i>Sphyrna mokarran</i>	1.92	1.75	-8.85	-	-38.20	0
Ginglymostomatidae	<i>Nebrius ferrugineus</i>	2.42	2.17	-10.33	100	0	0
Dasyatididae	<i>Dasyatis leylandi</i>	2.25	2.00	-11.11	-	-11.64	0
Dasyatididae	<i>Himantura toshi</i>	2.58	2.00	-22.48	-	-16.17	0

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We also found that despite Brewer et al. (2004) showing that some species are excluded by more than 97%, such as *Nebrius ferrugineus* and *Pastinachus sephen*, their removal rate rank could not be increased as a maximum rank of 3 was given in the previous risk assessment. This may be a result of biomass removal rate value ranges defining the three ranks being generalized across all 56 species. Consequently, these data categories may not be accurate enough to reflect changes in removal rates at the species level. In retrospect, species with high exclusion rates by TEDs should have initially received lower ranks in order for such an effective bycatch management strategy to be correctly reflected in the risk analysis.

The SRA method clearly does not reflect changes in risk due to changes in size selectivity, but may be suitable for once-off assessments of species where there are few data, and to help guide management and research where the relative risk of species is desired. The SRA model provided a critical first step in the process for assessing ecological sustainability, especially for speciose assemblages with limited data. However industry and management can benefit greatly from an approach that more accurately estimates absolute risk, and consequently can recommend a more targeted and hence, cost effective monitoring or mitigation program. Cheung et al. (2004) suggest an alternative approach using a fuzzy logic system to rank the relative vulnerability of species of extinction. This appears to be an improvement on previous ecological risk assessment methods, but again, this method provides only a relative indicator of risk. In the ecological auditing process underway in Australia by the Commonwealth's Department of Environment and Heritage, fisheries may need to provide more quantitative estimates of their potential impacts on the ecosystem.

An alternative approach may be to place greater reliance on the spatial distribution (e.g. geographic and water column distribution) of each species as a measure of the potential impacts from fishing. Some authors have used various aspects of geographic range, such as latitude (Dulvy and Reynolds, 2002) and "spatial behavior strength" (Cheung et al., 2004), as at least one major factor to assess the persistence of a marine species' population. Stobutzki et al. (2002) used "range" (i.e. geographic distribution) as a criterion and ranks species susceptibility based on how many of the 11 high effort fishing regions it occurs in; from most susceptible (<3 regions) to least susceptible (>6 regions). However, this criterion does not take into account whether the species is also distributed outside the fished region. This is a potentially dangerous assumption if the entire natural geographic distribution of a species is largely within high effort regions. In this scenario the species is assigned a low susceptibility to capture using the definition of Stobutzki et al. (2002), it may potentially be at far greater risk of overfishing than a species found in only one high effort area but distributed further into unfished regions.

Fishing effort in the NPF is highly aggregated, as is the case in many fisheries, and the actual fished region (i.e. greater than 5 boat days per year) is only about 6% of the entire NPF managed region (Zhou and Griffiths, *in press*; section 5.4.4). Consequently, there may be significant spatial refuge outside the fishery, where a species may be unaffected by fishing. Depending on the mobility of a species, individuals in these refuge areas may even replenish the proportion of the population taken by the fishery. Such an approach may enable scientists to discount a large number of species as being not 'at risk' from fishing purely on the basis of susceptibility to capture (distribution and catchability). The remaining species may then be dealt with in a more stringent manner, i.e. ongoing monitoring and population projection to better understand the potential impacts of fishing.

Objective 3: Development of a new quantitative risk assessment

Our results show that although effective management strategies may be implemented by fisheries, such as implementation of TEDs in the NPF, currently used qualitative ecological risk assessment methods may be inadequate in reflecting even the most obvious changes in risk to individual species. Demonstrating ecological sustainability is an enormous challenge for scientists and fisheries managers, especially in fisheries that interact with hundreds of species, such as those in tropical regions. However, it is clear that more rigorous methods need to be developed and adopted by fisheries if meaningful progress is to be made towards sustainable fisheries management and conservation.

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5.4.3 Estimating abundance from detection–nondetection data for randomly distributed or aggregated elusive populations

S. Zhou and S. Griffiths

Abstract

Estimating abundance is important in many ecological studies in order to understand the spatial and temporal dynamics of a population, which can assist in management and conservation. However, direct estimates of abundance can be difficult and expensive to obtain, particularly for wide-ranging, rare or elusive species. An alternative — estimating from detection–nondetection data — is a challenging but alluring concept to ecologists since the cost and effort of a study can be greatly reduced. This paper describes a method for estimating the abundance of randomly distributed or aggregated populations by using binary data where the probability of detection is less than one. The performances of the models were evaluated by computer simulations comprising 1,620 cases. The results show that the accuracy of the abundance estimates increases as the sampling rate, efficiency of survey method, and the number of repeated surveys increase, whereas the accuracy declines as individuals become more aggregated. For a randomly distributed population, using a sampling rate of 0.05 in a survey method with a detection probability of 0.5, and repeating surveys three times provides sufficient accuracy of abundance. For an aggregated population, to achieve reasonably accurate abundance estimates the sampling rate should be doubled and each cell should be repeatedly surveyed on 4 to 6 occasions.

Keywords: abundance, distribution, detection–nondetection, presence–absence, random, aggregated, Poisson, negative binomial, occupancy.

Introduction

Estimates of abundance can be critical to ecological studies of the dynamics of a population in space and time. Abundance data are particularly important in a management and conservation context as they give a direct measure of a species' viability in a discrete area, which can guide management decisions. Traditionally, the abundance of a species is estimated from the number of animals detected in surveys in the area of interest (Cochran, 1977; Thompson, 1992). Unfortunately, conducting count surveys is often expensive, particularly for species that are elusive or distributed over large areas, such as birds and fishes, as a large number of samples are required to adequately represent the population.

In view of the expense (and often the difficulty) of collecting count data, many ecologists are trialing detection–nondetection data (Azuma et al., 1990; Hanski and Gyllenberg, 1997; Kunin, 1998; Carroll et al., 1999; Strayer, 1999; Bayley and Peterson, 2001; MacKenzie et al., 2003; Bailey et al., 2004; Brotons et al., 2004; Gelfand et al., 2006). Although less informative, detection–nondetection (often inaccurately called presence–absence) data are often less expensive to collect and can be collected more rapidly across a large spatial scale because the sampling regimes are less restricted. Detection–nondetection data have been used to predict distributions of species, or site occupancy. Predicting the occupancy rate of elusive species depends on the probability that a species or individual is recorded in the sampling unit being sampled by a particular method, whether this be visual or audio detection, direct capture, or detection by some other means. This is most commonly referred to as “detection probability” or “detectability” (Thompson, 1992), either of which can be at the individual or species level.

Objective 3: Development of a new quantitative risk assessment

Detection probability at the level of the individual, denoted as d in this paper, can be defined as the probability of detecting one particular individual per unit of effort, for example one visit to a survey site (Thompson, 1992). It can also be interpreted as the proportion of individuals within one unit of area being detected by one unit of effort. In contrast, the detection probability at the species level, denote as D in this paper, is the probability of detecting any one or more individuals in the sampling area per unit of effort. That is, D is the probability of detecting the species in the area. If detecting one individual does not affect detection of other conspecifics (i.e. they are independent), then the relationship between d and D can be described by a binomial distribution: $D = 1 - (1-d)^n$, where n is the number of individuals in the sampling unit.

Attempts to estimate the site-occupancy rate based on detection and non-detection data when the detection probability is less than one have been successfully developed (MacKenzie et al., 2002, 2003; Tyre et al., 2003) and applied to cryptic terrestrial species, including owls and salamanders (MacKenzie, 2002; Bailey et al., 2004). A simulation study by Wintle et al. (2004) compared different methods for predicting site occupancy and demonstrated that a zero-inflated binomial model provides the least biased estimator of occupancy rate.

Parellel with the development of models for occupancy and detection probability using detection-nondetection data, there is growing interest in using occupancy rate and detection probability to infer information about abundance (Condit et al., 2000; Gaston et al., 1998, 2000; He and Gaston, 2000a, 2000b, 2003; He et al., 2002; Holt et al., 2002; Falster et al., 2001; Harte et al., 2001; Nielsen et al., 2005). However, there are great difficulties in applying such an approach to elusive populations, which is probably why so few studies tackle the problem.

A method of estimating abundance from presence-absence data for both randomly distributed and aggregated populations (He and Gaston, 2000), is most suitable for conspicuous immobile species, such as trees, where the detection probability equals one. Royle and Nichols (2003) proposed a more flexible and useful approach for estimating abundance from detection-nondetection data on elusive species by applying a mixture probability model. Two types of parametric distributions were tested for population estimation: a Poisson and a negative binomial. The latter was considered only briefly because of less convincing results (Royle and Nichols, 2003), but has since been modified or extended and applied to studies of terrestrial species (Royle, 2004; Kery et al., 2005; Royle and Link, 2005; Royle et al., 2005). Similar methods have been proposed for aggregated populations, but have assumed known parameters that describe the aggregation pattern and sampling efficiency or detectability (Mangel and Smith, 1990; Yamada and Zenitani, 2005).

The present paper describes simpler alternative methods of estimating the abundance of elusive, randomly distributed or aggregated populations from detection-nondetection data. Our methods specifically take the size of the study area and the size of the sampling unit into account. The models are somewhat simpler than earlier studies with such data (Royle and Nichols, 2003). They do not require integration over infinite supporting points (in practice the infinity has to be truncated to an arbitrary number, and this number is limited by the current computer technology), require less computational complexity in their operation, achieve convergence of parameters more easily and are more stable, especially with highly aggregated and abundant species. We used computer simulations to evaluate the performance of these models and the accuracy of the parameter estimates.

Materials and methods

Populations with a random distribution

Let us assume that a study area of size A is divided into A/a cells of equal size a of the same shape. Individuals of a particular species are independently and randomly distributed in area A (the area may be stratified to obtain such a distribution status). The conditional probability that a particular individual is present in one cell (sampling unit) with area size a , given it is present in area A , is therefore a/A . The number of individuals (X) in each cell follows a binomial distribution:

$$\Pr(X = N_i) = \binom{N}{N_i} \left(\frac{a}{A}\right)^{N_i} \left(1 - \frac{a}{A}\right)^{N-N_i}, \quad (1)$$

where N is the total number of individuals in the studied area, and N_i is the number of individuals in one sampling unit of size a . From (1) the probability that a sampling cell has zero individuals is:

$$\Pr(X = 0) = \binom{N}{0} \left(\frac{a}{A}\right)^0 \left(1 - \frac{a}{A}\right)^N = \left(1 - \frac{a}{A}\right)^N. \quad (2)$$

Hence, the probability that a cell has at least one individual is:

$$\psi = \Pr(X > 0) = 1 - \Pr(X = 0) = 1 - \left(1 - \frac{a}{A}\right)^N. \quad (3)$$

Since ψ is the probability of one or more individuals' presence in one sampling unit, $(1-\psi)$ [equation (2)] is the probability of absence. It is equivalent to assuming that, within a specific stratum, the number of individuals follows a Poisson distribution, i.e.:

$$\Pr(X) = e^{-\lambda} \frac{\lambda^X}{X!}, \quad (4)$$

where $\lambda = aN/A$ is the mean density (number of individuals per cell). Thus, the probability that one cell has one or more individuals is:

$$\psi = 1 - \Pr(X = 0) = 1 - e^{-\lambda}. \quad (5)$$

This is an alternative to the parameterisation of abundance N in equation (3). A probability model was proposed by MacKenzie et al. (MacKenzie et al., 2002, 2003; also Tyre et al., 2003) to estimate site-occupancy rate based on detection and non-detection data. Equations (3) and (5) can be incorporated into the site occupancy model to obtain the abundance of a randomly distributed population.

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The site occupancy model assumes each cell with an area size of a has a probability ψ of being occupied by at least one individual of the interested species. Conditional to the species existing in that unit of area, it may not be detected in every survey. The result for any given survey can therefore be considered as two binomial processes working simultaneously: the probability ψ that a species is present in the cell, and the probability D that one or more individuals of that species are detected, given that the species is indeed present in that cell. This type of problem is a mixture distribution, with a mixing probability of ψ and two binomial components (MacKenzie et al., 2002). Repeated surveys of the study area will allow estimation of these two probabilities. For cell i with an area size of a , assume that in the total of m_i surveys conducted, a species was detected in n_i surveys ($n_i \leq m_i$). If we assume that abundance within the cell remains constant during the period of repeated surveys (see discussion for such an assumption), then the likelihood that a species is present in cell i , ψ_i , and one or more individuals of that species are detected, D_i , is:

$$L(\psi_i, D_i | m_i, n_i) = \binom{m_i}{n_i} D_i^{n_i} (1 - D_i)^{m_i - n_i} \psi_i, \quad n_i > 0, \text{ detected} \quad (6)$$

$$L(\psi_i, D_i | m_i, n_i) = (1 - D_i)^{m_i} \psi_i + (1 - \psi_i), \quad n_i = 0, \text{ not detected.} \quad (7)$$

Assuming each unit of area a has an average probability of presence ψ and probability of detecting the species (i.e., one or more individuals) D (see discussion), substituting equations (3) or (5) into (6) and (7), we obtain the following joint likelihood function for all cells:

$$L(N, D | m_i, n_i, A, a) = \prod_{\substack{i=1 \\ n_i > 0}}^{C_1} \left\{ \binom{m_i}{n_i} D^{n_i} (1 - D)^{m_i - n_i} \left[1 - \left(1 - \frac{a}{A}\right)^N \right] \right\} \times \prod_{\substack{i=1 \\ n_i = 0}}^{C_0} \left\{ (1 - D)^{m_i} \left[1 - \left(1 - \frac{a}{A}\right)^N \right] + \left(1 - \frac{a}{A}\right)^N \right\} \quad (8)$$

where C_1 is the total number of sampled cells when $n_i > 0$, and C_0 is the total number of sampled cells when $n_i = 0$. By taking the logarithm of equation (8) as an objective function and maximizing it with regard to N and D , we can estimate the total abundance of a species in the area with a total of A/a units of cells. Since the probability of detection D may be depend on the sampling method used in the survey (e.g. size- and species-selective fishing gear used in fishery-related research) and site characteristics. We may incorporate these variables into D by using a logistic function (McKenzie, 2002):

$$D = \frac{1}{\exp[-(\alpha + \boldsymbol{\beta}\mathbf{M} + \boldsymbol{\gamma}\mathbf{H})] + 1}, \quad (9)$$

where vector \mathbf{M} could be survey methods or sampling gear types, \mathbf{H} could be specific habitats or fishing-area characteristics, and $\boldsymbol{\beta}$ and $\boldsymbol{\gamma}$ are the vectors of model parameters to be estimated. Model (8) and equation (9) imply that we did not take unknown local abundance (which is usually heterogeneous) into account for its effect on D . This will result in a poor estimate of abundance in each single cell. However, when we estimate the total abundance across entire area A by combining likelihood in all cells, model (8) can produce accurate estimates (see discussion).

Populations with an aggregated distribution

Negative binomial distribution (NBD) is the probability distribution of a certain number of failures and successes in a series of independent and identically distributed Bernoulli trials. NBD has been widely used to describe the spatial distribution of aggregated populations. Several parameterisation approaches are commonly used, although here we use two real-valued parameters π and k :

$$\Pr(X = N_i) = \binom{N_i + k - 1}{k - 1} (\pi)^k (1 - \pi)^{N_i}. \quad (10)$$

For our purposes of estimating a species' abundance, $X = 0, 1, 2, \dots$ is the number of individuals in each unit of area, π is the conditional probability that an individual is present in a cell given it is present in the study area (success, $1 \geq \pi \geq 0$, dependent on the size of sampling unit a , the total area A , and abundance N), and k is a parameter describing the extent of aggregation (overdispersion, $k > 0$). Formulae for $E(X)$ and $Var(X)$ for the NBD are given by

$$E(X) = \frac{k(1 - \pi)}{\pi}, \quad (11)$$

and
$$Var(X) = \frac{k(1 - \pi)}{\pi^2}. \quad (12)$$

Since $E(X)$ is the mean density in the entire area A , if we use the same notation in a Poisson distribution, i.e., $E(X) = \lambda = aN/A$, then the probability that one particular unit of area of size a has zero individuals becomes:

$$\Pr(X = 0) = (\pi)^k = \left(\frac{k}{k + \lambda} \right)^k = \left(\frac{Ak}{Ak + aN} \right)^k. \quad (13)$$

As for the randomly distributed population, the probability that a unit of area has at least one individual is:

$$\psi = \Pr(X > 0) = 1 - \Pr(X = 0) = 1 - \left(\frac{Ak}{Ak + aN} \right)^k. \quad (14)$$

It is important to note that there are two confounding parameters, N and k , in equations 13 and 14. It is very difficult (if not impossible) to estimate these two parameters simultaneously using the same procedure as described for the randomly distributed population. Because individuals are not randomly distributed, information is needed about their "aggregation pattern". The method proposed below may help to solve this problem.

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1. A study area of size A is divided into A/a cells. We randomly choose a sample with a total of c cells (sampling units) from the area A as our first set of samples and name this sample set as S1.
2. A cell immediately next to each sampled area in S1 is also chosen as our second set of samples, which has also a total of c cells. This sample set is named S2.
3. Detection–nondetection surveys are then conducted in both S1 and S2 sample set cells.
4. Survey results from S1 and S2 are combined as an amalgamated sample set S12. If one or more individuals are detected in either S1 or S2 of a pair of sampling cells, then detection = 1 for that amalgamated sampling cell, otherwise detection = 0 for that amalgamated sampling cell. Because cells in S1 and S2 have the same size and shape, each amalgamated sampling area has a size of $2a$, and therefore the total study area has $A/(2a)$ sampling units.
5. The three samples—S1, S2, and S12—contain information on the common unknowns N , k , and D . A combined pseudo–likelihood function can be then structured as (Breslow and Holubkov, 1997; Wang, 2001; Asparouhov, 2006):

$$L(N, k, D | m_i, n_i, A, a) = \prod_{S=1}^2 \left\{ \prod_{\substack{i=1 \\ n_i > 0}}^{C_{1,S}} \left\{ \binom{m_i}{n_i} D^{n_i} (1-D)^{m_i-n_i} \left[1 - \left(\frac{Ak}{Ak+aN} \right)^k \right] \right\} \right. \\ \left. \times \prod_{\substack{i=1 \\ n_i=0}}^{C_{0,S}} \left\{ (1-D)^{m_i} \left[1 - \left(\frac{Ak}{Ak+aN} \right)^k \right] + \left(\frac{Ak}{Ak+aN} \right)^k \right\} \right. \\ \left. \times \prod_{\substack{i=1 \\ n_i > 0 \\ S12}}^{C_{1,S12}} \left\{ \binom{2m_i}{2n_i} D^{2n_i} (1-D)^{2m_i-2n_i} \left[1 - \left(\frac{Ak}{Ak+2aN} \right)^k \right] \right\} \right. \\ \left. \times \prod_{\substack{i=1 \\ n_i=0 \\ S12}}^{C_{0,S12}} \left\{ (1-D)^{2m_i} \left[1 - \left(\frac{Ak}{Ak+2aN} \right)^k \right] + \left(\frac{Ak}{Ak+2aN} \right)^k \right\} \right\} \quad (15)$$

where $S = 1$ or 2 is the first sample S1 or the second sample S2; other notations are the same as previously described. In this model, we assume the aggregation parameter k is the same for S1, S2, and the amalgamated S12. Equation (12) indicates that the population variance $\text{Var}(X)$ changes as the mean density λ changes. The variation of the aggregation parameter k with changes in λ is a complex issue, which is discussed in relation to the results of the simulations (see discussion).

Step 2 in the above procedure is critical. Since we assume that the two units of areas or cells have the same aggregation parameter k , these two cells should be as close as possible. In practice, this can be achieved by surveys side by side. For example, in fishery surveys with demersal trawl gear, the vessel can tow two trawls of the same dimension. Detection–nondetection data are therefore recorded separately for each trawl and can then be combined to form an amalgamated sample. Alternatively, each randomly selected unit of area or cell can be divided into two sub–cells of equal size. Data are recorded separately for each sub–cell and then combined to form an amalgamated sample.

Simulation design for a randomly distributed population

Let us propose that study area A is divided into 100x100 cells of the same size and shape and the area contains a total of N individuals of the same species. Each individual is assigned to a randomly selected cell, a sample unit equal in area to an area of size a in equation (1). Each cell may contain zero, one, or more than one individual. Since all N individuals are distributed in area A , a sample of cells is randomly chosen without replacement by a sampling rate of SR from these 10,000 cells. Each sampled cell is surveyed m times with a detection probability d on a particular individual. Each survey is considered as a Bernoulli trial: detection = 1 if one or more individuals are detected, and detection = 0 if no detection at all. The probability of detecting any individual (detection at species level) is $D = 1 - (1-d)^{N_i}$ and the probability of detecting none is $1-D = (1-d)^{N_i}$, where N_i = number of individuals in cell i . Finally, model (8) is applied to estimate parameters D and N .

The simulation design is analogous to a factorial experiment where there are four factors: total abundance, sampling rate, detection probability, and number of surveys. Each of these factors contains several levels. We proposed three population sizes: $N = 5,000, 10,000, \text{ and } 20,000$, which results in a mean density $\lambda = 0.5, 1, \text{ and } 2$ individuals per cell. The sampling rate has four levels: $SR = 0.01, 0.05, 0.1, \text{ and } 0.2$, which results in a sample size of $c = 100, 500, 1,000, \text{ and } 2,000$ cells. The detection probability is set at $d = 0.3, 0.5, \text{ and } 0.7$. The number of surveys ranges from $m = 1$ to 9 per cell. Overall, this results in 324 simulation cases and 291,600 records.

Simulation design for an aggregated population

A hypothetical study area A is also divided into 100x100 cells of the same size and shape and the area contains a total of N individuals of the same species. The first 100 individuals are randomly distributed in A , while the remaining $N-100$ individuals are distributed in an aggregated pattern. Before distributing each individual in the area, the mass centre of the individuals already in the area is computed. Each additional animal is distributed according to a normal distribution with a mean at the mass centre and a standard deviation of $A_{N>0} \cdot cv$, where $A_{N>0}$ is the number of cells that have been occupied by one or more individuals, and cv is the coefficient of variance that determines the density of the aggregation. After all N individuals are distributed in area A , a sample of cells is randomly chosen without replacement by a sampling rate of SR from these 10,000 cells, which forms sample S1. Cells immediately next to each cell in S1 are taken as another sample S2. Each sampled cell is surveyed m times with a detection probability d . Each survey is considered as a Bernoulli trial as in the simulation of randomly distributed population. Survey results (number of surveys and whether detection = 1 or 0) from S1 and S2 are combined to form an amalgamated sample S12. Finally, model (15) is applied to estimate parameters $D, N, \text{ and } k$.

We used three aggregation levels: levels 1, 2, 3 correspond to coefficient of variations (cv) of 0.2, 0.3, and 0.4 respectively. A cv of 0.2 represents a highly aggregated population while $cv = 0.4$ more closely resembles a randomly distributed population (Table 12). The detection probability is increased to 4 levels: $d = 0.3, 0.5, 0.7, \text{ and } 0.9$. Other variables are the same as for the simulation of randomly distributed population, i.e., population $N = 5,000, 10,000, \text{ and } 20,000$, sampling rate $SR = 0.01, 0.05, 0.1, \text{ and } 0.2$, and the number of surveys $m = 1$ to 9 per cell.

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Table 12. Known parameters from randomly distributed and aggregated populations. Density = individuals/cell. Occupied mean density = mean density of occupied cells. Occupancy rate = Number of occupied cells/number of total cells (i.e., 10,000).

Population size	Aggregation level	Maximum density	Mean density	Occupied mean density	Occupancy rate
5000	Random	6	0.5	1.3	0.40
5000	Low	7	0.5	1.5	0.33
5000	Medium	10	0.5	1.9	0.26
5000	High	16	0.5	3.0	0.17
10000	Random	7	1.0	1.6	0.64
10000	Low	10	1.0	2.0	0.50
10000	Medium	15	1.0	2.9	0.35
10000	High	25	1.0	4.7	0.21
20000	Random	10	2.0	2.3	0.87
20000	Low	16	2.0	3.2	0.63
20000	Medium	23	2.0	4.3	0.46
20000	High	44	2.0	7.9	0.25

Model performance evaluation

The suitability of Models (8) and (15) is a fundamental concern before adopting the method. The accuracy of an estimator of model parameters is generally determined by two components: bias and variance. The simulation procedures described above result in a total of 291,600 records for a randomly distributed population, and 3,499,200 records for an aggregated population. The biases and standard errors of the estimated parameters are evaluated by comparing them with the known input values. The following statistics are used to evaluate the model's performance.

The primary statistic for bias is the relative error:

$$RE_i = \frac{T_i - F_i}{T_i} \quad (16)$$

where T_i is the true known value of a particular parameter (e.g. abundance) for a simulation case i , while F_i is the model fitted (estimated) value for that parameter. From RE_i , we have the mean relative error:

$$MRE = \frac{1}{l} \sum_{i=1}^l RE_i \quad (17)$$

for a certain group of simulation cases (e.g. average over all surveys). In this formula l is the number of simulation levels (e.g. survey has 9 levels).

From this, the mean absolute relative error is used:

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$$MARE = \frac{1}{l} \sum_{i=1}^l |RE_i|$$

(18)

Approximate standard errors (SE) of the estimates are used for evaluating uncertainty of model output. These approximate standard errors of the estimates are the square roots of the diagonal elements of the covariance matrix of the parameter estimates, which is the same as the inverse of the Hessian matrix (the matrix of second derivatives) of the likelihood. Again, a relative measure is used because it is more conducive to making comparisons across different simulation cases:

$$cv_i = \frac{SE_i}{F_i}$$

(19)

Recall that the main objective of this paper is to estimate total population size using detection–nondetection data that are obtained from sampling a portion of an area of interest with a detection probability < 1 . Note that the errors in the estimated abundance contain not only errors due to the models themselves, but also the sampling errors that occur because only a fraction of the population is included in the sample. Since the true number of individuals taken in the samples is known, sampling errors can be easily derived. Sampling errors are equivalent to the bias between the estimated abundance and the true abundance when the estimates are derived from perfectly (i.e. detection probability $d = 1$) collected count data, in contrast to detection–nondetection data. Absolute sampling errors are presented as a comparison with the total absolute relative errors of the estimates.

Results

The simulation for the randomly distributed population contains 4 sampling rates, 3 specified population sizes, 3 detection probabilities, and 9 levels of surveys. This results in a total of 324 simulation cases. The simulation for the aggregated population contains 3 levels of aggregations and 4 detection probabilities. Other factors and levels are the same as in the design for the random distribution (i.e., 4 levels of sampling rates, 3 levels of population sizes, and 9 levels of surveys). This results in a total of 1,296 simulation cases. To facilitate comparison between simulation cases, the results are added together for randomly distributed and aggregated populations.

Overall, model (8) yields good estimates for the randomly distributed populations, whereas model (15) produces viable estimates for the aggregated populations but are less accurate than the randomly distributed population. As expected, the levels of each factor chosen for simulation affect the outcomes. Generally, an increase in sampling rate SR , detection probability d , or the number of surveys m results in more accurate estimates. Absolute relative error < 0.01 for the random distribution pattern and < 0.05 for the aggregated pattern were achieved in many cases when the sampling rate, detection probability, and the number of surveys were sufficiently high. This is most clearly illustrated in Figure 21, where a population size of 10,000 and a detection probability of $d = 0.5$ were used. The estimated total abundance from models (8) and (15) contains two major sources of uncertainty: one purely due to sampling errors and the other due to the method itself, more specifically using detection–nondetection data with a detection

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probability < 1 and models (8) and (15) themselves. As the sampling rate increases, both sampling errors and modeling errors decrease. To examine the effect of certain factors on the estimation, in the following sections simulation cases are combined for other factors. Also, results for the case where all cells are surveyed only once ($m = 1$) are excluded because of their large error.

Sampling rate SR . Increasing sampling rate, or equivalently, increasing the number of cells in the sample, may reduce the bias (Figure 22). For the randomly distributed population, the relative error of abundance estimate almost linearly declines from $SR = 0.01$ to 0.1 . However, the decline in the relative error becomes insignificant after SR reaches 0.05 and there is little improvement from $SR = 0.1$ to 0.2 . For the aggregated population, estimation is poor when the sampling rate is low ($SR = 0.01$). Interestingly, an increase in the sampling rate beyond $SR = 0.05$ does not considerably reduce the error. Overall, sampling errors are about 39% ($SD = 33\%$, $n = 128$) as large as the total absolute relative errors in the abundance estimation.

Detection probability d . Increase of detection probability generally reduces estimation error (Figure 23). When $d = 0.5$ the error is much smaller than when $d = 0.3$. However, in both the randomly distributed and aggregated populations accuracy is not substantially improved by increasing d beyond $d = 0.5$. Overall, the sampling errors are about 38% ($SD = 21\%$, $n = 120$), as large as the total absolute relative errors in the abundance estimation.

Effect of the number of surveys m . When all cells are surveyed only once, the bias can be very high. It is not uncommon to see over 50% absolute relative errors (ARE) for the randomly distributed population and 100% ARE for the aggregated population. Bias reduces sharply when the number of surveys $m \geq 2$ (for the randomly distributed population) or $m \geq 3$ (for the aggregated population). For the majority of simulation cases, accuracy cannot be substantially improved beyond $m = 3$ for a randomly distributed population and about $m = 6$ for an aggregated population. When detection probability is high ($d = 0.7$ or 0.9), 4 surveys for the aggregated population will result in an accurate estimate when $m > 4$ (Figure 22 and Figure 23).

Total abundance N . The effect of total abundance on accuracy does not show a clear pattern. For example, the overall mean absolute relative error $MARE$ for surveys $m > 4$, which when combined with all other factors is 0.18, 0.12, and 0.16 for $N = 5,000$, 10,000, and 20,000, respectively. Among these errors, the overall $MARE$ of the sampling errors is 0.07, 0.06, and 0.05 for these three population sizes.

Aggregation levels. Estimation errors increase as the aggregation level increases. The abundance of a randomly distributed population is most accurately estimated, whereas the highly aggregated population has the poorest estimation. This pattern can be clearly seen by combining factors of sampling rates, population sizes, and detection probabilities (Figure 24). Combining over 3 levels of sampling rates ($SR = 0.05$, 0.1 , and 0.2), 3 levels of population sizes ($N = 5,000$, $10,000$, and $20,000$), 2 levels of detection probabilities ($d = 0.5$, and 0.7), and 4 levels of surveys ($m = 4$, 5 , 6 , and 7), we obtain abundance $MARE$ of 0.04, 0.08, 0.13, and 0.15 for the random distribution, low, medium, and high aggregations, respectively. For the same aggregation level, $MARE$ purely due to sampling errors is 0.04, 0.03, 0.04, and 0.09. The effects of aggregation on abundance estimates can also be observed from Figure 21, Figure 22, and Figure 23.

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Uncertainty of parameter estimates. Coefficient of variance of the estimated abundance decreases as detection probability d , number of surveys m , or sampling rate SR increases and as aggregation level decreases (Figure 25). The mean cv is small ($< 10\%$) for a randomly distributed population, but very high for a highly aggregated population ($\sim 40\%$).

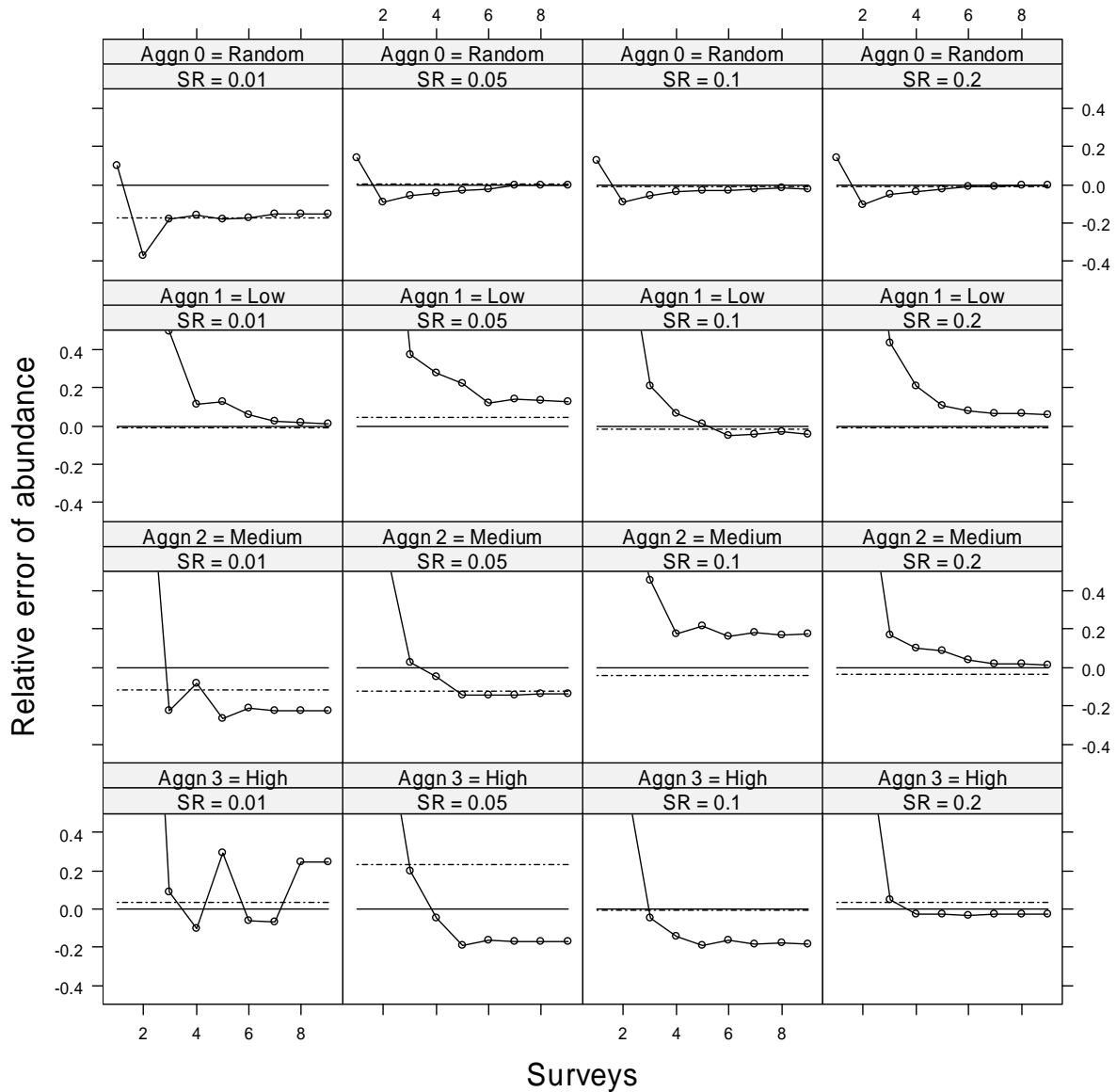


Figure 21. An example of relative errors (RE) of the estimated abundance for randomly distributed and aggregated populations. Total abundance $N = 10,000$ and detectability $d = 0.5$ were used in the simulation. The relative errors due to sampling errors are plotted as broken lines for comparison.

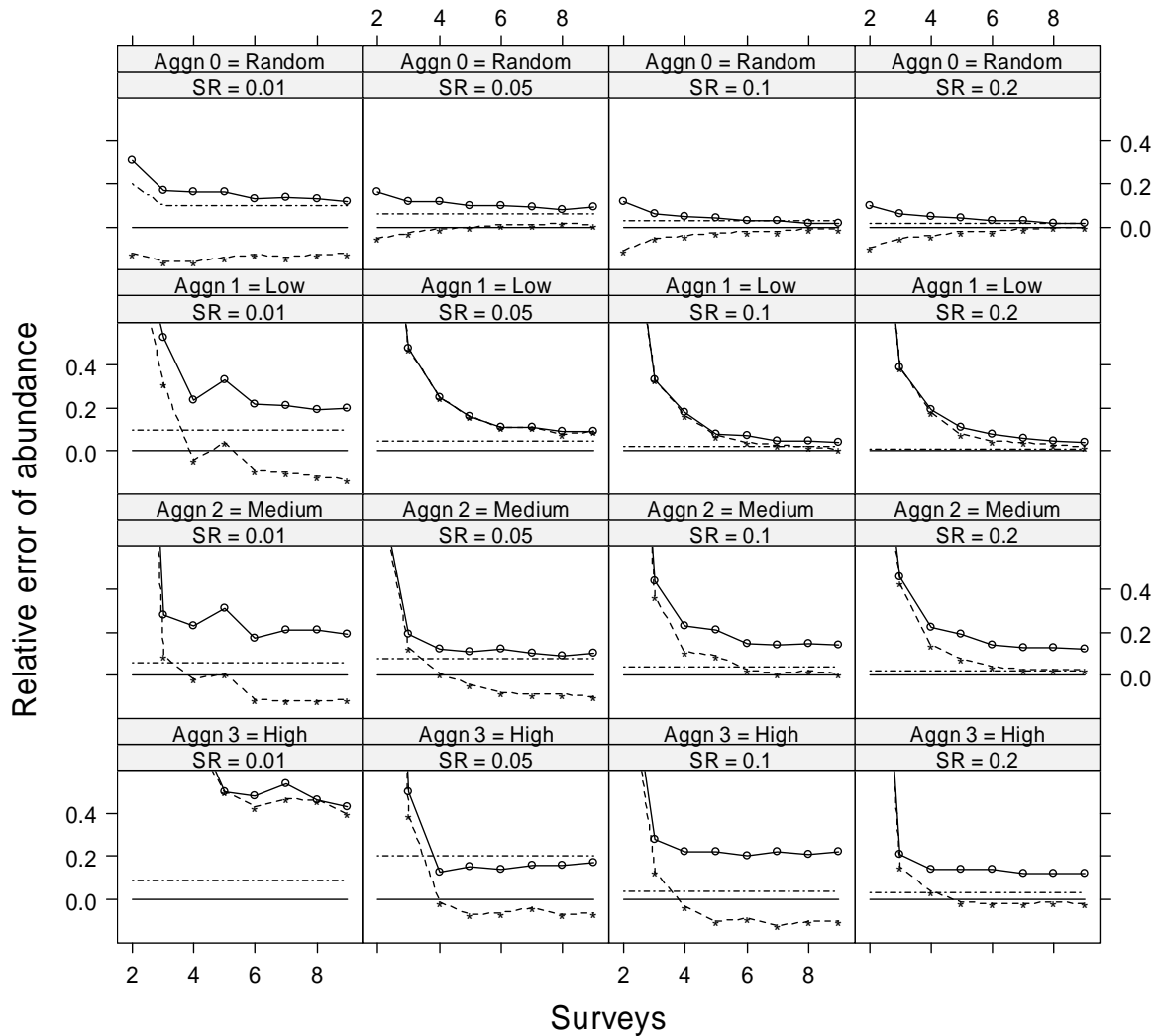


Figure 22. Effect of the number of surveys m (2 to 9 times), sampling rate SR , and aggregation level on the mean relative error (MRE , dashed line with “*”) and mean absolute relative error ($MARE$, solid line with “O”) of the estimated abundance N . Simulation cases are combined over three population levels and three (random distribution) or four (aggregated population) levels of detectability d . Mean absolute relative errors due to sampling error are plotted as broken lines for comparison.

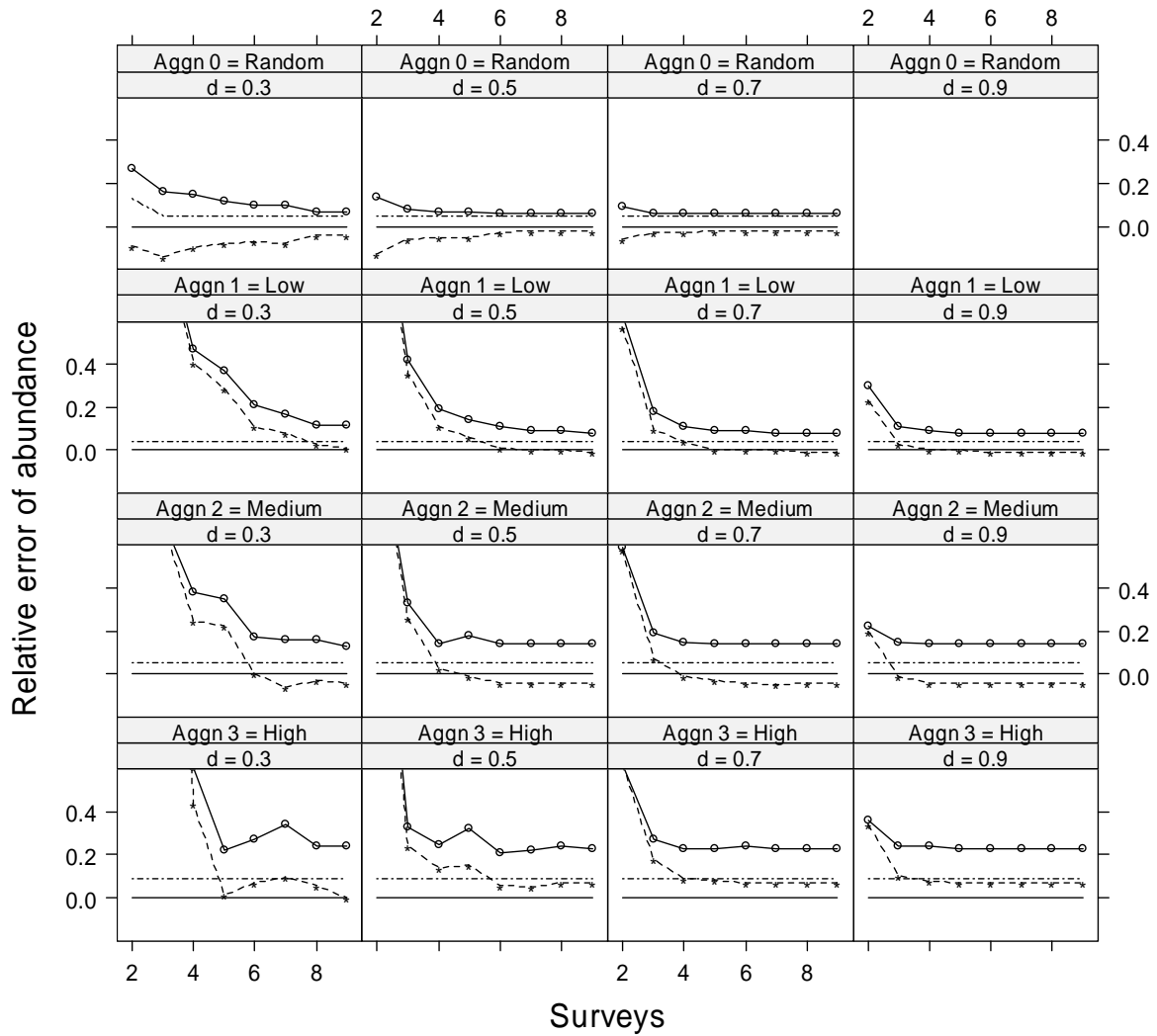


Figure 23. Effect of the number of surveys m (2 to 9 times), detectability d , and aggregation level on the mean relative error (MRE , dash line with “*”) and mean absolute relative error ($MARE$, solid line with “O”) of the estimated abundance N . Simulation cases are combined over three population levels and four sampling rates. Mean absolute relative errors due to sampling errors are plotted as broken lines for comparison.

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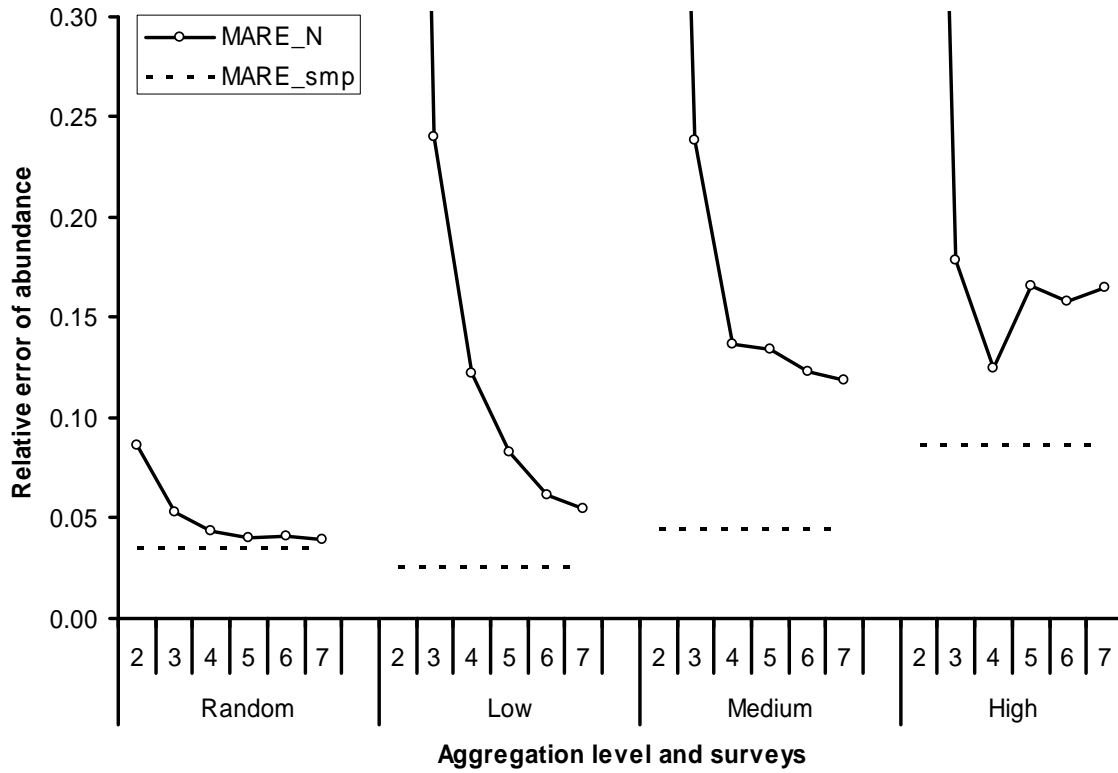


Figure 24. A glimpse of the simulation results of abundance estimation. The simulation cases are combined over 3 sampling rates ($SR = 0.05, 0.1, \text{ and } 0.2$), 3 population sizes ($N = 5,000, 10,000, \text{ and } 20,000$), and 2 detection probabilities ($d = 0.5, 0.7$). $MARE_N$ is the mean absolute error for abundance estimate; $MARE_{smp}$ is the mean absolute error due to sampling errors.

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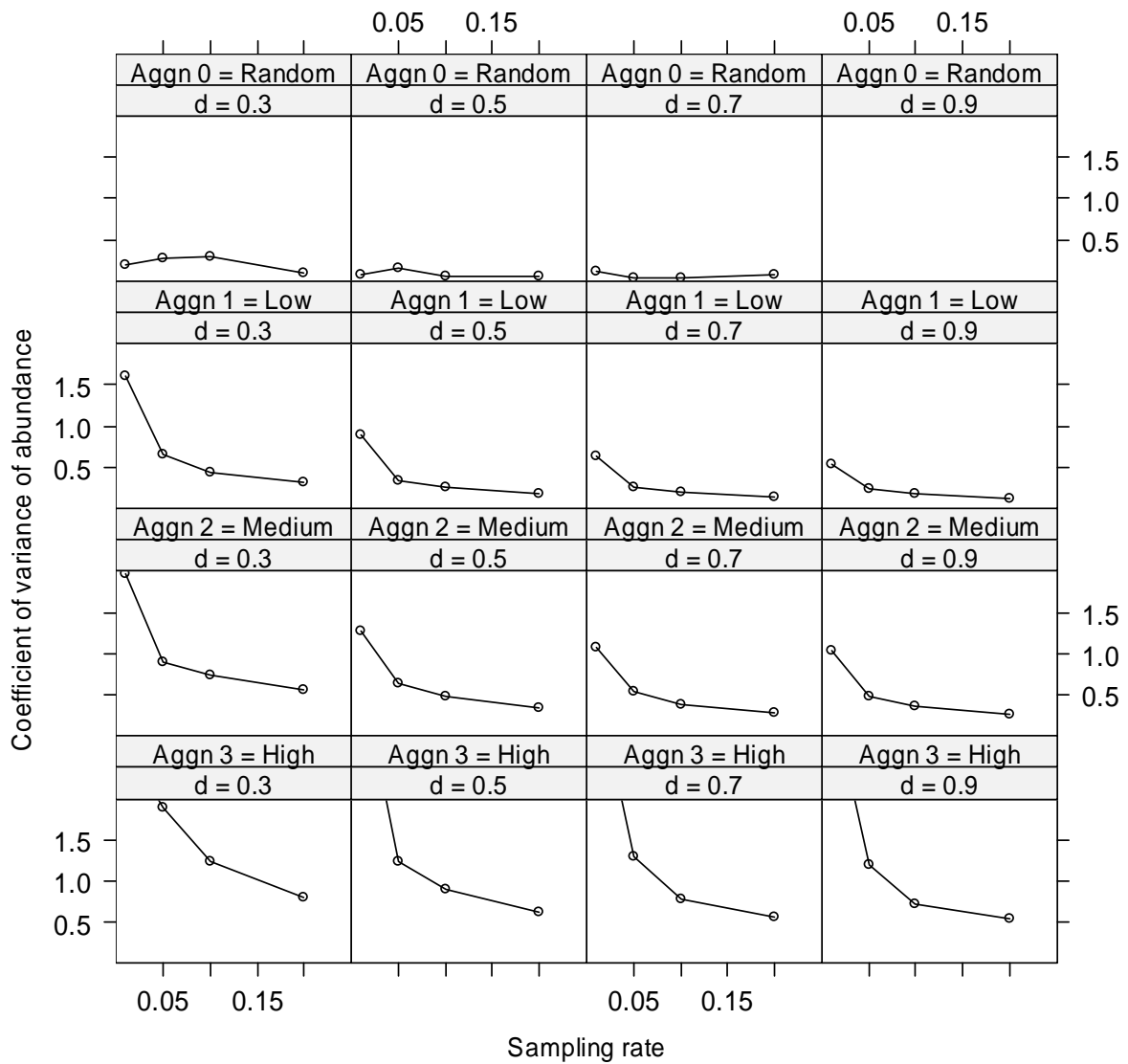


Figure 25. Coefficient of variance of the estimated abundance N when the number of surveys $m = 3$. Simulation cases are combined over three population levels.

Discussion

The challenge of estimating the abundance of species by using detection–nondetection data is well recognised among ecologists, considering the inherent sampling errors and imperfect probability of detecting species or individuals in either randomly distributed or aggregated populations (He and Gaston, 2003). The concept is particularly alluring, since the reduced cost and effort of collecting simple binary data may allow for more rapid and resourceful assessment of species populations. However, the enormous difficulties in achieving this goal are clearly evident by the few published research papers attempting to tackle the problem. Royle and Nichols (2003) first developed a method of estimating abundance from detection–nondetection data when detection is imperfect. Their method considers heterogeneity in detection probability, which is theoretically very attractive but requires integrating likelihood over the space from zero to infinity.

We compared Royle and Nichols' (2003) method with ours by using simulated data and found that their method is flexible and can provide accurate estimates, especially for randomly distributed and low aggregated populations. One advantage of our method is that it does not need integration, which may take a long computing time for abundant species and the computer may not even process a large factorial coefficient. For instance, some fish species have over a hundred individuals in a surveyed cell. For aggregated and abundant species, our method appears to be more stable, less sensitive to initially provided parameter values, and has a smaller bias with an increasing detectability. When modeling avian abundance from replicated counts using mixture models, Kery et al. (2005) pointed out that the mean/variance relationship implied by the negative binomial model may be extreme, which has an especially deleterious effect on estimation of very abundant species and those for which considerable variation is indicated. For a randomly distributed population, the results from the two methods are similar in many situations. We have employed our method to estimate the relative abundance of many bycatch fish species in an Australian trawl fishery for sustainability assessments, but have not included a description in this paper because of the length. However, as with most mathematical models, there are assumptions and caveats to consider before its application to empirical data, some of which may be difficult to meet in specific situations, while others may be relaxed.

One primary concern of our approach is the assumption of constant probability of detecting a particular species across space or that the probability can be modeled by covariates, which is a common assumption in other studies (MacKenzie et al., 2002, 2003; Royle and Nichols, 2003; Tyre et al., 2003). Models (8) and (15) indicate a constant probability of detecting species D across all sampled cells. From the relationship between detectability d and probability D of detecting any individual, it is clear that the D changes as the number of individuals in the survey cell changes (Royle and Nichols, 2003; MacKenzie and Royle, 2005). However, this assumption may be relaxed to some degree. First, since the relationship between d and D is nonlinear, the relative bias in D will decline rapidly as N increases, so that the magnitude of change will be less than the bias in d itself. Second, although estimates for a single cell are expected to be poor due to unknown local abundance, the method can produce very good estimates of the total or average abundance. This has been demonstrated by our simulation results.

It is important to state that our method requires repeated surveys over time at the same sampling areas. It is an implicit assumption of the model that the number of individuals in a particular cell

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should remain the same in sequential surveys. This is similar to the assumption that the population is closed (MacKenzie et. al., 2002, 2003; Royle and Nichols, 2003; Tyre et al., 2003; Royle, 2004; Kery et al., 2005; MacKenzie and Royle, 2005; Royle and Link, 2005; Royle et al., 2005). For mobile species such as fish, this assumption will almost certainly be violated. However, if the assumption is that the total population does not change during the repeated surveys, movement of individuals from one cell to another may not be a serious problem, since Models (8) and (15) combine samples from all surveyed cells. When an individual moves randomly from one location to another, the probability of detection D at the previous location reduces while D at the new location increases, thus negating the effect of change in each other. Further study is needed to test this hypothesis.

A second issue requiring consideration before using our method relates to its application to species with aggregated distributions. Our method involves two samples of smaller area sizes and then amalgamates these two samples to form a third sample that has a unit of area twice as large as the first two samples. Because the two samples of smaller area are adjacent to each other, it is assumed that aggregation patterns within these two different sampling scales are the same, that is, they share a common parameter k . From the mean (Equation 11) and variance (Equation 12) formula of negative binomial distribution, we derive:

$$k = \frac{(E[X])^2}{V[X] - E[X]}$$

Mean $E[X]$ and variance $V[X]$ of abundance differ at the two sampling scales. It has been argued that because k is the function of the mean and the variance, k must change with the scale when the quadrants are amalgamated (Kunin et al., 2000). On the other hand, this formula also indicates that k could remain unchanged after amalgamation because the mean and variance change at the same time. Kunin et al. (2000) gave two extreme examples, but also recognised that between these extremes, there should be an intermediate level of spatial autocorrelation where k remains constant as cells are amalgamated. Simulation results of our study show that k remains constant if the corresponding cells in sample 1 and sample 2 are chosen to be adjacent (Table 13).

Table 13. Mean, variance, aggregation parameter k and its standard deviation before and after amalgamating of the samples. S1 and S2 are the two original samples, S12 is the sample after amalgamation of S1 and S2.

Aggregation level	E[X]		Var[X]		k		SD[k]	
	S1, S2	S12	S1, S2	S12	S1, S2	S12	S1, S2	S12
High	1.16	2.33	13.13	50.20	0.14	0.14	0.01	0.01
Medium	1.14	2.28	5.71	20.23	0.37	0.37	0.10	0.07
Low	1.14	2.28	2.83	9.05	1.03	1.00	0.17	0.14

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It is not necessary to limit negative binomial distribution (NBD) to aggregated populations. NBD describes an entire spectrum of spatial patterns, from regular to random to aggregated populations (He and Gaston, 2000a, 2000b). Nevertheless, for the randomly distributed population, a model based on Poisson distribution (Model [8]) is accurate enough for abundance estimation, while NBD (Model [15]) shows no improvement in the estimates over model (8) (result not presented). Model (15) requires two initial samples with corresponding cells next to each other. Thus, it is neither helpful nor cost-effective to use Model (15) for a randomly distributed population.

When applying our proposed method, the importance of sampling scale needs consideration (He and Gaston, 2000b). Generally, more sampling units with small cell sizes are preferred over fewer sampling units with large cell sizes. For instance, Model (8) or (15) would not perform reliably if the cell sizes are so large that a species is detected in every cell. However, this may not be a significant concern in field studies of wide-ranging species, since limiting resources will generally preclude large sampling units being used in the survey design.

Our simulation results reveal that the reliability of abundance estimates gradually deteriorates from random distribution to low aggregation to high aggregation. It is conceivable that if individuals are extremely aggregated, the method may not provide useful abundance estimates. This problem is not unique to our method, but also to classical sampling with count data (Thompson, 1992). Further research on this problem is needed, such as possibly using nonrandom sampling techniques to collect data and then modelling them accordingly. Also, the aggregation described in the simulation is one possible nonrandom distribution pattern. The method proposed may also apply to other types of aggregation patterns, such as multiple clusters of different sizes. Further study should investigate the accuracy of using this method for all types of aggregation, including aggregation patterns, cluster sizes and distributions, and aggregation levels.

In conclusion, from the results of our model simulations we provide general recommendations for ecological studies that propose to estimate abundance from detection–nondetection data. If the species is believed to be randomly distributed, the survey design should have a sampling rate of no less than 0.05, use a survey method or gear that has a detection probability of no less than 0.5, and survey each sampled area at least 3 times. If the species is believed to have an aggregated distribution, the survey design should have a sampling rate of at least 0.1 (0.05 for each original samples), use a survey method that has a detection probability no less than 0.5, and repeat the surveys four to six occasions per sampled area.

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5.4.4 Sustainability Assessment for Fishing Effects (SAFE): an application to diverse elasmobranch bycatch in a tropical Australian prawn trawl fishery

S. Zhou and S. Griffiths

Abstract

We present a quantitative approach to the ecological risk assessment of diverse and data-poor bycatch assemblages impacted by fishing. We refer to this method as a Sustainability Assessment for Fishing Effects (SAFE). The method uses detection–nondetection data to estimate fishing impact, and compares the impact with sustainability reference points based on life–history parameters. We demonstrate the effectiveness of this method by assessing the impact between 1979 and 2003 of the tiger prawn fishery in northern Australia on the sustainability of 51 elasmobranch bycatch species. Fishing mortality rate was estimated from relative abundances of species in trawled and untrawled areas, catch rate, and the probability of escaping the fishing gear. To estimate the proportion of the population distributed within trawled areas, the fraction of the broader managed area being trawled was calculated from logbook data and coupled with detection–nondetection data collected from scientific surveys over 24 years. This estimate of species' abundance was then included in a model incorporating catch rate and escapement to give an estimate of the fishing mortality rate of the species. To guide management of bycatch species, we established two reference points based on natural mortality rate and growth rate: maximum sustainable fishing mortality rate and minimum unsustainable fishing mortality rate (which renders the population extinct). The annual impacted area (fishing effort ≥ 5 boat days) was estimated at about 6% of the fishery's managed area in recent years (1999–2003). The proportion of the 51 species' populations distributed within the fished area ranged between 0.02 and 1.00 with a mean of 0.36 (SD 0.31). Our results indicated that fishing impacts may have exceeded the minimum unsustainable fishing mortality rate for five species. However, the estimates were highly uncertain for these species. SAFE can also be used by fisheries to focus monitoring programs on potentially at-risk species to obtain additional data for further sustainability evaluation by more formal stock assessment techniques. Because the framework of SAFE is similar to the management of target species, it can be easily incorporated into existing fishery management strategies to fulfil ecosystem–based fishery management objectives.

Key words: Ecological risk assessment, bycatch, fisheries management, sustainability, detection, nondetection

Introduction

Ecosystem approaches to fisheries management are being developed worldwide to conform to increasingly stringent environmental and fisheries legislation. There is also a growing body of evidence that fishing activities have adversely affected other components of the ecosystem being fished, such as the capture of non–target species, “bycatch”, and physical damage to habitats (Hall and Mainprize, 2005). These impacts can lead to changes in biodiversity and ultimately change the overall functionality of the ecosystem (Pitcher and Chuenpagdee, 1994; Crowder and Murawski, 1998; Pauly et al., 1998; Dulvey et al., 2000; Harrington et al., 2005). The Food and Agriculture Organization of the United Nations recently developed technical guidelines for ecosystem approaches to fisheries management as supplements to the FAO Code of Conduct for Responsible Fisheries (FAO, 2003). The approaches aim to ensure that future generations will

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benefit from the full range of resources and services that fished ecosystems can provide by dealing with issues in a holistic manner, rather than focusing on target species or certain species groups, which are typically of economic importance or conservation concern.

Although broad management policies and objectives exist for ecosystem-based management (FAO, 2003), translating them into action is very difficult for three main reasons:

- 1) quantitative biological and catch data on all species in the ecosystem are usually too few to be useful in conventional population models or to assess the status of these populations under existing fishing regimes (Griffiths et al., 2006). This problem is exacerbated in tropical fisheries where non-target species and communities, such as the bycatch, can be species-rich, low economic value, rarely caught, or prohibited from capture such as protected or endangered species (Stobutzki et al., 2001a).
- 2) there are currently no quantitative methods that can incorporate the complexity of ecological interactions of entire ecosystems without intricate knowledge of every species, guild or ecological functional group in the system. Trophic mass-balance models, such as Ecopath (Polovina, 1984), are designed to undertake complex ecosystem analyses. But these models are data intensive, which restricts their use in data-poor fisheries. Furthermore, ecosystem models have not yet been proven as management tools, particularly in terms of making realistic predictions about the future state of an ecosystem (Mace, 2001).
- 3) there is a lack of clear and practical management objectives for non-target species in fisheries, such as the widely used Maximum Sustainable Yield (MSY) for target stocks.

As a result, fishery scientists and managers often do not have information they need to properly assess fishery impacts on non-target species and communities, and to develop management measures to ensure the fishery operates in an ecologically sustainable manner. To this end, ecosystem-based fisheries management is an important concept, but is currently not practicable. To move closer towards fulfilling the broad objectives of ecosystem-based fisheries management, approaches need to be developed that can cope with the diversity and limited data typical of many fisheries worldwide.

In Australia, new legislation and market drivers are forcing export fisheries to account for the broader impact of their operations on the ecosystem as a whole, such as the mitigation of bycatch and the conservation of biodiversity. The Australian Oceans Policy and the Environmental Protection and Biodiversity Conservation (EPBC) Act 1999 now require fisheries to demonstrate that they are operating in an ecologically sustainable manner. In the same vein, public pressure and market drivers are further influencing fisheries to demonstrate that they are ecologically sustainable (Aslin and Byron, 2003); for example by using Turtle Excluder Devices (TEDs) in trawl fisheries (Tucker et al., 1997; Brewer et al., 2006) and ‘dolphin friendly’ methods in tuna purse-seine fisheries (Hurwitz, 1995; Brown, 2005).

The Northern Prawn Fishery (NPF, Figure 26) is one of the first fisheries in Australia to attempt to tackle the challenge of demonstrating the ecological sustainability of its supporting ecosystem through ecological risk assessments (ERA) (Milton, 2001; Stobutzki et al., 2001a; 2002). The NPF, at about \$70 million a year, is the second most valuable Commonwealth-managed fishery in Australia. The fishery targets six prawn (shrimp) species with twin demersal trawl gear and operates as three spatially and temporally distinct sub-fisheries. The banana prawn fishery

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targets aggregations of *Peneaus merguensis* during both day and night over six weeks from April to June. Because the aggregations are often visible from “spotter planes” or on vessels’ echo sounders, they are often targeted with short trawls — generally less than 30 min. The much smaller red-legged banana prawn fishery operates only in the Joseph Bonaparte Gulf area. The main trawl impact is from the tiger prawn fishery, which primarily targets *P. semisulcatus* and *P. esculentus* only during the night from August to November. Because tiger prawns are generally more widely dispersed, the trawls are generally longer (~3 h), which often results in large catches of unwanted bycatch, most of which is discarded dead. Pender et al. (1992) estimated that around 33,000 t of bycatch was discarded annually in this fishery. The bycatch, which often comprises more than 95% of the catch, is diverse, including invertebrates (234 taxa), teleosts (366 spp.), elasmobranchs (51 spp.), sea turtles (8 spp.) and sea snakes (13 spp.) (Stobutzki et al. 2001a, Griffiths et al., 2004).

An innovative, semi-qualitative, attribute-based ecological risk assessment technique was developed concurrently by Stobutzki et al. (2001a, 2001b) and Milton (2001) and was applied to teleost, elasmobranch and seasnake bycatch in the NPF. The relative sustainability of bycatch species was examined by ranking species with respect to their susceptibility to capture; mortality due to prawn trawling; and capacity to recover once the population becomes depleted. The method is revolutionary since it requires only limited empirical data, has the capacity to assess the relative sustainability of hundreds of species, and is transferable to all types of fisheries (Hobday et al., 2006). It has been widely adopted by many fisheries in Australia (Stobutzki et al., 2001b, 2002; EA, 2002; Gribble et al., 2004; QDPI, 2004) for these reasons.

The Stobutzki et al. (2001b, 2002) and Milton (2001) method has, however, severe drawbacks. It is a ranking method, so provides only a *relative* measure of risk for each species, and gives no indication of whether the populations of the high-risk species are truly unsustainable, or the low-risk species are truly sustainable. Furthermore, the term “risk” and “sustainability” are not clearly defined, thus providing no basis on which to assess the status of individual bycatch species. Griffiths et al. (2006) recently demonstrated that this method is not sensitive to changes in the size selectivity of species as a result of changes in fisheries management strategies, and can inadequately reflect even the most obvious changes in risk to individual species. Clearly, for fisheries to demonstrate they are operating in an ecologically sustainable manner and conform to existing legislation or policies, a method is required that can quantify the level of fishery impact on individual species and assess the sustainability of the population, using practical and sensible biological reference points.

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

Materials and methods

Scientific surveys

Over 70 scientific voyages have been undertaken in the Northern Prawn Fishery (NPF) managed area between 1979 and 2003, mostly by CSIRO Marine and Atmospheric Research and a few by state fisheries agencies. They covered the entire NPF, although not in any one voyage (Figure 26). To assess the distribution of elasmobranch bycatch species in the NPF, we collected the data from logbooks and these surveys, which used a variety of sampling methods. Where no catch was recorded, we could not estimate abundance by conventional techniques. As this is a common problem in fisheries worldwide, we pooled the data from all scientific surveys to maximize the sample sizes and geographical coverage, but used detection–nondetection information to estimate bycatch species' distribution in the region.

In order to define the trawled and untrawled areas in the fishery and to model the abundance of bycatch species using detection–nondetection data, we defined a sampling unit as a 6 by 6 nautical mile grid, which is currently used in NPF logbooks for reporting purposes. There are a total of 6,963 grids in the NPF managed area. The composition of bycatch species varies spatially within the NPF (Blaber et al., 1990, 1994, Stobutzki et al., 2001b; Section 5.5.2), so we stratified the NPF managed area into five bioregions based on established bioregions for fishes (IMCRA, 1998) and expert opinion (Figure 26). During the surveys, a total of 5,835 samples had been taken in 924 grids, using trawl gear of various types. Some grids were repeatedly surveyed over years. The sampling rate in bioregion 4 was higher than in the other bioregions because it had a higher fishing effort (see below) and consequently was surveyed more often to investigate fishery–related problems.

Fishery logbooks from 1970 to 2003

The NPF operated nearly year–around before 1970, after which fishing seasons were imposed. Since the early 1990s the NPF has operated as three temporally or spatially separated sub-fisheries. The night–time tiger prawn fishery targets more dispersed tiger prawns and uses long trawls (3–4 h), which results in large volumes of bycatch (Stobutzki et al., 2001a). Therefore, in this paper we consider only the impact of the tiger prawn fishery on elasmobranch bycatch species.

We did not use raw logbook data, since historical logbooks contain errors and missing data. Instead, we used augmented and imputed logbook data from Dichmont et al. (2001).

Estimating fishery impacts

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

Proportion of NPF area fished

The proportion of the fishery's managed area that is trawled (the "fished area"), was taken to be the area comprised of grids where the total fishing effort recorded in augmented logbook data was \geq five boat days between 1999 and 2003. Five days of fishing effort is equivalent to about 10% of sea floor within the grid being systematically swept by prawn trawls in five years, assuming trawling occurs for 12.3 hours per day (Rawlinson, 2003) at a speed of 3.24 knots (Bishop, 2003) with a headrope length of 14 fathoms and a 0.66 spread ratio (Bishop and Sterling, 1999). Because trawl tracks often overlap, the actual impact was expected to be less than 10% (Stobutzki and Pitcher, 1999, Dichmont et al., 2001).

To determine the proportion of the NPF managed area that was fished, we then identified grids where the fishing effort was less than five boat days. This constituted the "unfished area". The ratio between the fished area (A_1) and the total NPF area (6,963 6 x 6 nm grids) was then used to estimate the proportion of the management area trawled (P_A):

$$P_A = \frac{A_{F=1}}{A_{F=1} + A_{F=0}} = \frac{\sum_{R=1}^5 G_{R,1}}{\sum_{R=1}^5 \sum_{F=0}^1 G_{R,F}}, \quad (1)$$

where G_R is the number of grids in bioregion R and subscript $F = 1$ is fished area and $F = 0$ is unfished area. A_F , rather than P_A , is directly fed into the next step for estimating the spatial distribution of bycatch species.

Spatial distribution of bycatch species

The distributions of individual bycatch species and their spatial overlap with the trawled area suggest which species are most likely to be affected by the fishery. Although the true impact of the fishery on a species' population would be best determined by taking into account its entire distribution, our main interest is the local sustainability within the approximate 700,000 km² area of the NPF.

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

Theoretical and field studies indicate that the pattern of presence and absence over a geographic area closely reflects actual animal abundance (Kunin, 1998; Kunin et al., 2000; He and Gaston, 2000a, 2000b; MacKenzie and Kendall, 2002; Nielsen et al., 2005). Estimating the proportion of a geographical area occupied by a particular species from such data has been considered efficient in long–term monitoring programs and metapopulation studies (Azuma et al., 1990; MacKenzie et al., 2004). A particular concern of using detection–nondetection data is the presence of false–negative (or false absence) errors. This can occur if a sample does not capture/detect a species

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when it is, in fact, present. To avoid this, the probability of false–negative errors should be incorporated into models of binary data (Bayley and Peterson, 2001; MacKenzie and Kendall, 2002; MacKenzie et al., 2002; MacKenzie et al., 2003; Tyre et al., 2003; Royle and Nicholes, 2003; Gu and Swihart, 2004), as we have done in our model (2) to estimate abundance.

The scientific surveys in this study spanned 24 years from 1979 to 2003 and samples were collected from 924 NPF grids. We assumed that, after stratification into the five bioregions, individuals of each elasmobranch species were randomly distributed within each region, as well as within the fished and unfished areas. We believe that this assumption was appropriate for tropical elasmobranchs, because they are generally not encountered in large aggregations. The probability that a surveyed grid is occupied by a particular species was held to be directly related to the total abundance of the species in the entire study area. Conditional on the species actually existing in a surveyed grid, it may not be captured in every survey. Therefore, the result for any given survey can be considered as two binomial processes working simultaneously: the probability that a species is present in the grid, and the probability that one or more individuals of that species are captured given the species is indeed present in the grid. Repeated surveys within the same grid allow the estimation of the total abundance, or mean density, of a species in the study area. Because the scientific surveys spanned many years, we needed to assume that the relative abundance of each bycatch species between fished areas and unfished areas remained constant during this period (see discussion). The model of Zhou and Griffiths (in press) has two components: firstly, for grids where one or more individuals were detected in at least one survey; and secondly, for grids where the species was not detected, but may actually be present. For each surveyed grid, assume that a total of m_i surveys have been conducted, of which a species has been captured in n_i surveys ($n_i \leq m_i$). The combined likelihood across all surveyed grids is:

$$L(N, D | m_i, n_i, A, a) = \prod_{\substack{i=1 \\ n_i > 0}}^{C_1} \left\{ \binom{m_i}{n_i} D^{n_i} (1-D)^{m_i-n_i} \left[1 - \left(1 - \frac{a}{A}\right)^N \right] \right\} \times \prod_{\substack{i=1 \\ n_i = 0}}^{C_0} \left\{ (1-D)^{m_i} \left[1 - \left(1 - \frac{a}{A}\right)^N \right] + \left(1 - \frac{a}{A}\right)^N \right\} \quad (2)$$

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

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$$D = \frac{1}{\exp[-(\alpha + \beta\mathbf{M} + \gamma\mathbf{H})] + 1} \quad (3)$$

where vector \mathbf{M} is the sampling gear type, vector \mathbf{H} is the area that each gear covers in each grid, and α , vectors β and γ are model parameters.

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

$$P_N = \frac{\sum_{R=1}^5 N_{R,F=1}}{\sum_{R=1}^5 \sum_{F=0}^1 N_{R,F}} \quad (4)$$

Where $F = 1$ refers to the fished area, whereas $F = 0$ refers to the unfished area.

Fishing mortality rate

The most important metric is the fishing-induced mortality rate (u), which is estimated by:

$$u = P_N q (1 - E) \quad (5)$$

where q is the species-specific catch rate and E is the species-specific probability of escapement from a trawl after a fish enters it. In this instance the catch rate can be considered the probability of a fish entering the trawl along a track. The model (5) implies that we simplified the fishing process to uniformly sweep a grid once a year.

Using commercial logbook data, we estimated that the average fishing effort in the fished areas from 1999 to 2003 (i.e. >5 boat days) was 43.5 boat-days per grid (SD = 46.1, $n = 1,157$). Using the same method and data described for P_A , this fishing effort could systematically sweep the seabed 1.7 times/year (approximately 95% CI 1.2 – 2.2 times/year). Considering other factors (e.g. fish avoidance, overlapping of trawl tracks), equation (5) that assumes a grid is uniformly swept once a year should be justified.

We had catch rate estimates for only five elasmobranch species (see results), so for other species, we estimated the catch rate by one of three options: (a) based on values estimated by Blaber et al., (1990) for the same species; (b) based on values of Blaber et al. (1990) for species having similar vertical distribution, size, and locomotory behaviour, referred to here as “ecomorphotypes” (Compagno, 1990; Bax et al., 1999); or (c) based on related species in the same genus for which catch rates were measured, since closely related species are likely to have similar vulnerability to capture.

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

Uncertainty assessment

The fished proportion of the NPF, P_A , is assumed to contain minimum uncertainty because the daily fishing locations are recorded in fishery logbooks. However, higher uncertainty may exist in the abundance estimates from scientific surveys, catch rates, and the probability of escapement due to TEDs. Since the abundance N and the resulting relative abundance (P_N) are key factors affecting the fishing-induced mortality rate (u), we evaluated uncertainty around these parameters. Approximate standard errors (SE) around N were derived from the square roots of the diagonal elements of the covariance matrix of the parameter estimates. This is the same as the inverse of the Hessian matrix (the matrix of second derivatives) of the likelihood. Variance of A was obtained from variance of N by a delta method (Zhou 2002). Variance of u in equation (5) was also derived by the delta method from variances of A , q , and E . Variances of q and E were calculated from binomial distributions, assuming both capture and escapement from trawl were binomial processes, i.e., $\theta \sim \text{Bin}(n, E[\theta])$, where n is the sample size from field experiments or assumed samples and $E[\theta]$ is the expected probability of capture or escapement estimated from field studies.

Bycatch management reference points

For target fish species, one conceptual and general management goal is to achieve maximum sustainable yield (MSY). However, there are no clear goals or practical guidelines for managing bycatch species. We propose two reference points for bycatch species. The first is the maximum sustained fishing mortality (MSM), which is equivalent to MSY and a fishing mortality rate u_{msm} that corresponds to MSM. This reference point may be too conservative as a constraint for harvesting economically valuable species. The second reference point, or threshold, is the minimum fishing mortality rate that is expected to eventually render a population extinct in the long term, referred to here as u_{crash} . This reference point corresponds to the management objective that the risk of possible extinction of any bycatch species should be avoided.

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

$$\frac{dB}{dt} = r\left(1 - \frac{B}{B_\infty}\right)B - FB = \frac{4m}{B_\infty}\left(1 - \frac{B}{B_\infty}\right)B - FB \quad (6)$$

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

$$F_{crash} = r = \frac{4m}{B_\infty} = \frac{2m}{B_m} \quad (7)$$

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

$$F_{msm} = \frac{m}{B_m} \quad (8)$$

In the above equations, r = intrinsic growth parameter, B_∞ = pristine biomass, m = maximum productivity (equivalent to MSY), and B_m = biomass at which MSM occurs. Corresponding to instantaneous fishing mortality rate, a fraction of population loss is: $u_{msm} = 1 - \exp(-F_{msm})$ and $u_{crash} = 1 - \exp(-F_{crash})$. For bycatch species in the NPF, there was insufficient information to conduct stock assessments to determine these parameters. The intrinsic ability of fish to sustain an extrinsic threat is fundamentally correlated with the life history traits of that species (Charnov, 1993; Jennings, 1998; Froese and Binohlan, 2000; Frisk et al., 2001; Denney et al., 2002; Frisk et al., 2004; Reynolds et al., 2005; Goodwin et al., 2006). Among the many life history parameters that describe the life history strategy of a fish species, natural mortality M has widely been used as surrogate for F_{msy} for target species (Alverson and Peryra, 1969; Gulland, 1970; Quinn and Deriso, 1999). Therefore, in this first method, we set $F_{msm} = M$.

It has been argued that using M as a surrogate for F_{msy} may be risky for some target species (Garcia et al., 1989; Quinn and Deriso, 1999). Thompson (1993) suggested that a fishing mortality rate under $0.8M$ should keep a stock from collapsing in a model containing a depensatory spawner–recruit relationship. Deriso (1982) developed an upper bound for exploitation rates based on the delay–difference model:

$$u_{upper} \leq \sqrt{\frac{1}{\rho l^2 r v}}, \quad (9)$$

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

We obtained natural mortality M and the growth coefficient ρ from the literature and one of five empirical equations depending on data availability:

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

(2) $M = 10^{0.566 - 0.718 \ln(L_\infty)} + 0.02T$; (Quinn and Deriso, 1999);

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

(4) $\ln(M) = 1.44 - 0.982 \ln(t_m)$; (www.Fishbase.org) (10)

(5) $\underline{M} = -\log_e(0.01)/t_m$; (Hoenig, 1983)

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

Results

Fraction of area trawled

The prawn trawling activity was aggregated in a relatively small area, mainly in regions 4 and 5 (Figure 26). During the last five years (1999–2003), the estimated mean annual impacts were 7% and 3% of NPF areas for effort > 0 boat–day and effort > 5 boat–day respectively (Figure 27). Because fishing grounds change every year, the total impact (effort > 5 boat–day) in the last five years was about 6%.

Trawling impacts on abundance distribution of bycatch species

The geographic distribution of the 51 elasmobranch species within fished areas and outside fished areas is shown in Table 14. Eight species were caught only in fished areas: *Orectolobus ornatus*, *Carcharhinus leucas*, *Carcharhinus albimarginatus*, *Squatina sp. A*, *Taeniura meyeri*, *Urogymnus asperrimus*, *Himantura jenkinsii*, and *Rhinobatos typus*, among which, four species were only caught once. Three abundant species were caught in more than 1,000 samples: *Carcharhinus dussumieri*, *Himantura toshi*, *Gymnura australis*.

The relative population of individual species within the fished area, P_N , ranged from 0.02 to 1.00 with a mean = 0.36 and $SD = 0.31$ ($n = 51$; Figure 28; Table 15). Eight species had an estimated P_N of 100% because all were caught in fished areas only. An additional eight species had greater than 40% of populations inside fished areas. However, the majority of species (30) had less than 30% of their population distributed in fished areas. The estimated relative abundance of some species was uncertain due to low detection rates.

Fishing-induced mortality rate

Estimated fishing impacts were reduced after we accounted for probabilities of capture and escapement. Most species (31) had a mean fishing mortality rate $u < 10\%$. Only eight species had an estimated mean $u > 30\%$ (Fig. 4); these species had been shown to have experienced the highest impact, based on their distributions (P_N).

Sustainability assessment

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

Based on natural mortality, the estimated fishing mortality rate at which a bycatch species can sustain the maximum fishing mortality rate (u_{msm}) ranged from 0.08 to 0.68, with a mean of 0.26 ($SD = 0.12$, $n = 51$, Table 15 and Figure 29). The majority of species (30) were estimated to be capable of sustaining fishing mortality rates of between 20% and 40%. Sixteen species had u_{msm} less than 20%, whereas the remaining five species were estimated to be able to sustain a fishing mortality rate greater than 40%. The fishing impacts on ten species may have exceeded u_{msm} (Table 15; Figure 29). These were the same species that had a fishing impact (u) of greater than 45%, except *Sphyrna mokarran*, which had a very low u_{msm} ($u_{msm} = 0.1$, whereas $u = 0.11$ for this species).

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

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Five species had an estimated u greater than u_{crash} : *Carcharhinus albimarginatus*, *Orectolobus ornatus*, *Squatina sp. A*, *Taeniura meyeni*, and *Urogymnus asperrimus* (Figure 30). In addition, five species had an estimated 95% confidence interval (CI) of u that covered the estimated u_{crash} value. On the other hand, only two species had estimated u and 95% CI greater than the point value of u_{crash} .

Discussion

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

The method we propose for sustainability assessment of multiple non-target species, SAFE, has two main components: the fishery impact estimated from simple distributional information and a sustainability benchmark established from life history traits. Because this framework is similar to the management of typical target species, the approach can be directly incorporated into existing fishery management strategies. The ease with which the data can be obtained makes the concept easily transferable between fisheries. Furthermore, fisheries do not need to invest significant resources in undertaking biological studies and comprehensive population surveys for every species interacting with the fishery.

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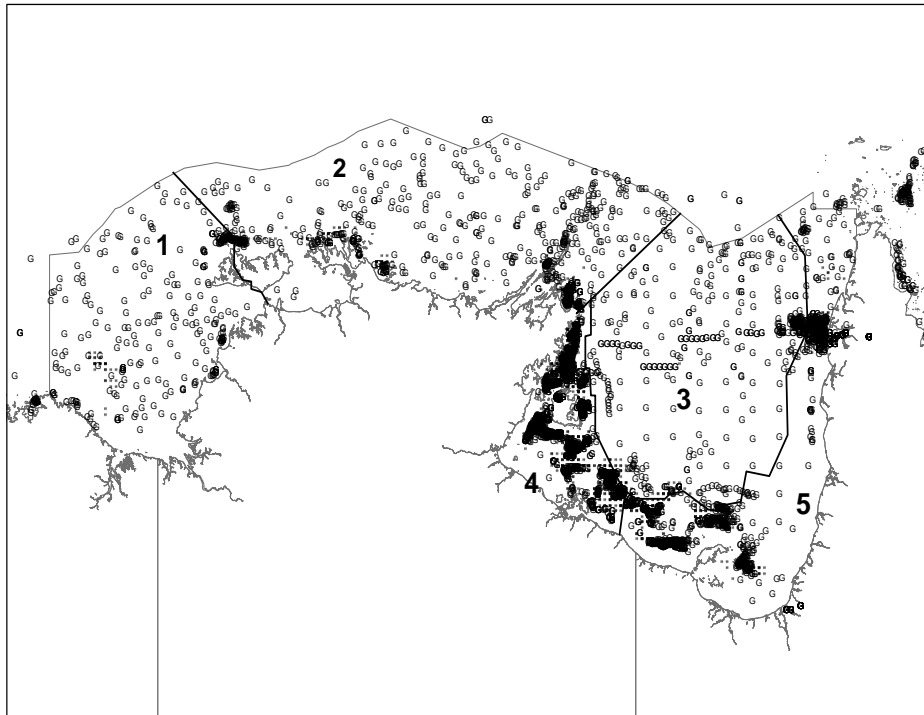


Figure 26. Distribution of samples taken in scientific surveys in NPF from 1979 to 2003 (+) and grids where tiger prawn fishing effort was greater than 5 boat-days from 1999–2003 (■). The NPF managed area is stratified into 5 bioregions based on the bioregions of IMCRA (1998).

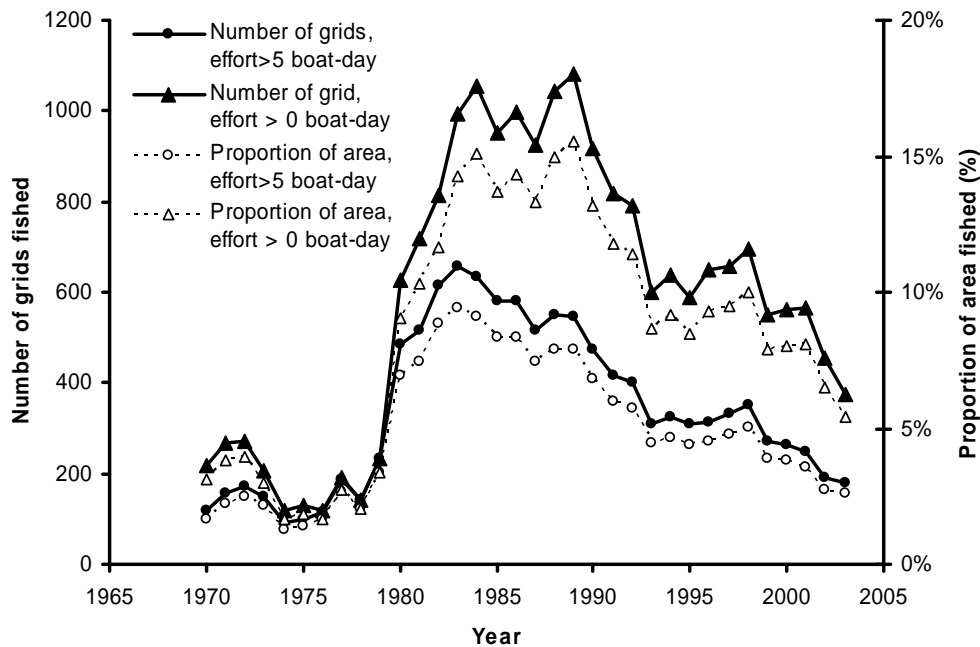


Figure 27. Number of grids and proportion of areas fished under two levels of fishing effort: effort > 5 boat-days and effort > 0 boat-days in the NPF area from 1979 to 2003.

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Table 14. Total observed detections and the number of grids where each species was recorded. Sample size: 4,441 for fished areas and 1,394 for unfished areas. Total grids surveyed: 233 for fished areas and 691 for unfished areas.

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

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Table 15. Estimated trawling impact on bycatch species' abundance distribution (P_N), probabilities of capture q and escapement E used to derive fishing mortality rate u , and comparison with reference points u_{msm} and u_{crash} . Numbers underlined are actual measurements from field studies.

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

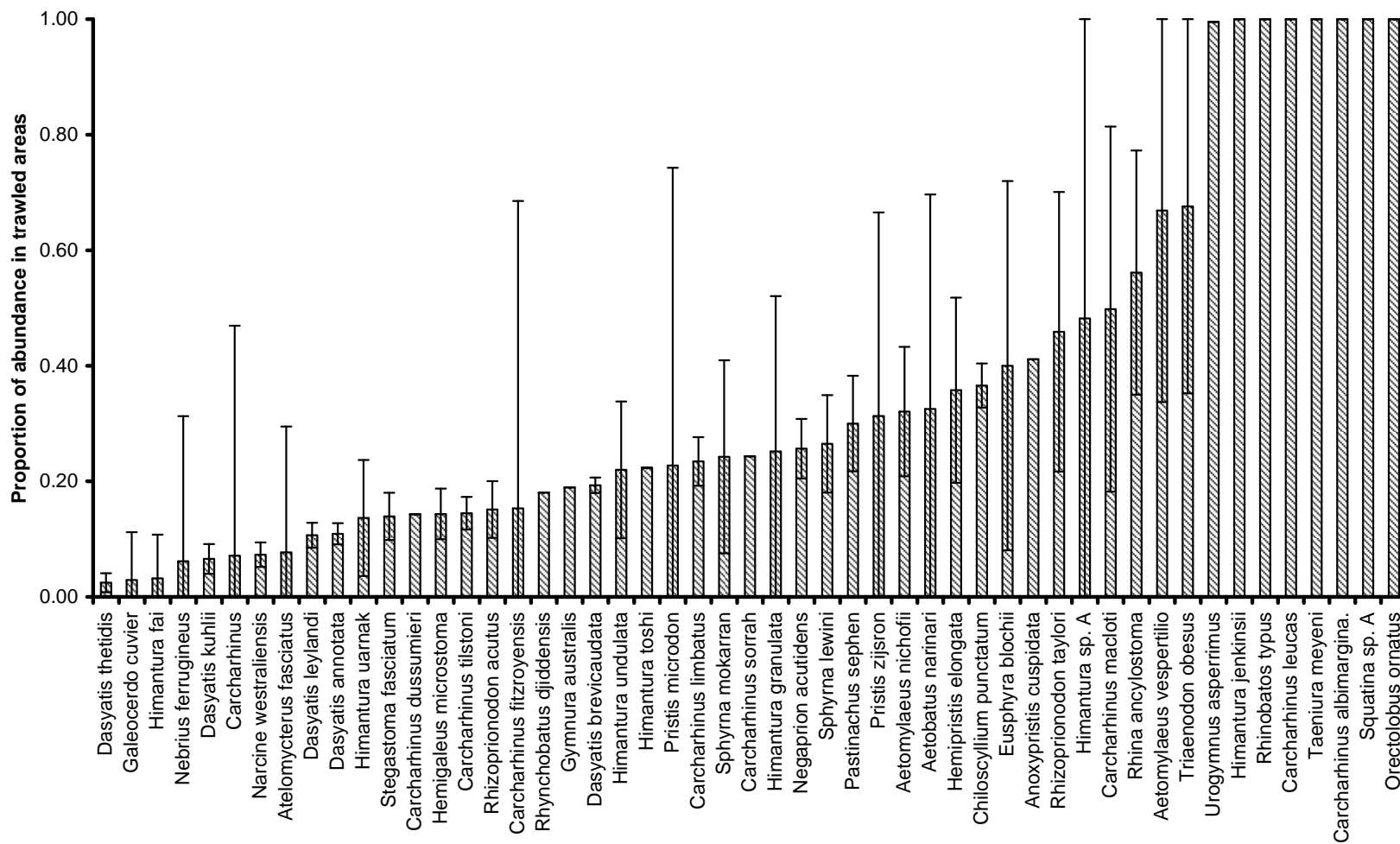


Figure 28. Estimated proportion of abundance within fished areas and 95% confidence intervals for the 51 elasmobranch bycatch species.

Objective 3: Development of a new quantitative risk assessment

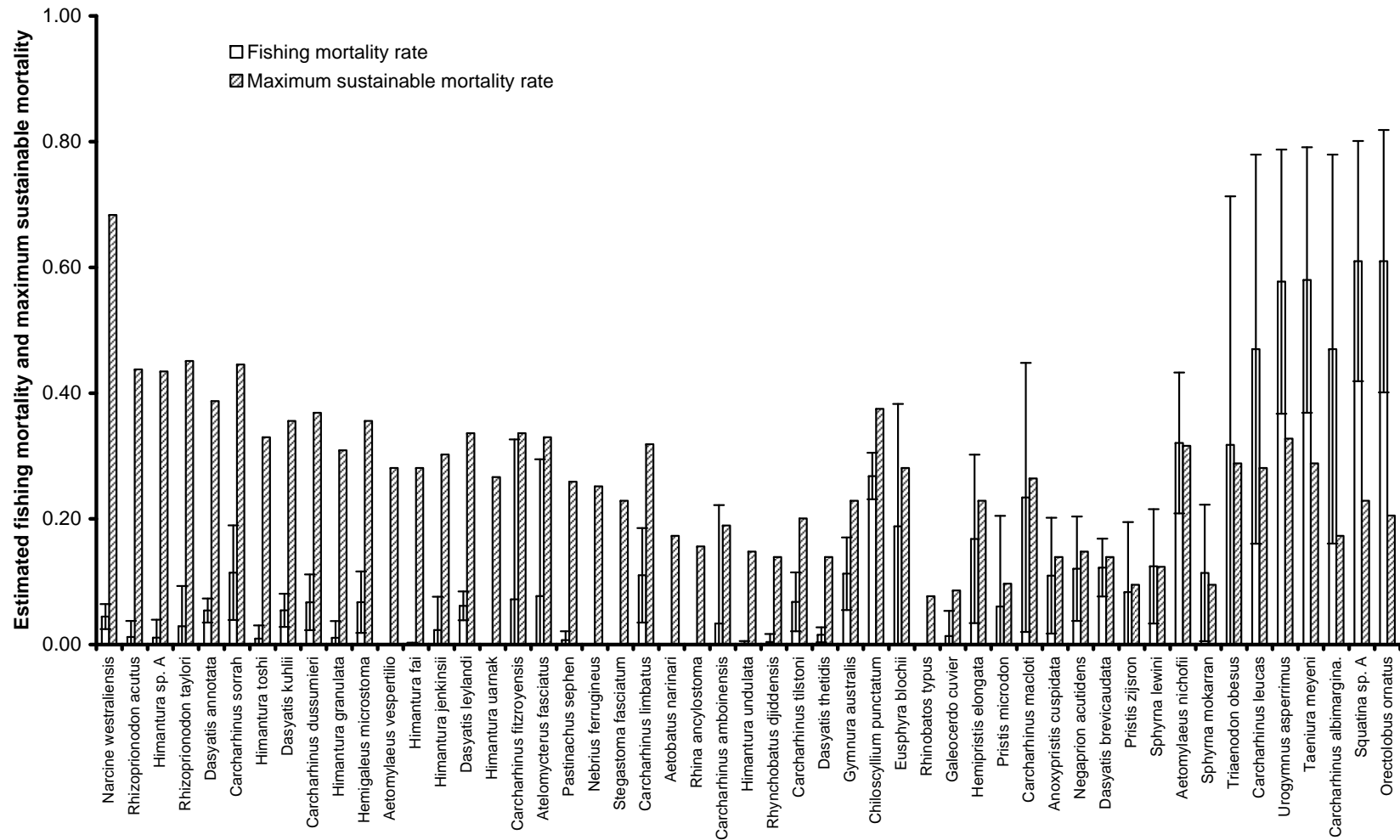


Figure 29. Comparison between estimated fishing mortality rates (u) ($\pm 95\%$ confidence intervals) from prawn trawling and the maximum sustainable fishing mortality rates u_{msm} for the 51 elasmobranch bycatch species.

Objective 3: Development of a new quantitative risk assessment

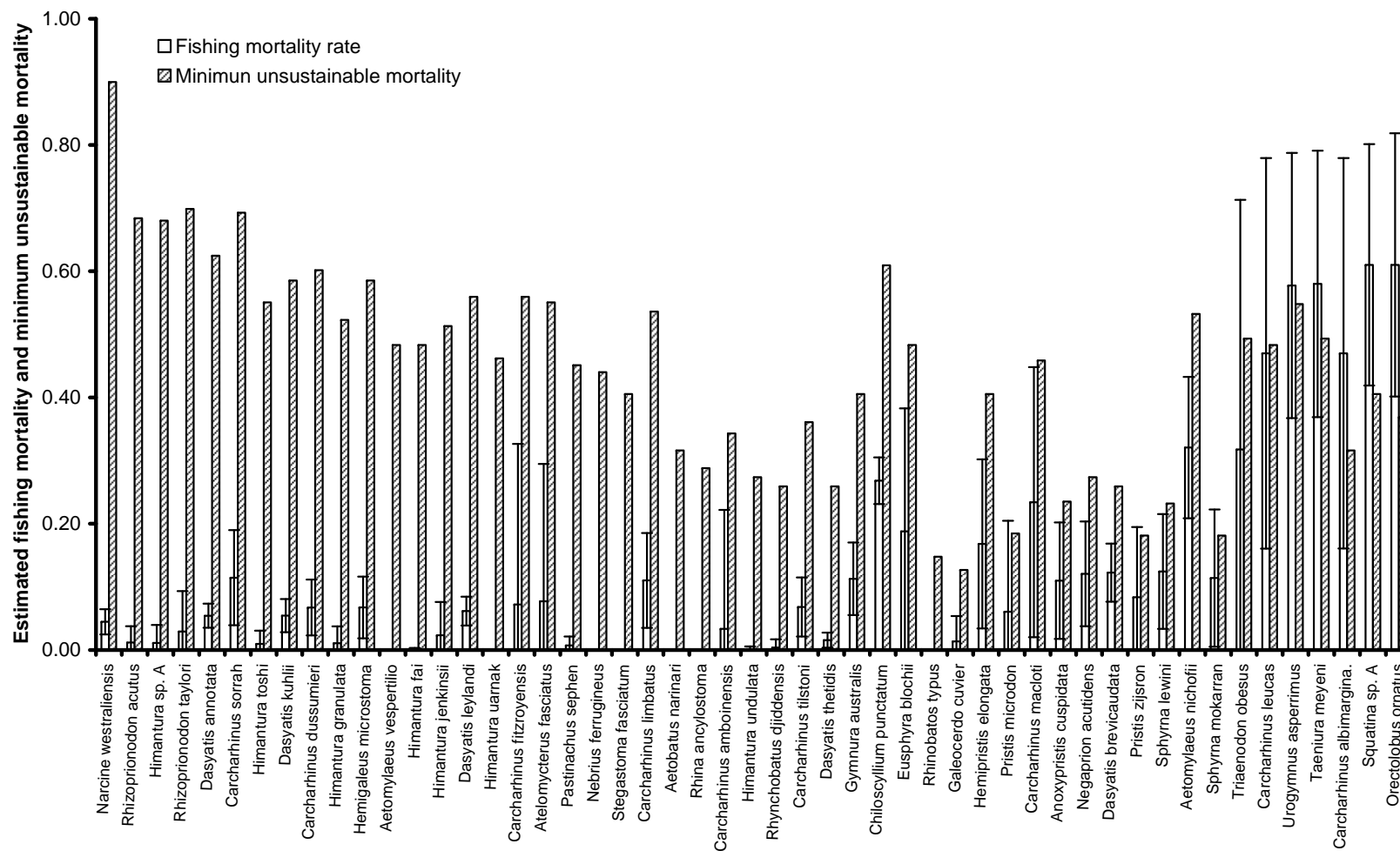


Figure 30. Comparison between estimated fishing mortality (u) from prawn trawling and the minimum unsustainable mortality rate (u_{crash}) for the 51 elasmobranch bycatch species.

Quantifying fishery impacts on bycatch populations

The relative abundance of species inside the fished area (P_N in eq. 4) is an important parameter for SAFE. We use simple detection–nondetection data to estimate abundance. This type of binary data may already have been collected for another purpose, and if more data are needed, the time and cost of collecting detection–nondetection data are lower than for count data. A simulation study demonstrated that estimating abundance from detection–nondetection data has a low bias when a grid is repeatedly surveyed on three or more occasions, the gear efficiency is ≥ 0.5 , and the sampling rate is $\geq 5\%$ (Zhou and Griffiths, in review). Furthermore, model (5) utilizes P_N , the ratio between the abundance of a species inside fished areas relative to its abundance in the total NPF areas area, rather than actual abundance N . Simulation results indicate that this relative quantity is less biased than the actual abundance itself (Zhou, unpublished data). Nevertheless, the actual sampling rate from the scientific survey may have been low. The scientific surveys covered 924 grids in NPF from a total of 5,835 samples, an average of 6.3 repeated surveys per grid. There are a total of 6,963 grids in NPF, meaning the nominal sampling rate is about 13%. However, the fishing gears swept only a small proportion of a grid. Consequently, the actual sampling rate should be less than the nominal sampling rate, which may have contributed to the large uncertainty in the estimated fishery impact on the population of some species.

We use the scientific surveys from 1979 to 2003 to increase spatial coverage and sample sizes. This practice implies that we assume the abundances in fished and unfished areas remain unchanged during the entire study period. If the prawn fishery reduces the population of bycatch species inside fished area, then this assumption could be violated, which would cause the relative abundance inside the fished area in recent years more likely to be overestimated than underestimated because the method uses data collected over a long period to derive relative abundance. Consequently, the recent trawl impact by the NPF on a population may be overestimated.

Our SAFE method uses fishing mortality rate as one ultimate single measure for quantifying the fishery impacts on bycatch populations. We encountered data limitation problems because many bycatch species are poorly studied. Without actual catch and escapement rates for bycatch species, data from previous studies (e.g. Blaber et al., 1990; Brewer et al., 2004) can provide estimates from closely related species, which generally have similar ecomorphotypic characteristics (Compagno, 1990).

Management reference points for sustainability of bycatch

Ecosystem–based fishery management is currently neither well defined nor understood (Brodziak and Link, 2002). Although concepts such as biodiversity, ecosystem integrity and ecosystem function are frequently cited in management policy, they are difficult to interpret at an operational level (Garcia and Staples, 2000; Murawski, 2000; Mace, 2001). Our method for estimating fishing mortality rate is useful and simple, but fishing mortality rate alone does not indicate whether a stock is sustainable or not under current fishing pressure. A metric determining optimal or threshold fishing mortality rate is required for evaluating the biological consequences of the estimated impact. Unfortunately, from a traditional stock assessment point of view, such a threshold requires a substantial understanding of the population, especially the stock–recruitment relationship. Even for many target fish species, there is often insufficient information to undertake formal stock assessment analyses. For data–poor bycatch species, we recommend using reference points or thresholds based on basic life history traits.

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The sustainability of a fish species depends on its ability to withstand the threats that influence their survival. This ability is fundamentally tied to the life history traits of the species. Growth patterns, body size, natural mortality rates, longevity, age at maturity, and reproductive output are often linked (Charnov, 1993; Jennings, 1998; Froese and Binohlan, 2000; Frisk et al., 2001; Denney et al., 2002; Frisk et al., 2004; Reynolds et al., 2005; Goodwin et al., 2006). The life history strategies that evolve have been shown to have a close relationship with the resilience of populations to fishing pressure (Jennings et al., 1999; Rochet, 2000; Frisk et al., 2001).

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

Optimal fishing mortality rate may not be the most appropriate management goal for bycatch species. Ecological sustainability is likely to be more acceptable to multiple users (Garcia and Staples, 2000) since the fishery does not aim to maximise the yield of bycatch, but ensure fishing impacts do not drive species to extinction. A stock is technically overfished when its biomass is lower than a biomass that produces maximum sustainable yield or is fished at a rate where yield–per–recruit is lower than the maximum level. However, such a stock is not necessarily unsustainable (Hilborn, 2002). Ecological risk assessment is a challenging and complex issue, especially in tropical trawl fisheries such as the NPF, where the diversity of bycatch is high but the catch is small and biological data limited. However, using knowledge of basic life history invariants (Charnov, 1993; Frisk et al., 2001; Williams and Shertzer, 2003) to develop biologically sensible management reference points or thresholds for bycatch species based on simple life history parameters is a practical approach.

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

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1998), could sustain a higher fishing mortality if the management objective is to maintain ecological sustainability rather than fishery profits. In the present study we suggested the minimum unsustainable fishing mortality rate (F_{crash}) be used as the threshold for managing non-target bycatch species. This reference point is usually considered less conservative than F_{max} or F_{msy} , but it is more realistic and acceptable to multiple users. Punt (2000) argues that for elasmobranch species where egg-pup production and subsequent recruitment are little understood, F_{max} can in fact be larger than F_{crash} . Therefore, we recommend F_{crash} as the management threshold for data-poor bycatch species.

At risk elasmobranch bycatch species

The results from SAFE indicate that if F_{crash} is used as the primary bycatch reference point in the NPF, the populations of five of the 51 species would be potentially at risk of becoming unsustainable under current fishing effort (*Carcharhinus albimarginatus*, *Orectolobus ornatus*, *Squatina* sp. A, *Taeniura meyeni* and *Urogymnus asperrimus*). These species were rarely recorded, and exclusively within the fished region: 100% of their population could therefore be exposed to trawling in the NPF, although they are also reported to occur outside the fishery (Last and Stevens, 1994). Possibly they were recorded only within the fished area because those areas were the focus of the surveys. Among these five species, *Carcharhinus albimarginatus* is a widely distributed pelagic species and is rarely caught in prawn trawls; the fishing impact on this species is likely overestimated. The other four species are relatively slow-moving benthic species, which are likely to have a high catchability by a trawl; hence they were each assigned the highest relative catch rate of 1. Since these rays are also typically relatively small (< 1 m disc width), they were each assigned a low probability of escaping through TEDs because they are small enough to pass through the spaces between the TED bars (Brewer et al., 2006). Also, the estimated fishing mortality rates of these species contain high uncertainty. Considering uncertainty associated with the estimated reference points, which is not included in this paper, we can only conclude that these species are potentially at risk of being unsustainable. Several other species have high distributional overlaps with the fishery, such as *Carcharhinus leucas*, *Rhincobatus typus* and *Himantura jenkensii* (100% overlap). However, these species are considered to be not 'at risk' because their catch rates are relatively low (0.47 for *C. leucas*) or their escapement is high (100% and 69% for *Rhincobatus typus* and *H. jenkensii*, respectively).

Sawfishes (Pristids) are a group of elasmobranchs that are particularly vulnerable to capture by trawl fisheries, but were not identified by our SAFE model as being potentially 'at risk'. These fishes are currently listed by the ICUN as protected or threatened and have been a significant management issue for the NPF in recent years. Sawfish are a difficult species to exclude from trawl gear, since they are large and slow-moving, which makes them susceptible to capture. Furthermore, because sawfishes have numerous teeth protruding from their rostrum that often get entangled in the meshes, TEDs are not nearly as effective as for other large elasmobranchs, but still result in 73% exclusion for one species (Brewer et al., 2006). However, there is evidence that sawfishes may be benthopelagic species that move into the water column at night (Peeverell, 2005), when NPF trawlers are operating. Nonetheless, we took a precautionary approach and allocated a catchability value of 1 for all sawfishes. The NPF impacts on sawfishes have not previously been quantified and shown to have a relatively minor impact, largely because sawfish populations are distributed across areas inshore and offshore of the trawl grounds (Peeverell, 2005).

This study assessed the impact of the NPF on bycatch species but did not consider the impact of other fisheries in the region that target elasmobranchs or catch them as bycatch. Populations of sawfishes and other elasmobranchs may be sustainable while being exposed to

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the impact of the NPF alone. However, their populations could potentially be at risk from the cumulative impacts of the state-regulated and illegal gillnet fisheries in the region. As a result, there is an urgent need to assess the cumulative impacts of fisheries on elasmobranch populations. Existing qualitative (Fletcher, 2005; Hobday et al., 2006) and semi-quantitative ecological risk assessment methods (*sensu* Stobutzki et al 2001; Milton, 2001; Cheung et al., 2004; Walker, 2004; Hobday et al., 2006; Astles et al., 2006) do not have the capacity to quantify the contribution of each fishery to the total impact on a species' population. However, the estimated fishing impacts in the present study are additive, so our SAFE method has the potential to study the cumulative impacts from fisheries and possibly other anthropogenic activities.

Comparisons with other assessments

The ability to quantitatively identify species 'at risk' is a significant improvement on previous qualitative (Fletcher et al., 2005) or semi-qualitative attribute-based risk assessment methods that can rank ambiguous risk in only relative terms (Stobutzki et al., 2000, 2001b; Cheung et al., 2004; Astles et al., 2006). A previous risk assessment of elasmobranch bycatch in the NPF (Stobutzki et al., 2002) listed the six highest risk elasmobranchs (four being Pristids), none of which were the species listed as 'at risk' in the present study. It is important to note that their assessment was undertaken prior to the introduction of TEDs in the NPF, making direct comparisons with their results difficult. However, Griffiths et al. (2006) reassessed the data of Stobutzki et al. (2002) after incorporating NPF TED exclusion rates and found the highest risk species were the same as for Stobutzki et al. (2002).

Two fundamental differences in the two models probably account for the difference in these results. Firstly, Stobutzki et al. (2002) allocated a rank to a number of criteria that related to the susceptibility of a species being caught by trawling and their ability to recover if the population became depleted. Their method is more data-intensive than ours requiring information for seven susceptibility criteria and six recovery criteria. They took the precautionary approach of giving the highest possible ranking to criteria for which they had no species-specific information. As a result, most of their highest risk species attained this status on the basis of a lack of data. In contrast, our method is less data demanding, requiring only three data types for assessing the fishery's impact, and one or two to assess sustainability. In cases where we had no species-specific data (e.g. catch rate or escapement rate), we substituted values from closely related species within the same genus or ecologically equivalent species ("ecomorphotype" *sensu* Compagno, 1990). We argue this is a valid approach since our model essentially deals only with susceptibility to capture, such as catchability by demersal trawls, and to escapement through TEDs, both of which are largely influenced by the size of the animal, body form and swimming performance. Therefore species may be more easily grouped into similar susceptibility probabilities, such as those suggested by Blaber et al. (1990) for trawl fisheries, rather than the method of Stobutzki et al. (2002) that requires information on criteria ranging from diet preference to diel vertical migration.

Secondly, the SAFE model is based on the simple presumption that if a species distribution does not overlap with the fishery, it cannot be at risk of depletion by the fishery's direct impact. Fishing effort in the NPF is highly aggregated, as in many fisheries: our results show that the actual fished region (i.e. greater than 5 boat days per year) is only about 3% of the entire NPF managed region. There may therefore be large spatial refuges outside the fished region where a species may be largely unaffected by fishing. This is translated into SAFE through a heavy reliance on the estimated spatial distribution of each species to quantitatively estimate the fishery's impact on each species' population. The proportion of the geographic

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distribution of a species' population that is potentially impacted by the fishery is estimated, and then the proportion of the water column that is fished by a prawn trawl in relation to the vertical distribution of the species.

In contrast, previous studies (Stobutzki et al., 2002) did not weigh geographic distribution, or “range”, as a highly important criterion, nor did it use spatial data to assess the fishery's impact in the same way as our model. Other authors have viewed aspects of geographic range in a similar way to Stobutzki et al. (2002), such as using latitude (Dulvey and Reynolds, 2002) and “spatial behaviour strength” (Cheung et al., 2004) as a factor in assessing the sustainability of fish populations. The “range” criterion of Stobutzki et al. (2002) ranks the susceptibility of species according to how many of the 11 high effort fishing regions that the species has been recorded. This ranges from most susceptible (species occurs in <3 regions) to least susceptible (species occurs in >6 regions). However, this criterion does not take into account whether the species is also distributed outside the fished region, as we have done in the present study. Their criterion may ultimately suggest that a species is at less risk (i.e. higher sustainability) if its entire natural geographic distribution extends across a high number of high effort regions.

In conclusion, we think our SAFE model improves on existing qualitative or semi-quantitative ecological risk assessment methods, since it can quantitatively identify species ‘at risk’ from fishing and assess their theoretical long-term sustainability. However, high uncertainty can be associated with many of our model parameters, which is inherent in using detection–non–detection data to estimate abundance (Zhou and Griffiths, in review) and dealing with low economic value, generally rare and inconsistently recorded bycatch species. We advocate that our SAFE ecological risk assessment model would be most effective for fisheries management when used in conjunction with an ongoing monitoring program. Because our model has the capacity to quantify the fishery impact on hundreds of species, which would normally be expensive and logistically difficult to monitor, it may serve as a ‘filtering’ or focussing mechanism, identifying species potentially ‘at risk’ to become candidates for monitoring. Ongoing monitoring of these species would, in time, provide data on the population, including additional biological data, that can be analysed more stringently with conventional stock assessment models.

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5.4.5 Sustainability Assessment for Fishing Effects (SAFE): an application to highly diverse and data-limited fish bycatch in a tropical Australian trawl fishery

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Abstract

Ecological risk assessment is a useful approach for assessing the sustainability of species impacted by fisheries and prioritising management issues. However, applying this to non-target species is challenging due to lack of clear operational objectives and scarcity of biological data. We used a new quantitative ecological risk assessment method, Sustainable Assessment for Fishing Effects (SAFE), to assess the ecological sustainability of 478 individual teleost bycatch species impacted by trawling in Australia's Northern Prawn Fishery. First, we estimated the fishing mortality rate of each species based on its spatial distribution and catch rate. Detection-nondetection data from historical surveys were used for estimating the spatial distribution, while the distribution of the fished regions in the NPF managed area was quantified from logbooks data. Second, given the fishing mortality on each species, we assessed the sustainability of each species by using two biological reference points based on natural mortality: maximum sustainable fishing mortality rate and minimum unsustainable fishing mortality. Despite tropical demersal trawl fisheries impacting many species, our analyses indicated that no species was clearly at risk of being unsustainable. This is attributed to many species being widely distributed into large unfished refuge areas. However, if uncertainty in the estimates is considered, two species, *Dendrochirus brachypterus* and *Scorpaenopsis venosa*, may be at potential risk of being locally unsustainable if the distribution pattern and the current fishing practice in the NPF remain unchanged. Although we demonstrated that the impact of NPF may be sustainable on bycatch populations, the cumulative impact of all fisheries in the region is required to assess the true sustainability of these populations. We discuss how the SAFE method may be extended to address this need.

Key words: Ecological risk assessment, teleost bycatch, prawn fisheries, sustainability, distribution, mortality, detection, nondetection, presence, absence

Introduction

The management paradigm of marine fisheries has traditionally been dominated by a focus on single target species. However, over the past decade the concept of ecosystem-based fishery management (EBFM) has begun to infiltrate the management approaches in many fisheries worldwide. Although the objectives of EBFM are variable and sometimes conflicting (Sainsbury et al., 2000), EBFM is generally considered as a holistic approach to ensuring the sustainability of the ecosystem that support fisheries (Larkin, 1996). Such objectives may be attractive as high level 'motherhood' fishery policies, but they do not provide a practical means in which to manage the balance between optimising fishery yields and maintaining ecosystem integrity and function (Link, 2002). To assess the ecological sustainability of fisheries, ecological risk assessment may provide a critical first step towards achieving ecosystem-based fishery management.

In recent years, a number of ecological risk assessment models have been developed in response to fishery policy and legislation in many fisheries worldwide to assess the wider ecosystem effects of fishing. These consist of qualitative (Fletcher et al., 2005; Astles et al.,

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2006; Hobday et al., 2006) and semi-quantitative attribute-based models (Milton, 2001; Stobutzki et al., 2001b; Cheung et al., 2004; Walker, 2004; Hobday et al., 2006), which have primarily been designed for data-limited fisheries. However, these methods provide only a relative ranking of risk for each species, and cannot quantify the fishery impact on a species or assess a species' long-term sustainability. Furthermore, some of these methods are not sensitive to changes in size selectivity of a fishery and can fail to reflect even the most obvious change in species sustainability due to management intervention, such as the reduction of elasmobranch bycatch by up to 99% in an Australian trawl fishery (Griffiths et al., 2006). As a result, fishery managers can greatly benefit from approaches that can utilise limited data to assess absolute risk.

Despite some currently used risk assessment models being capable of identifying species at highest potential risk from fishing, we recognised that there are very few practical biological reference points for data-limited, non-target species with which to assess ecological sustainability of potentially 'at risk' species in order to conform to legislative requirements (but see Diamond, 2004; Hall and Mainprize, 2004). Fishery scientists and managers are familiar with the biological reference points to assess sustainability of target species, such as maximum sustainable yield. However, no attempt has been made to extend this, or any similar concept, to the wider ecosystem impacted by fishing, such as bycatch. This may be attributed to a lack of biological and catch data for most bycatch assemblages, which can sometimes comprise hundreds of taxa.

A desirable solution for assessing the sustainability of data-poor, non-target species is to develop a simple method that can utilise a minimum of data and be rapidly applied. One simple and feasible approach is to assess the impact and sustainability of each species directly impacted by fishing and implement a management strategy that restricts the impact within sensible limit reference points. Such an approach was recently developed by Zhou and Griffiths (in press) – called a Sustainability Assessment for Fishing Effects (SAFE) – to quantitatively assess the sustainability of 51 data-poor elasmobranch bycatch species in a tropical Australian trawl fishery. They used binary detection-nondetection data to estimate fishing mortality rates, and assessed population sustainability using two reference points, μ_{msm} and μ_{crash} , primarily based on natural mortality estimates.

The main objective of this study was to extend the SAFE method of Zhou and Griffiths (in press) – initially developed for elasmobranchs – to assess the quasi-extinction risk of 478 teleost species impacted by prawn trawling in one of Australia's largest and most valuable fisheries, the Northern Prawn Fishery (NPF). In this paper we define risk as "unsustainable risk"; or the minimum level of fishing whereby a species' population will not be depleted or permanently damaged. A second objective was to develop biological reference points for data-poor non-target species that can be transferable between fisheries and allow management of bycatch impacts at the species level.

Materials and methods

We used data collected from over 70 scientific voyages between 1979 and 2003 in the Northern Prawn Fishery (NPF) managed area. Details of these collections are described by Zhou and Griffiths (in press a). We divided the NPF managed area into 6,963 sampling units of 6 by 6 nautical mile grids. Of the 70 scientific surveys, sampling using trawl gear of various types occurred in 1,380 of these cells and a total of 7,095 samples were taken. Although different fishing gears were used in the scientific surveys, the most frequently used

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gears were the Florida Flyer prawn trawl, the Frank and Bryce fish trawl, and Yankee Doodle 10 Fathom Prawn net.

The distribution of species commonly caught as bycatch varies spatially within the NPF (Blaber et al., 1990, 1994, Stobutzki et al., 2001b; Section 5.5.2). In order to estimate the abundance of individual species in the NPF from presence-absence data using the model of Zhou and Griffiths (in review b), we needed to stratify the NPF into regions to more realistically reflect the natural distribution of individual species. Therefore, we stratified the NPF-managed area into five bioregions based on established bioregions for fishes (IMCRA, 1999) and expert opinion (Figure 31).

In the NPF, the tiger prawn (*P. semisulcatus* and *P. esculentus*) fishery extends from August to November and trawling takes place only during the night. Because the fishery targets dispersed tiger prawns and uses long trawl hours (3-4 h), the bycatch is often large and numerous (Stobutzki et al., 2001a).

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

Fishing-induced mortality rate

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

$$u = \frac{N_I}{N_I + N_O} q(1 - E)(1 - s) \quad (1)$$

where N_I and N_O are the abundance of a species inside and outside trawl areas, respectively; q is the catch rate, E is the escapement rate, and s is the post-capture survival rate. This formula implies that we simplified the fishing process to uniformly sweep a grid once a year. From commercial logbook, we estimated that the average fishing effort in the fished area could systematically sweep the seabed 1.7 times a year (approximately 95% CI 1.2 – 2.2 times/year). Considering other factors (e.g.net avoidance by fish, overlapping of trawl tracks), it is reasonable in a modelling context to assume fished cells are swept, on average, once a year.

The key component of Eq. (1) is the relative abundance of each species that is exposed to trawling, $N_I/(N_I + N_O)$. We used the model of Zhou and Griffiths (in review b) to estimate N_I and N_O from detection-nondetection data. The model assumes that after stratification of the NPF into bioregions, individuals are randomly distributed within fished and unfished area

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within each bioregion, and fish density differs between fished and unfished areas within each bioregion. The probability that a surveyed grid is occupied by a particular species is directly related to the total abundance of the species in the entire bioregion. Because the survey data was collected over a 24 year period, we assumed the relative abundance of each bycatch species between the fished and unfished areas remained constant during the study period. We also assumed that the probability of capture of a particular species remained constant across all surveyed grids within each bioregion, but was specific to the fishing gear used. Fourteen gear types were used in the surveys, each having different species and size selectivity. These gears included a benthic sled, Engels trawl, Engels trawl fitted with codend cover, Florida Flyer benthic trawl, Florida Flyer trawl with codend cover, Florida Flyer trawl with a bycatch reduction device (BRD), Florida Flyer trawl fitted with BRD and codend cover, Florida Flyer trawl with a turtle exclusion device (TED), Frank and Bryce fish trawl, modified semi-pelagic Frank and Bryce trawl, Julie Ann net, modified semi-pelagic Julie Ann net, twin Florida flyer trawl with Texas drop-chain rig, and Yankee Doodle 10 Fathom Prawn net. We used a logistic model to incorporate gear-specific catchability into the model as described by Zhou and Griffiths (in review a).

We obtained species specific catch rates using one of following methods: i) from field studies (Pitcher et al., 2002); ii) based on related species in the same genus for which measurements were made, since closely related species are likely to have similar vulnerability to capture; iii) based on values estimated by Blaber et al. (1990) for the same species ; and iv) based on values of Blaber et al. (1990) but for species having similar vertical distribution, size, and locomotory behaviour, or “ecomorphotypes” (Compagno, 1990; Bax et al., 1999).

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

Uncertainty assessment

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

Management reference points

We modified two biological reference points proposed by Zhou and Griffiths (in review a) for managing non-target, and low economic value elasmobranch species, to be suitable for teleosts that tend to have higher productivity. The first reference point is the maximum sustained fishing mortality (MSM); equivalent to MSY and a fishing mortality rate (u_{msm}) corresponding to MSM. One of the methods that define the reference point is to set $u_{msm} = 1 - \exp(-F_{msm}) = 1 - \exp(-M)$, where F_{msm} is the instantaneous fishing mortality rate. However,

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setting $F_{msm} = M$ is not considered conservative, especially for species of high natural mortality (Garcia et al., 1989; Thompson, 1993; Quinn and Deriso, 1999). In this paper, we set $F_{msm} = \omega M$, where the scaling parameter ω is a function of M of between 0.5 and 1:

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

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

The second reference point or threshold, u_{crash} , is the minimum fishing mortality that eventually renders the population extinct in the long term. According to the Graham-Schaefer production model (Fletcher, 1978; Hilborn and Walters, 1992; Quinn and Deriso, 1999), $F_{crash} = 2F_{msm}$. i.e., $u_{crash} = 1 - \exp(-2F_{msm})$.

We obtained natural mortality M directly from the literature or using available biological parameters to estimate M using the empirical equations:

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

ii) $M = 10^{0.566 - 0.718 \ln(L_{\infty})} + 0.02T$ (Quinn and Deriso, 1999) and

iii) $M = 1.6 k$ (Jensen, 1996)

In these equations, k and L_{∞} are the von Bertalanffy's growth parameters, and T = average environment temperature (in this case 28 °C).

Results

Spatial distribution of bycatch species

Of the 478 teleost species included in this analysis, six and 94 species were caught only in fished and unfished areas, respectively. Except for the six species only caught in the fished area, no species had more than 50% of its population recorded inside the fished area (Figure 31). Most species had less than 30% of their populations recorded inside the fished area. However, considering that less than 6% of the entire NPF managed area is exposed to trawling, this meant that most species had a relatively high proportion of their population distributed within fished areas.

Fishing mortality rate

Prawn trawling does not catch all fish within the fished area, due to their vertical distribution in the water column and avoidance of the gear by a proportion of the fish in the path of the trawl. Estimated fishing mortality rates of individual species ranged from zero to 0.43 with a mean of 0.05 and standard deviation of 0.07. Nearly half (48%) of the species had fishing mortality rate ≤ 0.03 , while 95% of species had fishing mortality rate ≤ 0.20 (Figure 32). Typically, the higher the proportion of a species' population distributed in the fished area, the higher the fishing mortality rate.

Reference points for assessing species sustainability

Maximum sustainable fishing mortality (u_{msm})

Based on natural mortality, the estimated fishing mortality rate at which a bycatch teleost species can sustain maximum mortality (u_{msm}) ranged from 0.10 to 0.93, with a mean of 0.55 ($SD = 0.22$, $n = 474$, Table 16 and Figure 32). Only a few species had a maximum sustainable fishing mortality rate less than 20% or greater than 90%. The estimated fishing mortality for only two species, *Dendrochirus brachypterus* and *Scorpaenopsis venosa*, were found to exceed u_{msm} (Table 16, Figure 33). If uncertainty in estimated fishing mortality rate is taken into account, the 95% confidence intervals of u for 21 species exceeded u_{msm} . Four of these had very low mean estimated fishing mortality rates u (≤ 0.01) but very high uncertainty.

Minimum unsustainable fishing mortality rate (u_{crash})

The estimated minimum unsustainable fishing mortality rate ranged from 0.19 to 0.99 (mean 0.75 SD 0.20) for the 474 teleost species. No species had a mean estimated fishing mortality rate greater than u_{crash} . However, if uncertainty in the estimated fishing mortality rate is considered, the 95% CI of five species exceeded u_{crash} (Table 16, Figure 34). These were *Hemiramphus robustus*, *Lutjanus rufolineatus*, *Parascalopsis tosenis*, *Dendrochirus brachypterus* and *Scorpaenopsis venosa*, although the first three species had very low mean estimated fishing mortality rates u (≤ 0.01) but very high uncertainty.

DISCUSSION

In this study, we took the SAFE approach of Zhou and Griffiths (in review a) to undertake a rapid quantitative species-by-species assessment of the effects of fishing on the sustainability of 478 data-poor bycatch species in the NPF. We demonstrated that SAFE, initially developed for elasmobranch bycatch (Zhou and Griffiths in review a), can be easily applied to highly diverse and data-limited assemblages, and may be easily transferable to other species in other fisheries. This approach can circumvent full stock assessments on large numbers of impacted species by using simple fishery or research data and life history parameters that are relatively easy to obtain or estimate from the literature. Because this framework is similar to the typical management regimes used for target species, the approach can be directly translated and incorporated into existing fishery management strategies.

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Table 16. Parameters of species that have estimated upper 95% confidence intervals of fishing mortality rates greater than the maximum sustainable fishing mortality rate u_{msm} . Two underlined species have point estimate u greater than u_{msm} . Five species in bold have estimated upper 95% confidence intervals of fishing mortality rates greater than the minimum unsustainable fishing mortality rate u_{crash} . P_N = proportion of abundance in fished area; q = catch rate; M = instantaneous natural mortality.

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

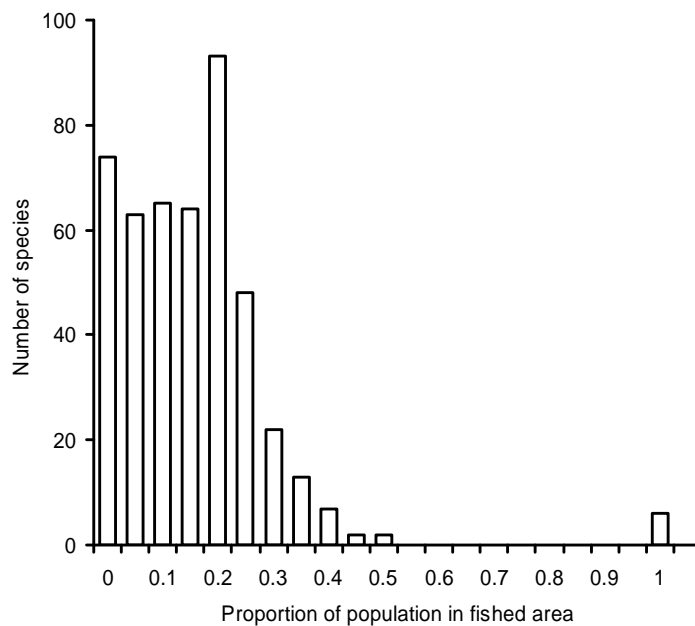


Figure 31. Spatial distribution of 478 teleost bycatch species in NPF fished area.

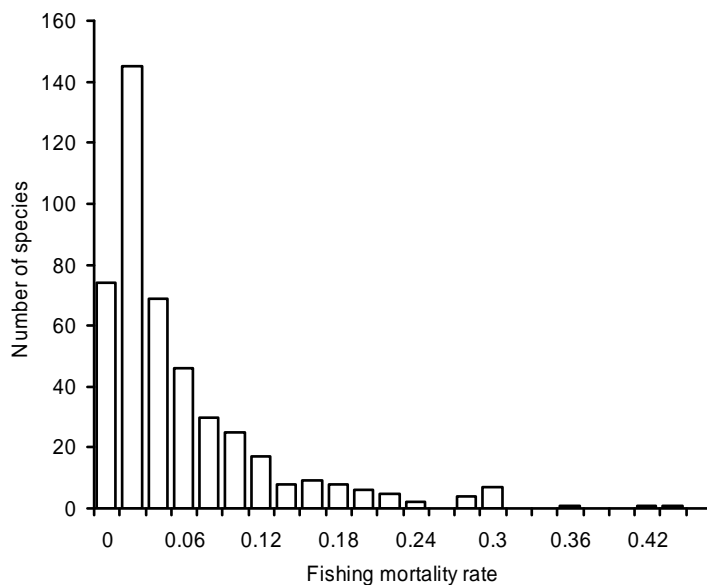


Figure 32. Distribution of the estimated fishing mortality rates for the teleost bycatch species.

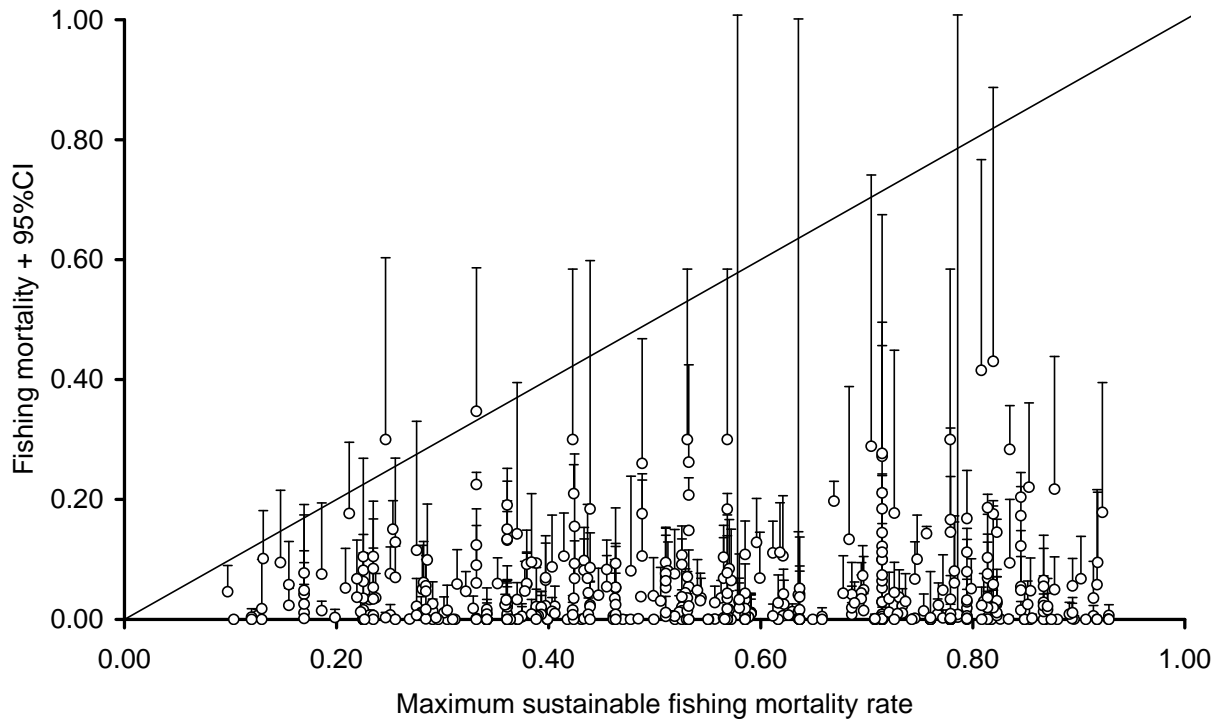


Figure 33. Comparison between estimated fishing mortality rates $u + 95\%$ confidence intervals from prawn trawling and the maximum sustainable fishing mortality rates u_{msm} for the 478 bycatch teleost species. The diagonal line is $u = u_{msm}$.

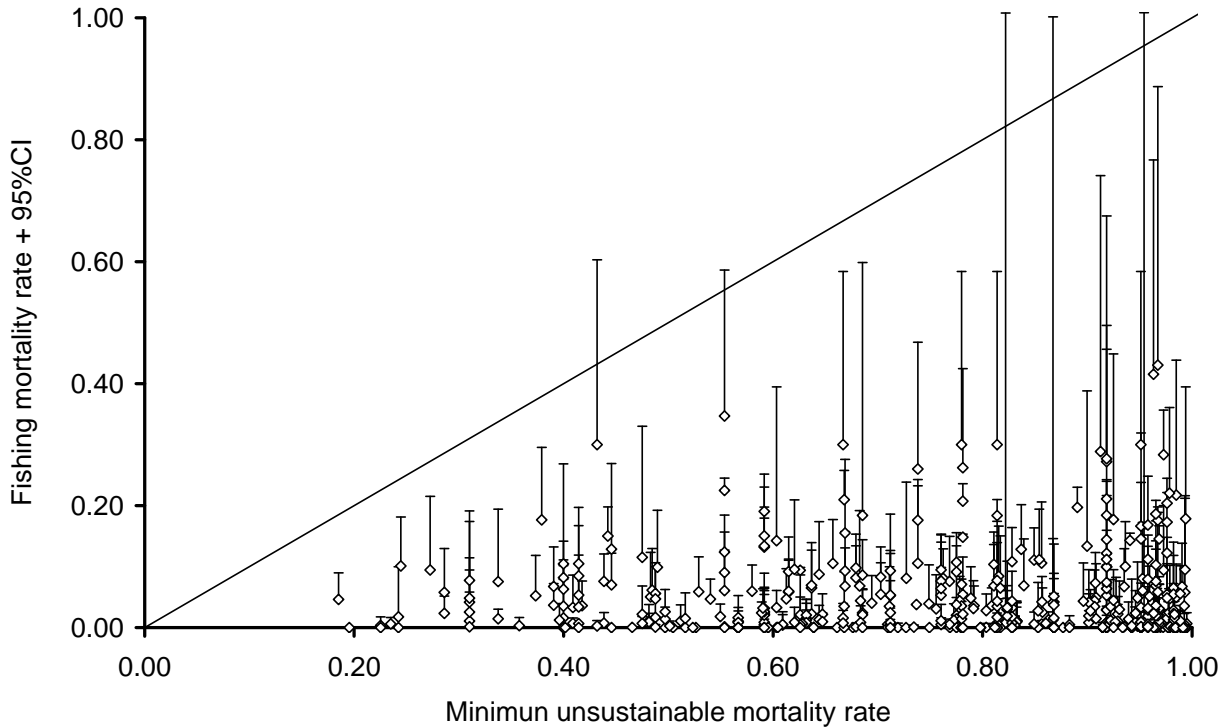


Figure 34. Comparison between estimated fishing mortality rates + 95% confidence intervals from prawn trawling and the minimum unsustainable mortality rate (u_{crash}). The diagonal line is $u = u_{crash}$.

Unlike currently used qualitative or semi-quantitative attribute-based methods (Stobutzki et al., 2001; Fletcher, 2005; Astles et al., 2006), SAFE focuses on one metric—fishing mortality rate. This allows the use of different methods to estimate the fishing impact depending on data available. For example, Pope et al. (2000) used very little catch data to estimate the current overall fishing mortality rate for two non-target species in the North Sea using a simple swept-area method. This involved determining the overlap of a fishery's fished region with a species' distribution and estimating the proportion of the population that is potentially impacted. This approach is useful for fisheries where the density of non-target species in fished area and unfished area is equal, and if it can be confidently assumed that the gear uniformly sweeps the entire fishery and captures all fish in the path of the net. Pearce and Boyce (2006) summarised methods for modelling distribution and abundance using presence-only data, which may be used to estimate the relative fishing mortality rate in a similar way as we have done in his paper. If more information is available, conventional methods, such as catch curve, length-cohort analysis (Jones, 1981; ICES, 1988), the catch-age method (Quinn and Deriso, 1999), or virtual population analysis can be used.

Although there are few methods that can be used to estimate fishing mortality of data-limited species, few biological reference points exist to assess the sustainability of non-target species. Pope et al. (2000) performed length-cohort analysis on catch-at-length data, and used a simple swept-area method to estimate the current overall fishing mortality rate and sustainability of two non-target species in the North Sea. They then assessed the capacity of these populations to withstand the given fishing mortality, by estimating the fishing mortality that would reduce the spawning-stock biomass per recruit to an arbitrary, but supposedly safe, percentage (5%) of its unfished level. Their method for estimating the capacity of the population to withstand fishing mortality requires both life history parameters and fishery information (natural mortality, age at capture, fishing mortality rate etc.), some of which may be difficult to obtain, especially in tropical fisheries where hundreds of bycatch species are often impacted (Stobutzki et al., 2001). In addition, setting an arbitrary value of 5% of the virgin spawners per recruit as a reference point needs further deliberation.

In this paper we recommend two alternative biological reference points for guiding fishery management of non-target bycatch: maximum sustainable fishing mortality rate (μ_{msm}), and minimum unsustainable fishing mortality rate (μ_{crash}). The SAFE approach uses a single life history parameter, natural mortality, for setting these sustainability reference points. Compared to population parameters such as abundance, population growth rate, and density-dependent parameters, life history parameters (e.g. von Bertalanffy growth parameters) that can be used to estimate natural mortality are generally more widely available (Pauly, 1980; Quinn and Deriso, 1999; www.Fishbase.org). It is well known that sustainability depends on a species' intrinsic ability to tolerate external pressures. This intrinsic ability is fundamentally related to a species' life history traits, which are linked by the life history invariant rule (Charnov, 1993; Jennings, 1998; Froese and Binohlan, 2000; Denney et al., 2002; Reynolds et al., 2005; Goodwin et al., 2006). For target fish species, natural mortality has been widely used as a surrogate for optimal fishing mortality since the 1960s (Alverson and Pereyra, 1969; Gulland, 1970). Because finer relationships between natural mortality rate and optimal fishing mortality rate may vary between taxonomic groups (Francis, 1974; Deriso, 1982; Garcia and Csirke, 1989; Clark, 1991), we applied a scaling parameter that is a linear decreasing function of natural mortality rate. Although we feel this approach is suitable for teleosts, further study is needed to establish a more rigorous relationship between biological reference points and life history parameters of a wider group of marine animals.

Our results indicate that few species are at risk of becoming unsustainable due to fishing in the NPF. This may be surprising considering the large number of species impacted (see review by Griffiths et al., 2004). Two main factors contributed to this outcome. First, the aggregated fishing area is small relatively to entire management area (~5%), which limits the proportion of the population exposed to fishing, and second, most teleost bycatch species have high resilience, having short life spans, small body sizes, fast growth rates, and high natural mortalities (Jennings, 1998; Denney et al., 2002; Frisk et al., 2004; Reynolds et al., 2005). However, in order for the true sustainability of species to be assessed the cumulative impacts of all fisheries in the NPF managed region needs to be considered. Although the vast majority of the NPF teleost bycatch species were assessed as being sustainable, many species such as Lutjanids are targeted in state and Commonwealth-regulated gillnet, fish trawl, and longline fisheries, (Zellar and Snape, 2006) and more recently, targeted in Illegal, Unregulated and Unreported (IUU) fisheries. It is possible to use SAFE to assess the cumulative impacts of these fisheries since the model uses a single fishing mortality rate as the standard measure of fishing impact. Therefore, fishing mortality rates from each source can be simply summed to estimate the total impact. Such a cumulative impact can then be evaluated against reference points to determine whether the species can sustain the combine fishing impacts.

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5.4.6 General discussion and conclusions

S. Griffiths

The management paradigm of many fisheries worldwide has undergone a significant transformation in the past decade, shifting from a single species (i.e. target species) focus to considering fishery impacts on entire ecosystems (Hall and Mainprize, 2004). This management approach, now widely referred to as Ecosystem-Based Fisheries Management (EBFM), has arisen in response to increasingly stringent worldwide environmental and fisheries policies and legislation that demand fisheries take greater responsibility for managing the direct and indirect impacts on the supporting ecosystem.

Unfortunately, the enormous complexity of marine ecosystems and the general paucity of quantitative data available for most species — particularly rare or low-value bycatch species that can comprise a significant component of both catches and the ecosystem — means that demonstrating the sustainability of all species impacted by a fishery is a difficult prospect. Although quantitative ecosystem models are becoming increasingly sophisticated (see Christensen and Pauly, 1992; Fulton et al., 2004), successfully modelling entire marine ecosystems and confidently using these models to manage fisheries may be many years from reality (Longhurst, 2006). Consequently, achieving the objectives of EBFM in its true sense may be unrealistic for most fisheries to achieve in the immediate future, despite the explicit requirements already existing within fisheries legislation in some countries (for examples see Scandol et al. 2005).

The aforementioned problems of EBFM are generally well recognised by researchers and fishery managers, but the general concept of maintaining ecosystem integrity and function is unanimously supported as an optimal management goal (Scandol et al., 2005). As a result, several fisheries scientists have developed a range of qualitative and semi-quantitative ecological risk assessment methods that can at least assess the risk of populations of individual species becoming unsustainable due to fishing (see section 5.4.1). Currently used methods cannot assess the indirect impacts of removing a species on gregarious and ‘keystone’ species, or changes in overall ecosystem functionality (but see Fletcher, 2005). However, they can be useful for identifying potential species of concern and prioritising management in the short-term (Fletcher, 2005; Astles et al., 2006).

The main drawback of ecological risk assessment methods currently used in fisheries is that most, if not all, provide only a relative ranking of risk and give little indication whether a species’ population is truly ‘at risk’ of becoming unsustainable due to fishing. Furthermore, by testing the performance of a widely used semi-quantitative attribute-based approach (Milton, 2001; Stobutzki et al. 2002) in Section 5.4.2 (and Griffiths et al. 2006), we demonstrated that these methods have technical flaws that can hinder their ability to correctly reflect even the most obvious changes in the relative risk of individual species. For example, we showed that 10 elasmobranch species caught as NPF bycatch decreased in relative sustainability after introduction of Turtle Excluder Devices (TEDs) into the fishery, which exclude up to 100% of animals from these species (Brewer et al., 2006).

Considering the increasing legislative demands on Australia’s Commonwealth and export fisheries to demonstrate the sustainability of all impacted species, primarily under the *EPBC* Act, we recognised an urgent need in this project for fishery managers to have access to more quantitative methods to assess ecological sustainability. Furthermore, we recognised that

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there are very few practical biological reference points for data-limited, non-target species with which to assess ecological sustainability in order to conform to legislative requirements (but see Diamond, 2004; Hall and Mainprize, 2004). In this section, we addressed these needs in fulfilling Objective 3 of the project: *to develop a new, innovative, quantitative method for defining the risk to the sustainability of bycatch species from prawn trawling, and apply the model to the bycatch of the NPF.*

SAFE - A new quantitative ecological risk assessment approach

In this section we described a new approach for quantitatively assessing the risk of individual bycatch species populations becoming unsustainable due to fishing in the NPF. We refer to this method as a Sustainability Assessment for Fishing Effects (SAFE). Considering the high diversity, limited data availability, and low economic value of NPF bycatch, we took a unique approach that could accommodate these difficult issues, but also provide biological meaningful results. The SAFE approach broadly consists of two separate components: 1) determining the fishing mortality rate of a species based on its spatial overlap with the fishery and their vulnerability to being capture by prawn trawls, and 2) use basic life history parameters to derive biological reference points characterising the capacity of the population to withstand the estimated fishing mortality rate.

We attempted to make the SAFE method conceptually simple, and operate using as few data as possible, in order for the method to be easily transferable to other fisheries. Hence, the first component of SAFE is based on simple detection-nondetection (or presence-absence) data, which are generally cheaper to collect and more widely available than count data. Section 5.4.3 describes an innovative model to estimate the abundance of randomly or aggregated populations from detection-nondetection data collected from repeated surveys, by incorporating detection probabilities and site occupancy rates. The model forms the basis of the first component in our SAFE method, but should have wide reaching ecological application.

Separate risk assessments were undertaken on elasmobranch and teleost bycatch species (sections 5.4.4 and 5.4.5), primarily due to the differences in their general life history traits. Although SAFE can be used in isolation as a fisheries management tool, owing to large number of bycatch species in the NPF and the high uncertainty around some estimates of life history parameters used in the model, we used SAFE as a 'screening' tool to identify potentially 'at risk' species. These species were then nominated as candidates for the long-term monitoring program in order to gather more detailed biological and time-series catch data in order to assess their sustainability in a more rigorous fashion than could be achieved by only using SAFE (see section 5.1).

A total of 51 elasmobranch and 478 teleost species were recorded as trawl bycatch in the NPF managed region in various surveys conducted by CSIRO and state fisheries agencies between 1979 and 2003. Of these species, only six were identified as being at potential risk of becoming unsustainable, since the fishing mortality rates on these species exceeded their biological reference points, u_{crash} . This reference point is the minimum fishing mortality rate that would eventually drive a population to extinction. This reference point is used in stock assessment of target species and usually requires more biological and catch data than is generally available for low-value, little-studied bycatch species. However, by using natural mortality as a surrogate for optimal fishing mortality (Thompson, 1993; Quinn and Deriso 1999) we were able to establish a practical and biologically meaningful reference point for bycatch that can be easily incorporated into existing fishery management strategies.

The potentially ‘at risk’ species comprise four elasmobranch species: *Orectolobus ornatus*, *Squatina sp. A*, *Taeniura meyeni* and *Urogymnus asperrimus*. The model also identified five additional elasmobranch and two teleost species whose upper 95% confidence intervals of the estimated fishing mortality rates were greater than u_{crash} . These potentially ‘at risk’ species were generally those being caught on a few occasions only in the fished area (see discussion in sections 5.4.4 and 5.4.5) where the model interpreted the species distribution as having a 100% overlap with the fishery. These species were investigated in detail to determine whether they were realistically likely to be ‘at risk’, given their distribution and preferred habitat documented in the literature, and determine whether the species may have been misidentified. In a few cases where distribution records were consistent with where the species was recorded in surveys, we sought expert opinion from scientists and NPF fishers in the project steering committee as to their likely risk from trawling in the absence of reliable catch data. For example, SAFE identified the elasmobranch, *Carcharhinus albimarginatus*, to be ‘at risk’ but this species is most commonly found around reefs (Last and Sevens, 1994), where NPF trawling is rare. Experts considered it unlikely that this large, fast-swimming pelagic species would be caught in a demersal trawl, and if so would likely escape through a TED given the results in Brewer et al. (2006).

Although we identified only six species potentially ‘at risk’ of becoming unsustainable due to the NPF’s fishing activities, in order for the true sustainability of species to be assessed the cumulative impacts of all fisheries in the NPF managed region needs to be the focus of future work. This will be particularly important for elasmobranchs since they are most vulnerable to decline from fishing activities, mainly due to their generally slow growth, low natural mortality rate and low reproductive potential (Stevens 1997; Walker 1998). Although the vast majority of the NPF elasmobranch bycatch species were assessed as being sustainable – including narrow and green sawfish – many are target species in state and Commonwealth-regulated gillnet and longline fisheries (e.g. Qld N3 and N9 fisheries) (Zellar and Snape, 2006) and more recently, targeted in Illegal, Unregulated and Unreported (IUU) fisheries for their fins. Because our SAFE approach uses measurable fishing mortality rate to gauge the fishery impact, this approach is likely to successfully quantify cumulative impacts from different sources.

Conclusions

- The management focus of many fisheries worldwide has shifted from single species to EBFM. Ecological risk assessment has been used as a cost-effective alternative to ecosystem models to assist fisheries in demonstrating ecological sustainability.
- Currently used ecological risk assessment methods provide only a relative measure of risk of a species’ population becoming unsustainable due to fishing. Performance testing of an existing method revealed a failure to reflect changes in the relative risk of elasmobranchs after significant exclusion by TEDs in the NPF.
- A Sustainability Assessment for Fishing Effects (SAFE) was developed as a new quantitative model for highly diverse, data-poor bycatch assemblages. The model comprises two components: 1) estimating the fishing mortality rate from the relative abundance of animals caught in fished, relative to unfished areas, and 2) assessing sustainability using a simple biological reference point (u_{crash}) based on basic life history parameters.

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- The first component of SAFE can incorporate a new model designed for data-limited fisheries to cost-effectively estimate abundance of randomly or aggregated fish populations using detection-nondetection data from repeated surveys.
- From the risk assessment of 51 elasmobranch bycatch species, four were considered potentially ‘at risk’ of becoming unsustainable under current fishing levels. These species – *Orectolobus ornatus*, *Squatina sp. A*, *Taeniura meyeni* and *Urogymnus asperrimus* – are recommended for inclusion in the NPF monitoring program.
- Of the 478 teleost bycatch species assessed, only two: *Dendrochirus brachypterus* and *Scorpaenopsis venosa*, were considered to be at potential risk and recommended to be included in the NPF monitoring program.
- The SAFE approach successfully quantified fishing impacts on the sustainability of 539 NPF bycatch species. However, most species identified as high risk attained this status due to being rarely caught, and only from fished areas. This gives a false impression that the fishery impacts the species’ entire population.
- Detailed investigation of geographic range and consultation with scientific and industry experts regarding vulnerability to capture in trawls was a successful ad hoc method of reassessing rarely caught species initially identified as being ‘at risk’.
- Despite our conclusions that most bycatch species are sustainable given the current impacts of the NPF, the cumulative impact from other state and IUU fisheries may be significant for some species, especially elasmobranchs. Further research is required to extend the SAFE approach to assess cumulative fishery impacts in northern Australia.

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5.5 Objective 4 – To provide the first description of the bycatch from the Joseph Bonaparte Gulf

5.5.1 Introduction

S. Griffiths

The Joseph Bonaparte Gulf (JBG) comprises a small (30,000 km²) but important component of the 700,000 km² Northern Prawn Fishery managed area situated on the border of Western Australia and the Northern Territory in the Timor Sea. It is one of the few areas in the NPF where the red-legged banana prawn (*Penaeus indicus*) is the primary target species. Despite the JBG fishery comprising less than 5% of the area of the NPF, it contributes approximately 65% of the NPF's red-legged banana prawn catch (~360 t annually between 2000-2004) and about 20% of the NPF's total banana prawn catch (combined *Penaeus merguensis* and *P. indicus*) (Loneragan et al., 2002).

The JBG is a unique component of the NPF in that fishing takes place both day and night, in both NPF fishing seasons and in relatively deep water between 35 and 70 m. The region also experiences an extremely large tidal range (6-8 m), which restricts fishing to those periods around neap tides when the tidal range and currents are minimal (Kenyon et al., 2004).

In addition to the unique biophysical and fishery characteristics of the JBG, the limited information available on the faunal assemblages in the region (see Ramm et al., 1990) suggests that the species richness, composition and overall biomass of bycatch impacted by the fishery is very different to that recorded throughout other regions in the NPF, such as the Gulf of Carpentaria (Stobutzki et al., 2001, 2003). As a result, more reliable quantitative data is required on the basic composition of species impacted by fishing in the region and how this impact may vary in space and time. This will greatly improve our knowledge of the JBG fishery impacts on bycatch and provide critical information for the quantitative ecological risk assessment model developed in this project (Section 5.4). Consequently, the NPF will be better equipped to quantitatively assess the long-term sustainability of all species impacted throughout the entire NPF managed region. This will allow for more informed management of the fishery by identifying and monitoring species potentially at high risk of depletion due to fishing, and assist the fishery in conforming to current legislation, such as the Environment Protection and Biodiversity Conservation Act 1999.

The primary objective of this part of the project component was to document the bycatch community of the JBG fishery by describing the species composition, seasonal and diel variation in the bycatch assemblage, to quantify catch rates of individual bycatch species, and to discuss implications of these results for management and the NPF long-term monitoring program. However, the project also had an opportunity to comprehensively sample the bycatch of the Gulf of Carpentaria (GoC) common banana prawn fishery (mainly *Penaeus merguensis*), which is also undescribed, even in regions of highest fishing effort.

The banana prawn fishery in the GoC differs significantly to the tiger prawn fishery both spatially and temporally, and with respect to the gear used. As a result, anecdotal evidence suggests the composition and biomass of bycatch in the banana prawn fishery is significantly different. However, there is no quantitative information available that describes the catches of the banana prawn fishery other than for the target species (Taylor, 2002). Although not an initial objective of the project, it was decided the opportunity to sample the GoC banana fishery bycatch should be used to better understand the impacts of the banana fishery on non-

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target species. This data will provide critical information for ecological risk assessments that may be undertaken on the banana fishery in future.

This section consists of two papers that provide the first descriptions of i) the bycatch of the *Penaeus indicus* banana prawn fishery in the JBG and ii) of the bycatch the *Penaeus merguensis* banana prawn fishery in the Gulf of Carpentaria, including a comparison of impacts between the banana and tiger prawn fisheries.

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5.5.2 Species composition and temporal variation of prawn trawl bycatch in the Joseph Bonaparte Gulf, northwestern Australia

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Abstract

The Joseph Bonaparte Gulf banana prawn subfishery is an important component of Australia's Northern Prawn Fishery (NPF). However, the species composition of the large volumes of bycatch caught in this region is poorly known. We sampled the prawn trawl bycatch of the Joseph Bonaparte Gulf from 53 commercial trawls over two years. These samples contained 195 taxa from 85 families; 117 teleost taxa (112 species) contributing 90.9% of the total biomass, 68 invertebrate taxa (50 species; 7.7% of biomass), six species of elasmobranchs (1.3% of biomass) and three species of sea snakes (<1% of biomass). The species composition of this bycatch is distinctly different from that of other tropical regions, including the neighbouring Gulf of Carpentaria in the NPF. The estimated 4,934 t of bycatch taken annually in the JBG consists mainly teleosts (4486 t), invertebrates (382 t) and small elasmobranchs (66 t), with around 4,000 t (81.6% of the total biomass) coming from just six teleost families: Synodontidae (17.7%), Rhinoprenidae (15.9%), Trichiuridae (14.1%), Sciaenidae (12.3%), Engraulidae (10.9%) and Polynemidae (10.7%). Of the other taxa, around 58% occurred in less than 10% of trawls and 28% occurred in only one trawl. Eight species have never been recorded from other bycatch studies in northern Australia. The total bycatch take and its teleost component varied seasonally, while some abundant species also showed seasonal and diel differences in their catch rates and size composition. The data collected in this study will be the basis for a long-term bycatch monitoring program in the region. This will improve the accuracy of quantitative risk assessments used to demonstrate the sustainability of bycatch populations impacted by fishing in northern Australia.

Keywords: Northern Prawn Fishery; Red-legged banana prawn; *Penaeus indicus*; Seasonal; Diel.

Introduction

Being able to demonstrate ecological sustainability is increasingly important to managers of many of the world's fisheries (Hall and Mainprize, 2005; Dichmont et al., 2007). A growing body of literature confirms many types of fishing have a significant effect on non-target populations, habitats and communities (Sainsbury, 1987; Hall, 1996; Pauly et al., 2001; Kaiser et al., 2002). In many countries, commercial fishing is now driven by fisheries and environmental legislation, market drivers and public perception to demonstrate that they are operating in an ecologically sustainable manner (Sainsbury and Sumaila, 2001; Hall and Mainprize, 2005). For example, in Australia, the Environment Protection and Biodiversity Conservation Act 1999 requires all export fisheries to demonstrate that their fishing practices are ecologically sustainable. Furthermore, in a directive from the Australian government in 2005, Commonwealth fisheries are required to halve their current bycatch levels by 2008. However, to satisfy these requirements it is necessary to obtain knowledge of the species and habitats impacted; a considerable challenge in fisheries that interact with highly diverse communities, such as tropical trawl fisheries.

Demersal prawn (shrimp) trawling is a relatively non-selective fishing method, and the bycatch is often a significantly higher proportion of the catch biomass than the target species (Saila, 1983; Andrew and Pepperell, 1992). Bycatch from prawn trawls was estimated in 1993

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to be around 11.2 million t worldwide (Alverson et al., 1994). Australia's Northern Prawn Fishery (NPF) is one of Australia's largest, most valuable and remote fisheries that targets penaeid prawns. Input controls are the primary management strategy used in the NPF, with limits on the number of trawlers (83 in 2006) and spatial and temporal restrictions on fishing. There are also restrictions on fishing gear, in particular headrope length which is regulated by possession of statutory fishing rights (SFR) units. In 2006, a total of 132 class B SFRs were allocated among the 83 boats in the fishery (Griffiths et al., 2006). The discarded bycatch from the NPF was estimated in 1991 to be about 30,000 t (Pender et al., 1992); five times the retained prawn catch. The NPF bycatch is very diverse: comprising over 390 species of teleosts, 43 elasmobranchs, 9 sea snakes, 8 turtles, and over 200 invertebrate taxa (Stobutzki et al., 2001b; Brewer et al., 2006). Some of these taxa are protected, listed or endangered (IUCN, 2006), and many are rare. However, these studies focused on the bycatch in the Gulf of Carpentaria and did not include the Joseph Bonaparte Gulf (JBG) which primarily targets the red-legged banana prawn (*Penaeus indicus*).

The JBG comprises about 30,000 km² of the westernmost portion of the NPF (Figure 35). Fishing for the red-legged banana prawn (*P. indicus*) is permitted day and night in both NPF fishing seasons: autumn (April–mid-June) and spring (late August–November). Fishing takes place in waters 35–70 m deep, with most fishing effort between 50 and 60 m. The trawling regime for this species is similar to the tiger prawn subfishery in other regions of the NPF, where the total duration of individual trawls are usually long (~ 3 h). Although the JBG fishery comprises less than 5% of the area of the NPF, it contributes about 65% of the NPF's red-legged banana prawn catch (~370 t annually between 2000 and 2004) and around 20% of the NPF's total banana prawn catch (combined *Penaeus merguensis* and *P. indicus*) (Loneragan et al., 2002). Due to the large tidal range (6–8 m) and its reputed influence on prawn abundance in the region (pers comm. M. Farrell; P. Williamson; C. Terjensen—NPF skippers), fishing generally occurs during the week of neap tides when the tidal range and currents are minimal.

The NPF is currently developing new methods for assessing the sustainability of diverse bycatch populations (Stobutzki et al., 2001a & 2002; Zhou and Griffiths, in press). Furthermore, the NPF is instituting a long-term bycatch monitoring program that seeks to ensure the sustainability of all non-target species impacted by trawling (NORMAC, 2003). However, these programs require a detailed knowledge of the species being impacted, their geographic distribution and catch rates. Although the JBG fishery contributes a significant proportion to the prawn catch of the NPF, its bycatch is undescribed.

In order to begin to assess the impacts of trawling on the bycatch assemblages of this fishery, the current study was undertaken to: i) describe the bycatch composition; ii) describe the seasonal and diel variation in bycatch composition; iii) quantify catch rates of bycatch species in the JBG; iv) provide an annual estimate of total bycatch and a bycatch to prawn ratio for the period 2000–2004; and, v) based on these results, suggest possible implications for management.

Materials and methods

Collection of data

The bycatch of 53 trawls was sampled opportunistically by trained crew and scientific observers from 14 commercial fishing vessels in the JBG between 10 May 2003 and 14 September 2005; 19 samples from the autumn season (April to mid-June) and 34 samples from the spring season (late August to November) (Figure 35). The fishing effort for red-legged banana prawns is concentrated in the northwest region of the JBG (Loneragan et al., 2002). Subsequently, the samples collected were considered to adequately represent the spatial and depth extent of the fishery (Figure 35). Of these samples, 29 were taken during the day, and 24 during the night. All vessels used Florida Flyer twin rig net configuration (mean headrope length 21.2 ± 0.46 m) with 50mm diamond mesh and a codend of 150 meshes long by 150 meshes round. Trawl duration ranged from 0.5 to 4 hours (mean 2.9 ± 0.1 h) at speeds of about 3.2 knots and were conducted in depths between 45 and 68 m.

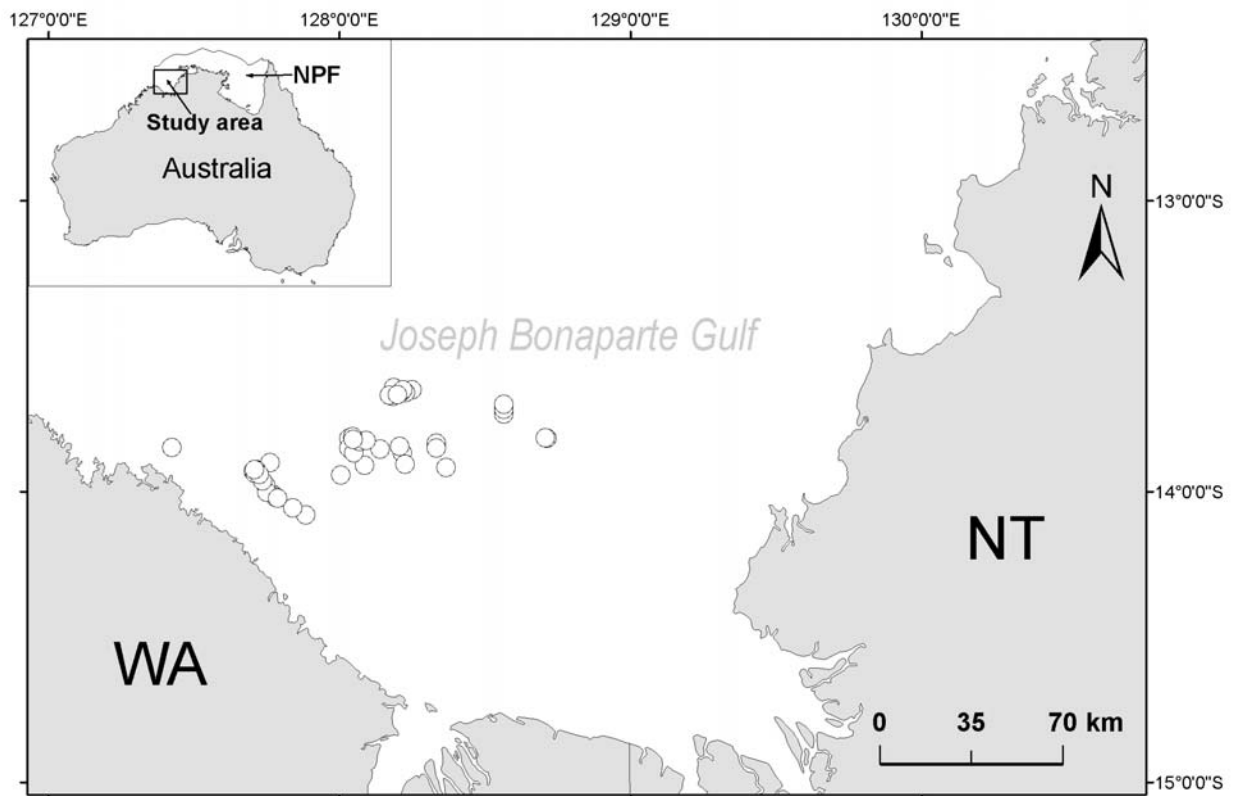


Figure 35. Map of the Joseph Bonaparte Gulf study region on the border of Western Australia (WA) and the Northern Territory (NT) within the Northern Prawn Fishery managed area (NPF). Circles represent sampling locations.

All trawl nets sampled had rigid Turtle Excluder Devices (TEDs), as required by legislation for this fishery (Brewer et al., 2006). TED bar spacing ranged from 95 to 120 mm. TEDs successfully exclude most turtles, large sharks (> 1 m length) and rays (>1 m disc width), and most large sponges (>30 cm width) in this fishery (Brewer et al., 2006). Consequently, only the smaller bycatch species—including most fishes and invertebrates, sea snakes and smaller

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elasmobranch species—were sampled adequately by nets fitted with TEDs and are reported in this study.

The catches of small bycatch species were typically large, each net weighing on average around 500 kg per trawl. Consequently, trained crew or scientific observers subsampled one randomly selected net, collecting at least 10% of the small bycatch after spilling the catch onto the sorting tray or a covered seawater hopper. This method of subsampling has been shown to adequately represent the bycatch composition (Heales et al., 2000). To scale up the subsample, experienced skippers estimated the total catch (bycatch + prawns) from the sampled net. Subsamples were labeled, frozen onboard and sent to CSIRO laboratories for processing.

Several larger elasmobranch species, which are caught in JBG trawl catches, were included to complete the bycatch description (Table 18; Elasmobranchs b). These species were recorded from the scientific observer operations, not from small bycatch subsamples, and therefore were not included in the catch rate analysis.

In the laboratory, animals from each subsample were identified to the lowest taxonomic level (usually species) and counted. The total weight of each taxa was recorded (0.1 g), and 20 randomly chosen individuals, as suggested by Stobutzki et al. (2003), from each species were measured for their standard length (SL in mm) or total length for some species (TL in mm). Taxa other than teleosts were measured for their total length (sharks), disc width (rays), snout-vent length (sea snakes), carapace length or width (crustaceans) and mantle length (cephalopods).

Data analyses

In order to assess how well the subsamples represented the taxa of the JBG fishing grounds, we plotted the cumulative percentage of species detected, against the cumulative percentage of bycatch weight sorted (Figure 36). To estimate the annual bycatch taken from the JBG, the hourly catch rate data derived from this study was combined with total fishing effort. The effort data (number of days fished) was compiled from the Australian Fisheries Management Authority (AFMA) commercial logbook data from fishing effort at greater than 35 m depth and west of 129 degrees longitude and for the years 2000–2004. To calculate the total bycatch for each year, we assumed that mean daily (24 h) total bycatch rates were constant throughout both fishing seasons—autumn and spring. We raised these estimates from our study to the total annual fleet effort in the JBG (Table 17).

The number of individuals and the biomass of each species from each trawl were calculated by multiplying the subsample by a raising factor based on the subsample to total bycatch weight ratio (see Stobutzki et al., 2001b). The catch rates of numbers ($n\ h^{-1}$) and biomass ($kg\ h^{-1}$) of each species were standardised to account for differences in gear specifications and trawl time between vessels. We considered each trawl an independent sample of the demersal population as trawls were non-overlapping (i.e. there was assumed to be no local depletion effects). As is often the case in trawl sampling, there are several potential biases that are difficult, or impossible, to eliminate in order to obtain an accurate estimate of number or biomass of each species caught. These can include size and species-specific selectivity of the trawl net and the speed at which it is towed (Blaber et al., 1990), and differential exclusion by TEDs and Bycatch Reduction Devices (Brewer et al., 2006). We consider our samples as being representative of the fleet and our standardization procedures using measurable factors (e.g. trawl duration, trawl speed, headrope length) allow quantitative comparisons of our samples in relative terms.

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There are several ways to analyse multivariate data depending on the specific hypotheses being posed. In this study we were interested whether the mean biomass and number of species in the overall catch, and the most abundant teleost and invertebrate species caught, differed between day and night and seasons. Secondly, we were interested in determining whether the overall assemblage structure (species composition and the relative abundance and biomass of each species) differed between day and night and seasons. To answer the first question we used a univariate approach by using a two-factor analysis of variance (ANOVA) to compare the mean number of individuals and biomasses of the ten most abundant bycatch species for both teleosts and invertebrates between seasons and time of day. Cochran's and Shapiro–Wilk's tests were used to analyse homogeneity of variances and normality of the data, respectively (Zar, 1984). Data were transformed by $\log_{10}(x+1)$ where necessary, which eliminated heteroscedastic variances. Student–Newman–Keuls (SNK) tests were used for a posteriori comparison among means (Zar, 1984).

To answer the second question, the multivariate approach of non-metric multidimensional scaling (nMDS) was used to examine similarities in fish assemblage structure among seasons and time of day (day or night). Data were fourth-root transformed to reduce the influence of highly abundant taxa, and a similarity matrix constructed using the Bray–Curtis similarity coefficient. The degree of similarity between samples was graphically represented in the form of an ordination plot, which is a two-dimensional representation of the multidimensional relationships between all samples. If an ordination is used in isolation to interpret differences between a priori groups of samples (i.e. seasons) potential problems can arise when the two-dimensional representation of the multidimensional relationships is poor, which can be gleaned from an ordination's "stress" value. Ordinations with stress values of less than <0.2 are useful for interpreting relationships among samples, although some caution should be exercised when values approach the upper end of this range (Clark, 1993). Stress values can increase with sample size, or increase variability among samples. Therefore, other analyses that do not rely upon visual interpretation of a two-dimensional ordination to allow multivariate differences between samples to be statistically tested should be employed. Because the stress values in some of our ordinations were nearing 0.2, analysis of similarities (ANOSIM) was used to test whether fish assemblages in a priori groups differed statistically (Clarke 1993). This analysis involved generating 9999 random permutations of the data in order to calculate the probability whether observed differences in the structure of the bycatch assemblages could arise by chance. Similarity percentages (SIMPER) were then used to determine which species were responsible for differences between groups defined by ANOSIM as being statistically different. All multivariate analyses were carried out with the PRIMER package (Plymouth Routines In Multivariate Ecological Research; version 5.2.2).

Kolmogorov–Smirnov tests were used to determine whether the size composition of individual bycatch species significantly differed between seasons and time of day (day or night). Because this is a pair-wise comparison, interactions between season and time of day were not considered.

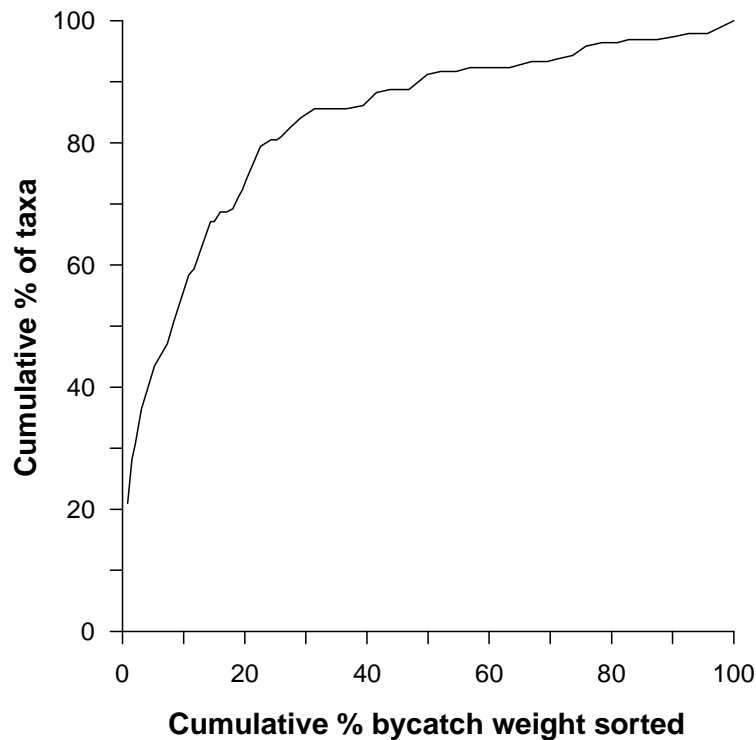


Figure 36. Cumulative percentage of taxa identified plotted against the cumulative bycatch weight processed for bycatch caught in the JBG region.

Results

General catch characteristics

The total of 1,518 kg of bycatch processed from 53 trawl subsamples consisted of 73,289 individuals from 195 taxa (85 families), of which 171 were identified to species (Table 18). A species-cumulative catch curve shows that the number of samples collected is representative of the majority of taxa from these fishing grounds (Figure 36). About 58% of taxa occurred in less than 10% of trawls, while 28% occurred in only one trawl. The contribution of teleosts, invertebrates, elasmobranchs and sea snakes to the total bycatch biomass was 90.9%, 7.7%, 1.3% and <1% respectively. Numbers of individuals in the bycatch were also dominated by teleosts (65.9%), invertebrates (34%), with elasmobranchs (<0.1%) and sea snakes (<0.1%) contributing fewer individuals.

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Table 17. Annual estimates of bycatch (tonnes) and mean catch rates taken in the Joseph Bonaparte Gulf region, 2000-2004. Annual catches of banana prawns were derived from AFMA logbook data.

Year	Hours fished	Total bycatch (t)	Teleosts (t)	Invertebrates (t)	Elasmobranchs (t)
2000	11236	4769	4336	369	64
2001	6877	2919	2654	226	39
2002	12337	5236	4761	405	70
2003	14857	6305	5733	487	85
2004	12821	5441	4948	421	73
Mean	11626	4934	4486	382	66
Mean Catch rate		424.4 kg h ⁻¹	385.9 kg h ⁻¹	32.8 kg h ⁻¹	5.7 kg h ⁻¹

The eight most speciose families accounted for 36% of species. These were Portunidae, Carangidae, Penaeidae, Sciaenidae, Apogonidae, Clupeidae, Engraulidae and Squillidae (Table 19). Synodontidae contributed nearly 18% to the total bycatch biomass, mainly due to the large contribution by *Harpadon translucens* (16.5%). Rhinoprenidae and Trichiuridae, both represented by a single species, also contributed significantly to the biomass: 15.9% and 14.1% respectively. Portunidae were the most numerically dominant family, contributing 20.5% of the bycatch.

A total of 112 teleost species from 61 families was recorded, with nine species occurring in more than 80% of trawls (Table 18). The teleost species with the highest mean catch rates were *H. translucens*, *R. pentanemus*, *T. lepturus*, *P. nigripinnis* and *J. laevis* (Table 18).

A total of 50 invertebrate species from 20 families was recorded from the bycatch subsamples (Table 18). Portunidae and Penaeidae were the most speciose, represented by 13 and 8 species, respectively. Portunidae contributed the most to the invertebrate bycatch: 5.7% of the total biomass, and 73% by weight and 60% by number of the invertebrate component. The swimming crab *Charybdis callianassa* was the most numerically abundant of all bycatch species (Table 18), and accounted for 56% of the total invertebrate biomass. Only four invertebrate species occurred in more than 80% of subsamples (Table 18).

Six elasmobranch species were identified in the subsamples (Table 18; Elasmobranchs a). *Dasyatis annotata* was the most abundant of these (Table 18), occurring in 40% of subsamples and contributing 64% of the total number of elasmobranchs caught.

A total of 11 sea snakes from three species (*Hydrophis elegans*, *Disteira major* and *Lapemis hardwickii*) were recorded from 27 trawls. The most common species were the first two, occurring in 22% and 11% of trawls, respectively (Table 18).

Seasonal trends

Mean catch rates of the total bycatch per trawl differed significantly between seasons with higher catches in spring for biomass ($F = 9.92$; $P < 0.01$) and number ($F = 4.81$; $P < 0.05$) (Figure 37). However, the mean number of species did not differ significantly between spring and autumn ($F = 2.459$; $P > 0.05$).

Teleosts showed a similar pattern to the total bycatch. The mean catch rates differed significantly between spring and autumn, with higher catches in spring for biomass ($F = 12.826$; $P < 0.01$) (Figure 37) and number (sp: $14,454 \pm 2,833h^{-1}$ cf. au: $4,521 \pm 1,167h^{-1}$; $F = 12.826$; $P < 0.001$). However, the mean number of teleost species did not differ significantly between spring and autumn ($F = 4.026$; $P > 0.05$). The mean catch rates of invertebrate bycatch did not differ significantly between spring and autumn for biomass ($F = 0.145$; $P > 0.05$), number ($F = 0.4268$; $P > 0.05$) or the mean number of invertebrate species ($F = 0.8531$; $P > 0.05$).

Of the ten most abundant teleost species, five had mean catch rates (by biomass or number) that differed significantly between seasons (Table 20). These five species had significantly higher catch rates by both biomass and number during spring (Figure 38).

For the ten most abundant invertebrate species, only three differed significantly in their catches between seasons (Table 21). Two species — *Trachypenaeus curvirostris* and *Metapenaeopsis novaeguineae* — had a significantly higher mean biomass and number per trawl during spring, while *Trachypenaeus gonospinifer* had a significantly higher mean biomass and number per trawl in autumn (Table 21, Figure 38).

Length-frequency distributions of the species dominating the small bycatch were examined for seasonal variation in the size structure of their populations. Kolmogorov–Smirnov tests showed that six species — *Trichiurus lepturus* ($D = 0.4592$, $P < 0.001$), *Setipinna tenuifilis* ($D = 0.3230$, $P < 0.001$), *Johnius laevis* ($D = 0.2990$, $P < 0.001$), *Pellona ditchela* ($D = 0.2795$, $P < 0.001$), *Thryssa setirostris* ($D = 0.2338$, $P < 0.001$) and *Charybdis callianassa* ($D = 0.5857$, $P < 0.001$) — had significantly higher proportions of smaller individuals during autumn (Figure 39). Three species — *Harpadon translucens* ($D = 0.2294$, $P < 0.001$), *Rhinoprenes pentanemus* ($D = 0.3399$, $P < 0.001$) and *Polydactylus nigripinnis* ($D = 0.2793$, $P < 0.001$) — had significantly higher proportions of smaller individuals during spring.

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Table 18. Percentage occurrence and mean catch rates of individual bycatch taxa from the Joseph Bonaparte Gulf. Species in bold were recorded from samples collected opportunistically from various sampling methods and quantitative catch data are not available. Symbol (–) indicates that the number of individuals was not recorded.

Bycatch group	Taxa	Family (unless stated)	% occurrence (n = 53)	Mean Biomass (kg h ⁻¹) ± (se)	Mean number (n h ⁻¹) ± (se)
Teleosts	<i>Trichiurus lepturus</i>	Trichiuridae	98	59.665 (12.308)	878.859 (183.003)
	<i>Polydactylus nigripinnis</i>	Polynemidae	94	44.983 (22.587)	957.055 (515.174)
	<i>Johnius laevis</i>	Sciaenidae	94	43.823 (12.997)	1872.039 (586.200)
	<i>Setipinna tenuifilis</i>	Engraulidae	94	29.763 (5.624)	1569.985 (311.598)
	<i>Rhinoprenes pentanemus</i>	Rhinoprenidae	92	67.402 (19.585)	2442.553 (723.690)
	<i>Johnius borneensis</i>	Sciaenidae	89	4.078 (0.840)	150.578 (31.102)
	<i>Paraplagusia longirostris</i>	Cynoglossidae	89	3.861 (0.990)	87.316 (18.923)
	<i>Saurida argentea</i>	Synodontidae	85	4.716 (1.205)	53.020 (15.095)
	<i>Cociella hutchinsi</i>	Platycephalidae	81	3.299 (0.668)	79.324 (14.311)
	<i>Harpadon translucens</i>	Synodontidae	79	69.935 (28.916)	812.557 (383.286)
	<i>Pellona ditchela</i>	Clupeidae	79	13.161 (7.945)	548.522 (301.193)
	<i>Atrobucca brevis</i>	Sciaenidae	77	1.461 (0.372)	46.121 (10.868)
	<i>Thryssa setirostris</i>	Engraulidae	72	14.391 (7.746)	341.547 (163.816)
	<i>Apogon poecilopterus</i>	Apogonidae	68	0.791 (0.177)	154.375 (34.097)
	<i>Terapon theraps</i>	Terapontidae	64	2.696 (0.961)	103.712 (36.98)
	<i>Apogon fasciatus</i>	Apogonidae	60	0.586 (0.175)	83.718 (25.292)
	<i>Euristhmus nudiceps</i>	Plotosidae	58	0.976 (0.327)	27.660 (9.736)
	<i>Upeneus sulphureus</i>	Mullidae	55	1.608 (0.636)	42.894 (16.302)
	<i>Benthoosema pterotum</i>	Myctophidae	53	0.130 (0.048)	248.728 (75.214)
	<i>Psenopsis humerosa</i>	Centrolophidae	49	1.445 (0.442)	49.013 (17.114)
	<i>Synagrops philippinensis</i>	Percichthyidae	45	0.325 (0.106)	56.163 (16.175)
	<i>Leiognathus ruconius</i>	Leiognathidae	43	0.179 (0.089)	64.781 (30.591)
	<i>Inegocia harrisii</i>	Platycephalidae	42	1.049 (0.506)	27.944 (11.515)
	<i>Lepidotrigla russelli</i>	Triglidae	42	0.247 (0.092)	19.666 (11.709)
	<i>Paramonacanthus filicauda</i>	Monacanthidae	38	0.134 (0.059)	13.828 (7.288)
	<i>Sphyraena obtusata</i>	Sphyraenidae	36	0.893 (0.276)	13.597 (4.193)
	<i>Nemipterus hexodon</i>	Nemipteridae	34	1.064 (0.396)	17.222 (6.537)
	<i>Lagocephalus spadiceus</i>	Tetraodontidae	34	0.177 (0.049)	6.242 (1.575)
	<i>Ilisha lunula</i>	Clupeidae	30	1.549 (0.907)	27.842 (16.483)
	<i>Pomadasys maculatus</i>	Haemulidae	25	0.335 (0.19)	6.741 (3.119)
	<i>Leiognathus blochii</i>	Leiognathidae	25	0.056 (0.026)	17.667 (8.148)
	<i>Arnoglossus waitei</i>	Bothidae	25	0.011 (0.005)	5.481 (2.985)
	<i>Thryssa hamiltonii</i>	Engraulidae	23	1.291 (0.685)	14.762 (8.332)
	<i>Epinephelus sexfasciatus</i>	Serranidae	23	0.304 (0.135)	3.920 (1.636)
	<i>Larimichthys pamoides</i>	Sciaenidae	21	1.240 (0.733)	41.575 (22.535)
	<i>Trixiophichthys weberi</i>	Triacanthidae	21	0.260 (0.140)	11.785 (6.569)
	<i>Arius nella</i>	Ariidae	19	0.628 (0.404)	8.910 (4.794)
	<i>Cynoglossus ogilbyi</i>	Cynoglossidae	19	0.102 (0.037)	4.467 (1.541)
	Bregmacerotidae	Bregmacerotidae	19	0.012 (0.008)	8.918 (4.714)
	<i>Zabidius novemaculeatus</i>	Ephippidae	17	0.200 (0.144)	5.910 (4.102)
	<i>Cylichthys hardenbergi</i>	Diodontidae	15	0.747 (0.430)	8.859 (4.745)
	<i>Carangoides malabaricus</i>	Carangidae	15	0.372 (0.265)	9.368 (5.665)
	<i>Valamugil perusii</i>	Mugilidae	15	0.302 (0.230)	5.198 (3.921)
	<i>Parastromateus niger</i>	Carangidae	15	0.277 (0.157)	3.718 (2.012)
	<i>Sirembo imberbis</i>	Ophidiidae	15	0.022 (0.010)	1.969 (0.945)
	<i>Apogon albimaculosus</i>	Apogonidae	15	0.013 (0.006)	2.033 (1.018)
	<i>Bregmaceros maclellandi</i>	Bregmacerotidae	15	0.009 (0.005)	5.035 (3.378)
	<i>Megalaspis cordyla</i>	Carangidae	13	0.297 (0.146)	10.259 (4.917)
	<i>Saurida undosquamis</i>	Synodontidae	13	0.190 (0.127)	7.398 (4.155)
	<i>Psettodes erumei</i>	Psettodidae	11	0.236 (0.110)	0.917 (0.468)
	<i>Pseudorhombus arsius</i>	Paralichthyidae	11	0.200 (0.098)	1.972 (1.000)
	<i>Caranx bucculentus</i>	Carangidae	11	0.152 (0.081)	1.741 (0.884)
	<i>Elates ransonnetii</i>	Platycephalidae	11	0.041 (0.025)	3.161 (2.041)

Objective 4: Bycatch descriptions of banana prawn fisheries

Bycatch group	Taxa	Family (unless stated)	% occurrence (n = 53)	Mean Biomass (kg h ⁻¹) ± (se)	Mean number (n h ⁻¹) ± (se)
	<i>Muraenesox bagio</i>	Muraenesocidae	9	0.129 (0.072)	0.824 (0.467)
	<i>Stolephorus</i> spp.	Engraulidae	9	0.012 (0.006)	0.982 (0.520)
	<i>Cottapistus cottoides</i>	Tetrarogidae	9	0.008 (0.004)	0.847 (0.416)
	<i>Johnius</i> cf <i>trewavasae</i>	Sciaenidae	8	1.508 (1.499)	69.402 (68.940)
	<i>Rastrelliger kanagurta</i>	Scombridae	8	0.188 (0.103)	1.368 (0.743)
	<i>Herklotsichthys lippa</i>	Clupeidae	8	0.049 (0.031)	1.653 (1.001)
	<i>Apistus carinatus</i>	Apistidae	8	0.037 (0.027)	1.533 (1.107)
	<i>Secutor insidiator</i>	Leiognathidae	8	0.029 (0.018)	3.576 (2.254)
	<i>Caranx kleinii</i>	Carangidae	8	0.026 (0.015)	0.714 (0.362)
	<i>Apogon truncatus</i>	Apogonidae	8	0.012 (0.010)	3.023 (2.471)
	<i>Aploactis aspera</i>	Aploactinidae	8	0.010 (0.008)	1.233 (0.767)
	<i>Lagocephalus lunaris</i>	Tetraodontidae	6	0.694 (0.521)	0.709 (0.524)
	<i>Setipinna paxtoni</i>	Engraulidae	6	0.432 (0.358)	21.569 (16.199)
	<i>Ulua aurochs</i>	Carangidae	6	0.079 (0.050)	1.100 (0.736)
	<i>Leiognathus bindus</i>	Leiognathidae	6	0.033 (0.024)	15.394 (12.727)
	<i>Decapterus russelli</i>	Carangidae	6	0.033 (0.022)	0.794 (0.516)
	<i>Gerres filamentosus</i>	Gerreidae	6	0.022 (0.014)	0.820 (0.519)
	<i>Arius thalassinus</i>	Ariidae	6	0.021 (0.012)	0.430 (0.263)
	<i>Protonibea diacanthus</i>	Sciaenidae	6	0.012 (0.007)	0.224 (0.129)
	<i>Terapon jarbua</i>	Terapontidae	6	0.010 (0.005)	0.224 (0.129)
	<i>Minous versicolor</i>	Synanceiidae	6	0.008 (0.005)	0.517 (0.319)
	<i>Tetrabrachium ocellatum</i>	Tetrabrachiidae	6	0.002 (0.002)	0.394 (0.243)
	<i>Laeops parviceps</i>	Bothidae	6	0.002 (0.001)	0.748 (0.408)
	<i>Pomadasyus kaakan</i>	Haemulidae	4	0.197 (0.169)	0.181 (0.127)
	<i>Paraplagusia bilineata</i>	Cynoglossidae	4	0.124 (0.121)	1.195 (1.087)
	<i>Alepes apercna</i>	Carangidae	4	0.059 (0.056)	0.460 (0.379)
	<i>Rhynchostracion nasus</i>	Ostraciidae	4	0.050 (0.050)	0.369 (0.313)
	<i>Euristhmus lepturus</i>	Plotosidae	4	0.044 (0.031)	0.150 (0.107)
	<i>Carangoides talamparoides</i>	Carangidae	4	0.026 (0.024)	0.587 (0.417)
	Mullidae	Mullidae	4	0.007 (0.006)	0.235 (0.172)
	<i>Saurida</i> spp.	Synodontidae	4	0.003 (0.002)	0.249 (0.183)
	<i>Saurenchelys finitimus</i>	Nettastomatidae	4	0.002 (0.002)	0.470 (0.330)
	<i>Repomucenus belcheri</i>	Callionymidae	2	<0.001 (0.000)	0.279 (0.279)
	<i>Champsodon vorax</i>	Champsodontidae	2	<0.001 (0.000)	0.153 (0.153)
	<i>Muraenichthys</i> spp.	Ophichthidae	2	<0.001 (0.000)	0.224 (0.224)
	<i>Eleutheronema tetradactylum</i>	Polynemidae	2	0.090 (0.090)	0.341 (0.341)
	<i>Scomberoides tol</i>	Carangidae	2	0.080 (0.080)	0.102 (0.102)
	<i>Carangoides humerosus</i>	Carangidae	2	0.041 (0.041)	0.341 (0.341)
	<i>Pentaprion longimanus</i>	Gerreidae	2	0.018 (0.018)	2.928 (2.928)
	<i>Priacanthus tayenus</i>	Priacanthidae	2	0.016 (0.016)	0.153 (0.153)
	<i>Echeneis naucrates</i>	Echeneidae	2	0.014 (0.014)	0.217 (0.217)
	<i>Apogon septemstriatus</i>	Apogonidae	2	0.012 (0.012)	3.829 (3.829)
	Sciaenidae	Sciaenidae	2	0.010 (0.010)	0.341 (0.341)
	<i>Paraplagusia sinerama</i>	Cynoglossidae	2	0.008 (0.008)	0.137 (0.137)
	<i>Batrachomoeus trispinosus</i>	Batrachoididae	2	0.007 (0.007)	0.093 (0.093)
	<i>Austronibea oedogenys</i>	Sciaenidae	2	0.007 (0.007)	0.137 (0.137)
	<i>Lutjanus malabaricus</i>	Lutjanidae	2	0.005 (0.005)	0.088 (0.088)
	<i>Mene maculata</i>	Menidae	2	0.005 (0.005)	0.217 (0.217)
	<i>Lagocephalus sceleratus</i>	Tetraodontidae	2	0.004 (0.004)	0.224 (0.224)
	<i>Sardinella albella</i>	Clupeidae	2	0.004 (0.004)	0.217 (0.217)
	<i>Dussumieria elopsoides</i>	Clupeidae	2	0.004 (0.004)	0.164 (0.164)
	<i>Antennarius hispidus</i>	Antennariidae	2	0.004 (0.004)	0.133 (0.133)
	<i>Pterois russelii</i>	Pteroidae	2	0.003 (0.003)	0.086 (0.086)
	<i>Liocranium praepositum</i>	Tetrarogidae	2	0.003 (0.003)	0.088 (0.088)
	<i>Lumiconger arafura</i>	Congridae	2	0.003 (0.003)	0.246 (0.246)
	<i>Chelmon muelleri</i>	Chaetodontidae	2	0.002 (0.002)	0.093 (0.093)
	<i>Pantolabus radiatus</i>	Carangidae	2	0.002 (0.002)	0.090 (0.090)
	<i>Brachypleura novaezeelandiae</i>	Citharidae	2	0.002 (0.002)	0.153 (0.153)
	<i>Uroconger lepturus</i>	Congridae	2	0.001 (0.001)	0.098 (0.098)
	<i>Lagocephalus inermis</i>	Tetraodontidae	2	0.001 (0.001)	0.070 (0.070)

Objective 4: Bycatch descriptions of banana prawn fisheries

Bycatch group	Taxa	Family (unless stated)	% occurrence (n = 53)	Mean Biomass (kg h ⁻¹) ± (se)	Mean number (n h ⁻¹) ± (se)
	<i>Saurida longimanus</i>	Synodontidae	2	0.001 (0.001)	0.308 (0.308)
	<i>Apogon melanopus</i>	Apogonidae			
	<i>Lophichthys boschmai</i>	Lophichthyidae			
	<i>Siphamia roseigaster</i>	Apogonidae			
Elasmobranchs (a)	<i>Dasyatis annotata</i>	Dasyatidae	40	1.141 (0.266)	6.731 (1.374)
	<i>Rhizoprionodon acutus</i>	Carcharhinidae	9	0.289 (0.126)	1.013 (0.484)
	<i>Eusphyra blochii</i>	Sphyrnidae	8	0.163 (0.095)	0.517 (0.307)
	<i>Gymnura australis</i>	Gymnuridae	6	1.166 (0.955)	0.771 (0.483)
	<i>Himantura toshi</i>	Dasyatidae	6	0.651 (0.615)	0.724 (0.670)
	<i>Hemigaleus australiensis</i>	Hemigaleidae	2	0.240 (0.240)	0.740 (0.740)
Elasmobranchs (b)	<i>Rhyncobatus australiae</i>	Rhinidae			
	<i>Carcharhinus tilstoni</i>	Carcharhinidae			
	<i>Chiloscyllium punctatum</i>	Hemiscyllidae			
	<i>Anoxypristis cuspidata</i>	Pristidae			
	<i>Carcharhinus dussumieri</i>	Carcharhinidae			
	<i>Pristis zijsron</i>	Pristidae			
Invertebrates	<i>Charybdis callianassa</i>	Portunidae	98	22.555 (5.629)	3205.650 (848.980)
	<i>Trachypenaeus gonospinifer</i>	Penaeidae	91	2.136 (0.859)	1264.131 (465.979)
	<i>Metapenaeopsis novaeguineae</i>	Penaeidae	91	1.560 (0.285)	328.146 (72.638)
	<i>Solenocera australiana</i>	Solenoceridae	81	1.341 (0.304)	196.382 (40.374)
	<i>Trachypenaeus curvirostris</i>	Penaeidae	74	0.763 (0.221)	160.276 (52.042)
	<i>Oratosquilla gravieri</i>	Squillidae	66	0.255 (0.055)	38.383 (9.208)
	<i>Sepia elliptica</i>	Sepiidae	62	0.585 (0.120)	26.414 (6.773)
	<i>Trachypenaeus anchoralis</i>	Penaeidae	60	0.403 (0.208)	89.200 (42.114)
	<i>Photololigo</i> spp.	Loliginidae	58	0.512 (0.134)	57.344 (18.254)
	<i>Portunus hastatoides</i>	Portunidae	51	0.368 (0.182)	133.541 (67.653)
	<i>Portunus gracilimanus</i>	Portunidae	43	0.301 (0.151)	59.929 (31.044)
	<i>Charybdis truncata</i>	Portunidae	34	0.352 (0.207)	58.094 (35.72)
	<i>Abralia armata</i>	Enoploteuthidae	34	0.108 (0.036)	28.946 (9.768)
	Hydroida	Order Hydroida	34	0.059 (0.026)	–
	<i>Pontocaris arafurae</i>	Crangonidae	34	0.007 (0.002)	9.461 (3.748)
	<i>Proclites levicarina</i>	Pandalidae	30	0.014 (0.009)	13.573 (8.862)
	<i>Metapenaeopsis palmensis</i>	Penaeidae	28	0.107 (0.049)	24.167 (11.238)
	<i>Dictyosquilla tuberculata</i>	Squillidae	26	0.023 (0.008)	3.315 (1.101)
	Scyphozoa	Class Scyphozoa	23	0.335 (0.176)	4.568 (1.743)
	<i>Portunus sanguinolentus</i>	Portunidae	23	0.322 (0.131)	2.775 (1.006)
	<i>Atypopenaeus formosus</i>	Penaeidae	21	0.026 (0.011)	6.970 (3.164)
	<i>Thenus indicus</i>	Scyllaridae	19	0.145 (0.061)	2.135 (0.968)
	<i>Jonas leuteanus</i>	Corystidae	17	0.012 (0.005)	1.650 (0.547)
	Paguroidea	Superfamily	15	0.003 (0.002)	1.505 (0.786)
	<i>Sphenopus marsupialis</i>	Spenopidae	13	0.078 (0.064)	2.029 (1.487)
	Crinoidea	Class Crinoidea	11	0.062 (0.042)	0.567 (0.264)
	<i>Chaetodiadema granulatum</i>	Diadematidae	9	0.017 (0.009)	0.529 (0.242)
	<i>Charybdis feriata</i>	Portunidae	8	0.127 (0.105)	1.644 (1.154)
	Porifera	Phylum Porifera	8	0.064 (0.047)	–
	<i>Octopus</i> spp.	Octopodidae	8	0.018 (0.010)	0.850 (0.475)
	<i>Biarctus sordidus</i>	Scyllaridae	8	0.009 (0.007)	2.064 (1.802)
	Sargassaceae	Sargassaceae	8	0.007 (0.004)	–
	<i>Metapenaeopsis</i> spp.	Penaeidae	8	0.003 (0.002)	6.355 (3.325)
	<i>Charybdis yaldwyni</i>	Portunidae	8	0.003 (0.002)	0.396 (0.204)
	<i>Euprymna cf tasmanica</i>	Sepiolidae	8	0.003 (0.001)	0.617 (0.331)
	Nephtheidae	Nephtheidae	6	0.027 (0.014)	0.093 (0.093)
	Pennatulidae	Pennatulidae	6	0.011 (0.010)	1.017 (0.823)
	Penaeidae	Penaeidae	6	0.004 (0.003)	7.506 (5.964)
	<i>Panulirus polyphagus</i>	Palinuridae	4	0.096 (0.074)	0.493 (0.387)
	<i>Oratosquilla interrupta</i>	Squillidae	4	0.012 (0.009)	0.597 (0.460)
	<i>Harpisquilla harpax</i>	Squillidae	4	0.012 (0.009)	0.294 (0.222)

Objective 4: Bycatch descriptions of banana prawn fisheries

Bycatch group	Taxa	Family (unless stated)	% occurrence (n = 53)	Mean Biomass (kg h ⁻¹) ± (se)	Mean number (n h ⁻¹) ± (se)
	<i>Chloeia flava</i>	Amphinomidae	4	0.004 (0.003)	0.335 (0.260)
	<i>Charybdis hellerii</i>	Portunidae	4	0.004 (0.002)	0.159 (0.114)
	<i>Phalangipus longipes</i>	Majidae	4	0.003 (0.003)	0.711 (0.619)
	<i>Phalangipus australiensis</i>	Majidae	4	0.003 (0.002)	0.911 (0.756)
	<i>Sepia papuensis</i>	Sepiidae	4	0.002 (0.001)	0.163 (0.118)
	<i>Lupocyclus rotundatus</i>	Portunidae	4	0.001 (0.001)	0.277 (0.203)
	Ophiuroidea	Class Ophiuroidea	2	<0.001 (0.000)	0.093 (0.093)
	<i>Cryptolutea</i> sp.	Pilumnidae	2	<0.001 (0.000)	0.056 (0.056)
	<i>Harrovia tuberculata</i>	Pilumnidae	2	<0.001 (0.000)	0.370 (0.370)
	<i>Lissocarcinus arkati</i>	Portunidae	2	<0.001 (0.000)	0.086 (0.086)
	<i>Atergatopsis tweediei</i>	Xanthidae	2	<0.001 (0.000)	0.090 (0.090)
	Echinoidea	Class Echinoidea	2	0.024 (0.024)	1.850 (1.850)
	<i>Erugosquilla woodmasoni</i>	Squillidae	2	0.015 (0.015)	0.433 (0.433)
	<i>Sepiella weberi</i>	Sepiidae	2	0.013 (0.013)	0.740 (0.740)
	Stichopodidae	Stichopodidae	2	0.011 (0.011)	0.170 (0.170)
	Holothuridae	Holothuridae	2	0.010 (0.010)	0.433 (0.433)
	Asciidiidae	Asciidiidae	2	0.010 (0.010)	0.246 (0.246)
	<i>Thalamita sima</i>	Portunidae	2	0.008 (0.008)	0.459 (0.459)
	<i>Parapenaeopsis arafurica</i>	Penaeidae	2	0.004 (0.004)	0.349 (0.349)
	<i>Carcinoplax purpurea</i>	Goneplacidae	2	0.004 (0.004)	0.919 (0.919)
	<i>Lupocyclus tugelae</i>	Portunidae	2	0.003 (0.003)	0.220 (0.220)
	<i>Charybdis jaubertensis</i>	Portunidae	2	0.002 (0.002)	0.206 (0.206)
	<i>Sepia pharaonis</i>	Sepiidae	2	0.001 (0.001)	0.088 (0.088)
	<i>Axiopsis consobrina</i>	Axiidae	2	0.001 (0.001)	0.900 (0.900)
	<i>Glabropilumnus seminudus</i>	Pilumnidae	2	0.001 (0.001)	0.278 (0.278)
	<i>Parapenaeus</i> spp.	Penaeidae	2	0.001 (0.001)	0.450 (0.450)
	<i>Trachypenaeus granulosus</i>	Penaeidae	2	0.001 (0.001)	0.089 (0.089)
Other	Debris and rocks		2	0.002 (0.002)	–
Reptiles	<i>Hydrophis elegans</i>	Hydrohidae	22		
	<i>Disteira major</i>	Hydrohidae	11		
	<i>Lapemis hardwickii</i>	Hydrohidae	4		

Objective 4: Bycatch descriptions of banana prawn fisheries

Table 19. Families caught as bycatch from trawls in the Joseph Bonaparte Gulf, with the percentage contribution to the overall number and biomass of the catch. The * denotes families where not all specimens could be identified to species.

Family/ other	Species	Number	Biomass (%)
Teleosts			
Carangidae	12	0.17	0.34
Sciaenidae	*8	12.91	12.34
Apogonidae	6	1.46	0.33
Clupeidae	5	3.42	3.49
Engraulidae	5	11.54	10.86
Synodontidae	*5	5.17	17.71
Cynoglossidae	4	0.55	0.97
Leiognathidae	4	0.6	<0.1
Tetraodontidae	4	<0.1	0.21
Platycephalidae	3	0.65	1.04
Ariidae	2	<0.1	0.15
Bothidae	2	<0.1	3.33
Congridae	2	<0.1	<0.1
Gerreidae	2	<0.1	<0.1
Haemulidae	2	<0.1	0.13
Plotosidae	2	0.16	0.24
Polynemidae	2	5.67	10.67
Terapontidae	2	0.62	0.64
Tetrarogidae	2	<0.1	<0.1
Bregmacerotidae	*2	<0.1	<0.1
Mullidae	*2	0.26	0.38
Antennariidae	1	<0.1	<0.1
Apistidae	1	<0.1	<0.1
Aploactinidae	1	<0.1	<0.1
Batrachoididae	1	<0.1	<0.1
Callionymidae	1	<0.1	<0.1
Centrolophidae	1	0.29	0.34
Chaetodontidae	1	<0.1	<0.1
Champsodontidae	1	<0.1	<0.1
Citharidae	1	<0.1	<0.1
Diodontidae	1	<0.1	0.17
Echeneidae	1	<0.1	<0.1
Ephippidae	1	<0.1	<0.1
Lutjanidae	1	<0.1	<0.1
Menidae	1	<0.1	<0.1
Monacanthidae	1	<0.1	<0.1
Mugilidae	1	<0.1	<0.1
Muraenesocidae	1	<0.1	<0.1
Myctophidae	1	1.47	<0.1
Nemipteridae	1	0.1	0.25
Nettastomatidae	1	<0.1	<0.1
Ophidiidae	1	<0.1	<0.1
Ostraciidae	1	<0.1	<0.1
Paralichthyidae	1	<0.1	<0.1
Percichthyidae	1	0.33	<0.1
Priacanthidae	1	<0.1	<0.1
Psettodidae	1	<0.1	<0.1
Pteroidae	1	<0.1	<0.1
Rhinoprenidae	1	14.46	15.95
Scombridae	1	<0.1	<0.1
Serranidae	1	<0.1	<0.1
Sphyraenidae	1	<0.1	0.21

Objective 4: Bycatch descriptions of banana prawn fisheries

Table 5.5.2-3 continued

Family/ other	Species	Number	Biomass (%)
Synanceiidae	1	<0.1	<0.1
Tetrabrachiidae	1	<0.1	<0.1
Triacanthidae	1	<0.1	<0.1
Trichiuridae	1	5.2	14.12
Triglidae	1	0.12	<0.1
Ophichthidae	*1	<0.1	<0.1
Elasmobranchs			
Dasyatidae	2	<0.1	0.42
Carcharhinidae	1	<0.1	<0.1
Gymnuridae	1	<0.1	0.28
Hemigaleidae	1	<0.1	<0.1
Sphyrnidae	1	<0.1	<0.1
Invertebrates			
Portunidae	13	20.5	5.69
Penaeidae	*11	11.17	1.18
Squillidae	5	0.25	<0.1
Sepiidae	4	0.16	0.14
Pilumnidae	3	<0.1	<0.1
Majidae	2	<0.1	<0.1
Scyllaridae	2	<0.1	<0.1
Amphinomidae	1	<0.1	<0.1
Axiidae	1	<0.1	<0.1
Corystidae	1	<0.1	<0.1
Crangonidae	1	<0.1	<0.1
Diadematidae	1	<0.1	<0.1
Enoploteuthidae	1	0.17	<0.1
Goneplacidae	1	<0.1	<0.1
Palinuridae	1	<0.1	<0.1
Pandalidae	1	<0.1	<0.1
Sepiolidae	1	<0.1	<0.1
Solenoceridae	1	1.16	0.32
Spenopidae	1	<0.1	<0.1
Xanthidae	1	<0.1	<0.1
Ascidiidae		<0.1	<0.1
Class Crinoidea		<0.1	<0.1
Class Echinoidea		<0.1	<0.1
Holothuridae		<0.1	<0.1
Loliginidae		0.34	0.12
Nephtheidae		<0.1	<0.1
Octopodidae		<0.1	<0.1
Order Hydroida		<0.1	<0.1
Superfamily		<0.1	<0.1
Pennatulidae		<0.1	<0.1
Sargassaceae			<0.1
Stichopodidae		<0.1	<0.1
Class Scyphozoa			<0.1

MDS ordinations showed clear seasonal differences in the JBG bycatch assemblage for either biomass or numbers of individuals (Figure 40). ANOSIM further supported the MDS results, showing significant differences between seasons for biomass ($R=0.256$, $P=0.0000$) and number of individuals ($R=0.217$, $P=0.001$). SIMPER showed that seven species contributed 72% to the dissimilarity of the biomass of the bycatch assemblages between fishing seasons. These species were *Rhinoprenes pentanemus* (16%), *Trichiurus lepturus* (12.9%), *Harpadon translucens* (11%), *Johnius laevis* (10.6%), *Setipinna tenuifilis* (8.1%), *Charybdis callianassa*

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(7.2%) and *Polydactylus nigripinnis* (5.7%), with all (except for *C. callianassa*) having a higher catch biomass during spring. With respect to the number of individuals, SIMPER showed that six species contributed 65% to the dissimilarity of the bycatch assemblages between fishing seasons: *C. callianassa* (17.1%), *R. pentanemus* (13.9%), *S. tenuifilis* (10.1%), *J. laevis* (9.9%), *Trachypenaeus gonospinifer* (8.9%) and *T. lepturus* (5%). Two invertebrates, *C. callianassa* and *T. gonospinifer*, had higher catch rates during autumn; while four teleosts, *R. pentanaemus*, *S. tenuifilis*, *J. laevis* and *T. lepturus*, had higher catch rates during spring.

Diel trends

The total bycatch per trawl did not differ significantly between day and night (both seasons combined) in either the biomass ($F = 2.429$; $P > 0.05$), or number ($F = 1.088$; $P > 0.05$) (Figure 37). Likewise, the mean number of species did not differ significantly between day and night catches ($F = 0.0028$; $P > 0.05$).

For the teleost component, there was no significant difference between day and night catches (both seasons combined) in either the biomass ($F = 3.1851$; $P > 0.05$) or total numbers of teleosts ($F = 3.905$; $P > 0.05$). Similarly, the mean number of teleost species did not differ significantly between day and night ($F = 0.018$; $P > 0.05$). The mean catch rates of two of the most abundant teleosts—*Pellona ditchela* and *Thryssa setirostris*—were significantly higher during the day than the night (Table 20, Figure 41).

There was no difference in either the total biomass ($F = 0.5607$; $P > 0.05$) or total number ($F = 0.1144$; $P > 0.05$) of invertebrates in catches between day and night (both seasons combined). The mean number of invertebrate species did not differ significantly between day and night ($F = 0.0185$; $P > 0.05$). Of the ten most abundant invertebrates, only *Photololigo* spp. showed a significant diel difference in catch rates with higher biomass and numbers per trawl caught during daylight hours (Table 21, Figure 41).

The size structures of the most abundant bycatch species differed significantly between day and night for four species: *T. lepturus* ($Z = 1.875$, $P < 0.05$), *P. nigripinnis* ($Z = 1.81$, $P < 0.05$), *S. tenuifilis* ($Z = 2.017$, $P < 0.05$) and *H. translucens* ($Z = 5.359$, $P < 0.05$). Length-frequency histograms show that smaller individuals of *H. translucens* and *T. lepturus* were caught during the day than at night (Figure 42). In contrast, the length–frequency histogram for *P. nigripinnis* indicates that smaller individuals of this species were caught at night, while no pattern for *S. tenuifilis* was apparent (Figure 42).

MDS ordinations did not show any clear day/night differences in the JBG bycatch species composition for either biomass or numbers of individuals (Figure 40). ANOSIM supported this finding for biomass ($R = 0.04$, $P = 0.073$). However, ANOSIM showed a significant difference between day and night for numbers of individuals ($R = 0.056$, $P = 0.041$). SIMPER supported this finding, showing that six species contributed 64% to the dissimilarity of the bycatch assemblages: *C. callianassa* (15.8%), *R. pentanemus* (15.2%), *J. laevis* (10.9%), *S. tenuifilis* (9.1%), *T. gonospinifer* (7.6%) and *T. lepturus* (5.4%). Three species – *C. callianassa*, *S. tenuifilis* and *T. lepturus* – had higher catch rates during the day, while the other three species had higher catch rates during the night.

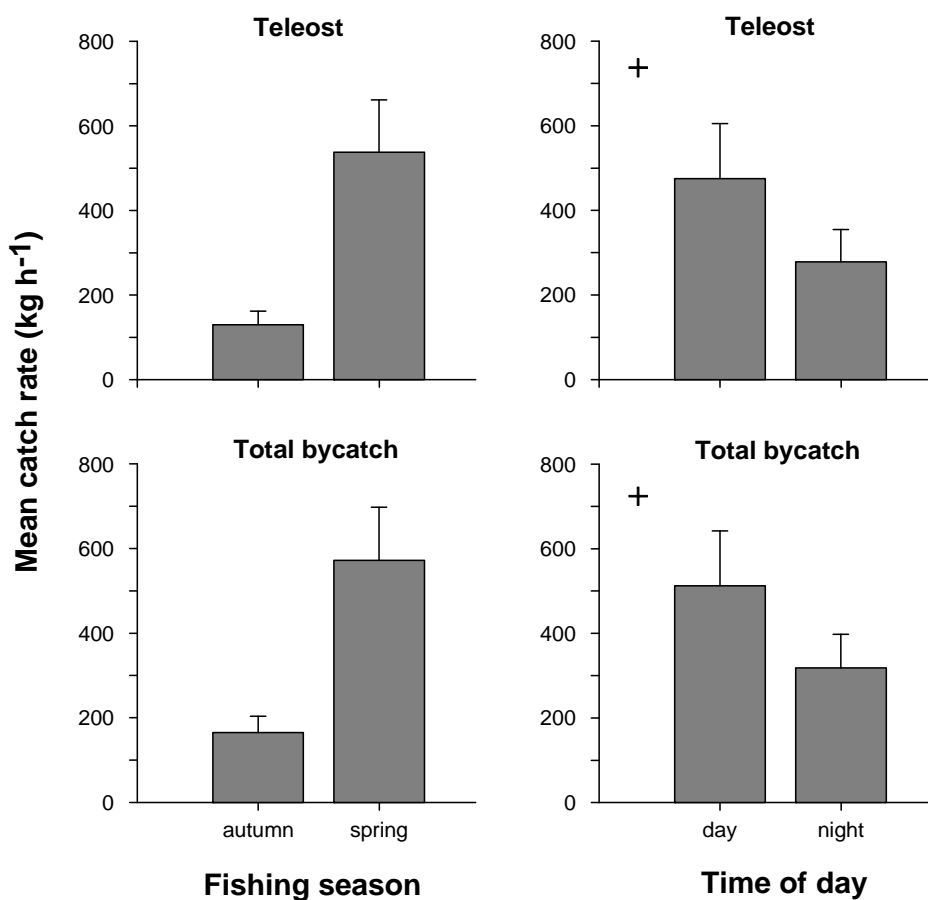


Figure 37. Mean catch rates (\pm SE) for teleosts and total bycatch between prawn fishing seasons and time of day. Means that did not differ significantly in ANOVAs are denoted by +.

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Table 20. Results of two-factor fixed ANOVAs of biomass and number of individuals for the ten most abundant teleost species caught as bycatch in the Joseph Bonaparte Gulf; testing for differences between seasons and day and night (D–N). Mean squares (MS) are shown. Degrees of freedom are shown in parentheses. Significant results are shown in bold and significance levels shown as: * = $P < 0.05$; ** = $P < 0.01$; *** = $P < 0.001$.

Species	Season (1)	Day–night (1)	Season x Day–Night	Residual (49)
Biomass				
<i>Apogon poecilopterus</i>	0.1833	0.1051	0.0100	0.1220
<i>Benthoosema pterotum</i>	0.0002	0.0096	0.0001	0.0046
<i>Thryssa setirostris</i>	0.0002	16.0157***	0.1838	1.0452
<i>Pellona ditchela</i>	0.0163	7.9439*	0.0015	1.1576
<i>Harpadon translucens</i>	46.2056***	1.5116	0.0515	1.9274
<i>Trichiurus lepturus</i>	3.1175	4.4827	0.1249	1.9405
<i>Polydactylus nigripinnis</i>	16.5600**	1.7848	0.0009	1.4552
<i>Setipinna tenuifilis</i>	23.7701***	3.3216	0.0003	0.8536
<i>Johnius laevis</i>	22.1187***	0.2197	0.0034	1.2440
<i>Rhinoprenes pentanemus</i>	18.7819**	2.9154	2.1045	1.7483
Number				
<i>Apogon poecilopterus</i>	10.7462	1.6845	0.3370	4.9108
<i>Benthoosema pterotum</i>	3.5475	12.3837	0.0002	5.7669
<i>Thryssa setirostris</i>	8.0000	72.8446***	0.5059	3.3892
<i>Pellona ditchela</i>	2.6218	31.8477*	0.0709	4.7392
<i>Harpadon translucens</i>	94.0161***	3.8947	0.2957	3.542
<i>Trichiurus lepturus</i>	1.0199	5.3465	0.5446	2.8702
<i>Polydactylus nigripinnis</i>	42.111***	3.0688	1.0245	2.7773
<i>Setipinna tenuifilis</i>	60.1002***	7.0005	0.4240	2.4834
<i>Johnius laevis</i>	50.5946***	1.5386	0.7012	2.7261
<i>Rhinoprenes pentanemus</i>	36.6504**	8.9826	3.5903	3.0213

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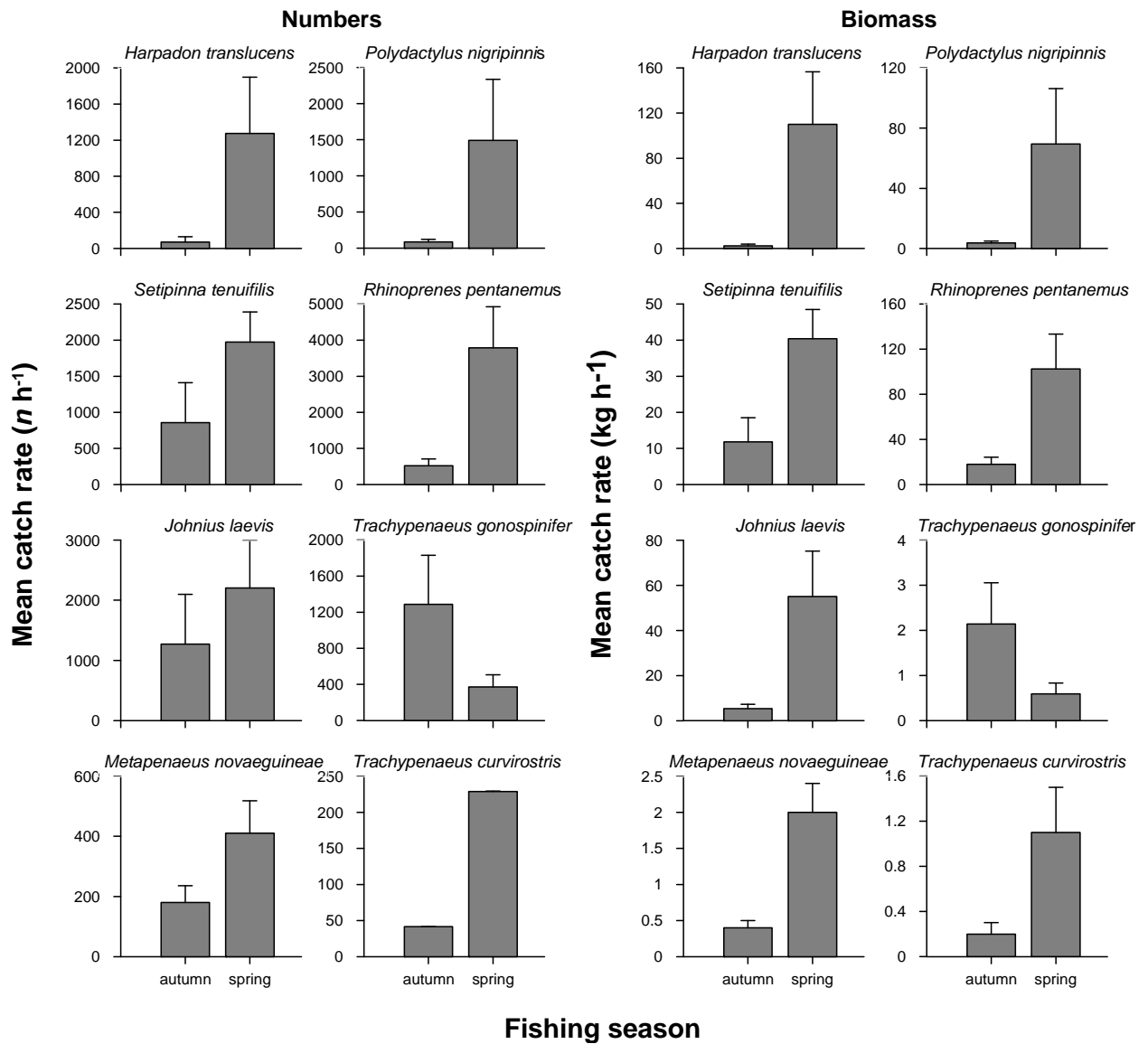


Figure 38. Bycatch species that had significantly different mean catch rates (\pm SE) between fishing seasons (autumn/spring), for both numbers and biomass data.

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Table 21. Results of two-factor fixed ANOVAs of biomass and number of individuals of the ten most abundant invertebrate species caught as bycatch in the Joseph Bonaparte Gulf; testing for differences between seasons and day and night (D-N). Mean squares (MS) are shown. Degrees of freedom are shown in parentheses. Significant results are shown in bold and significance levels shown as: * = $P < 0.05$; ** = $P < 0.01$; *** = $P < 0.001$.

Species	Season (1)	Day-night (1)	Season x Day-Night	Residual (49)
Biomass				
<i>Photololigo</i> spp.	0.0032	0.5893**	0.0195	0.0695
<i>Portunus gracilimanus</i>	0.0643	0.0534	0.0176	0.0732
<i>Trachypenaeus curvirostris</i>	0.7607*	0.0240	0.0588	0.1130
<i>Trachypenaeus gonospinifer</i>	1.1128*	0.5155	0.0002	0.1872
<i>Portunus hastatoides</i>	0.0203	0.0357	0.0659	0.0884
<i>Metapenaeopsis novaeguineae</i>	1.6042**	0.0296	0.0036	0.1494
<i>Solenocera australiana</i>	0.3357	0.0242	0.3192	0.2330
<i>Charybdis callianassa</i>	0.0945	1.1774	0.4924	1.0438
<i>Sepia elliptica</i>	0.0210	0.0086	0.0452	0.0845
<i>Oratosquillina gravieri</i>	0.0001	0.0011	0.0077	0.0251
Number				
<i>Photololigo</i> spp.	6.4756	20.7436*	0.9014	2.8952
<i>Portunus gracilimanus</i>	3.2132	0.0478	2.0456	3.2344
<i>Trachypenaeus curvirostris</i>	19.2665*	12.0045	7.5220	3.4839
<i>Trachypenaeus gonospinifer</i>	17.880*	7.9340	0.0001	3.9402
<i>Portunus hastatoides</i>	4.9740	0.3057	0.7750	4.8205
<i>Metapenaeopsis novaeguineae</i>	19.2422**	8.1744	0.3882	2.5756
<i>Solenocera australiana</i>	0.1534	11.0532	0.2362	4.1103
<i>Charybdis callianassa</i>	3.1989	0.9763	0.8736	1.7050
<i>Sepia elliptica</i>	3.1831	0.5594	2.1963	2.1929
<i>Oratosquillina gravieri</i>	2.1767	2.7841	1.4021	2.6705

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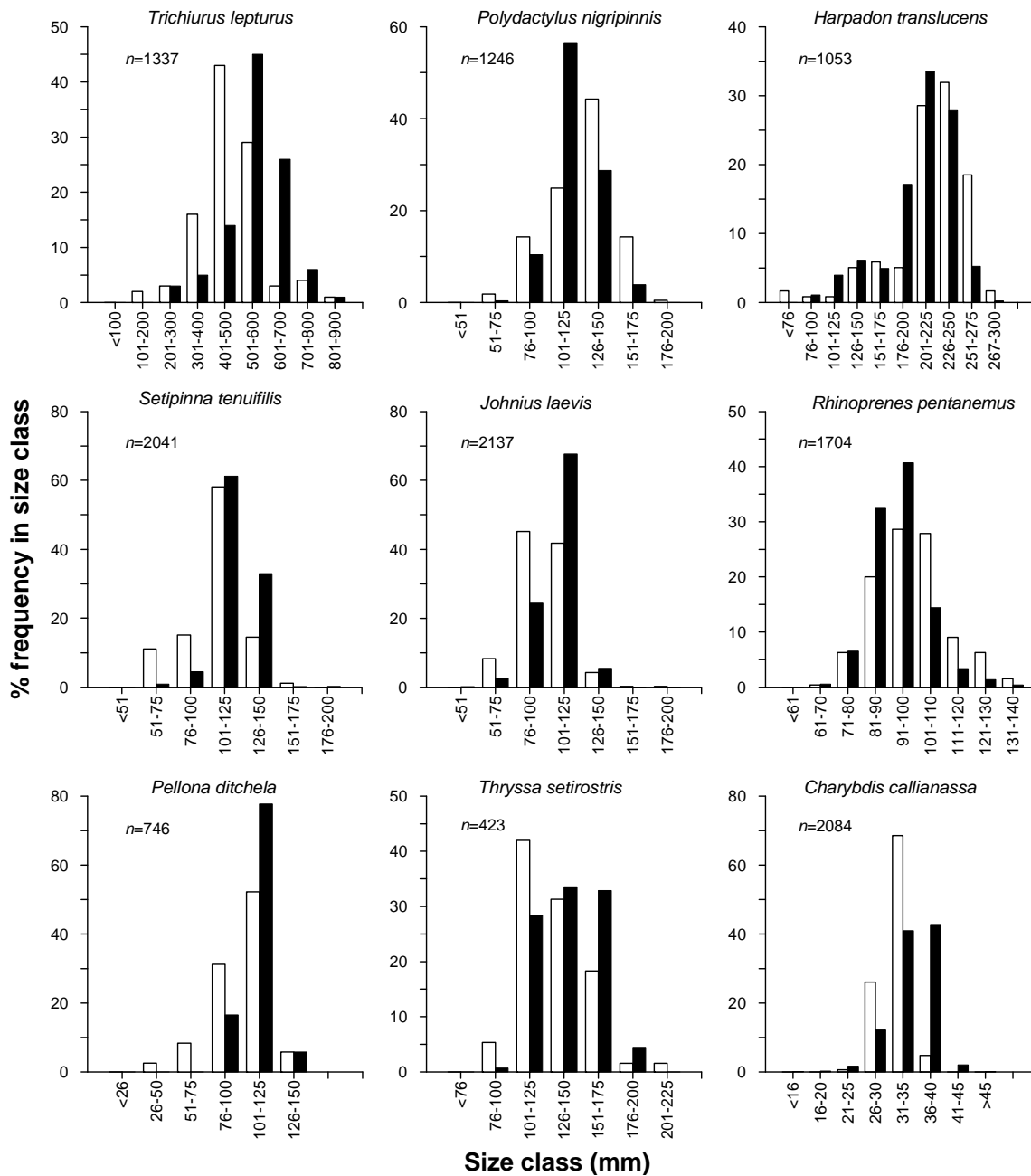


Figure 39. Percentage length–frequency distributions of the nine most abundant species caught in the autumn (white bars) and spring (black bars) fishing seasons. All measurements are standard length (SL), except for *T. lepturus* (total length) and *C. callianassa* (carapace width).

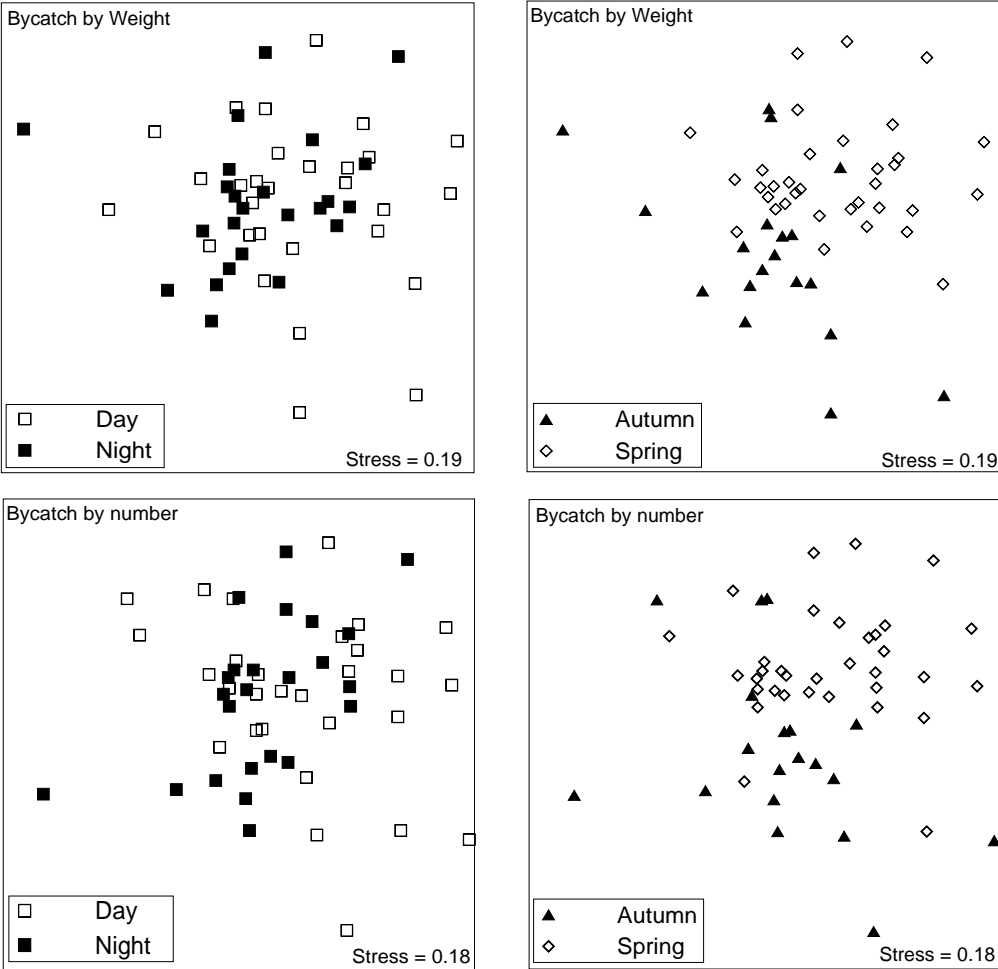


Figure 40. Non-metric MDS ordination plots showing diel and seasonal comparisons of bycatch species composition based on numbers and biomass.

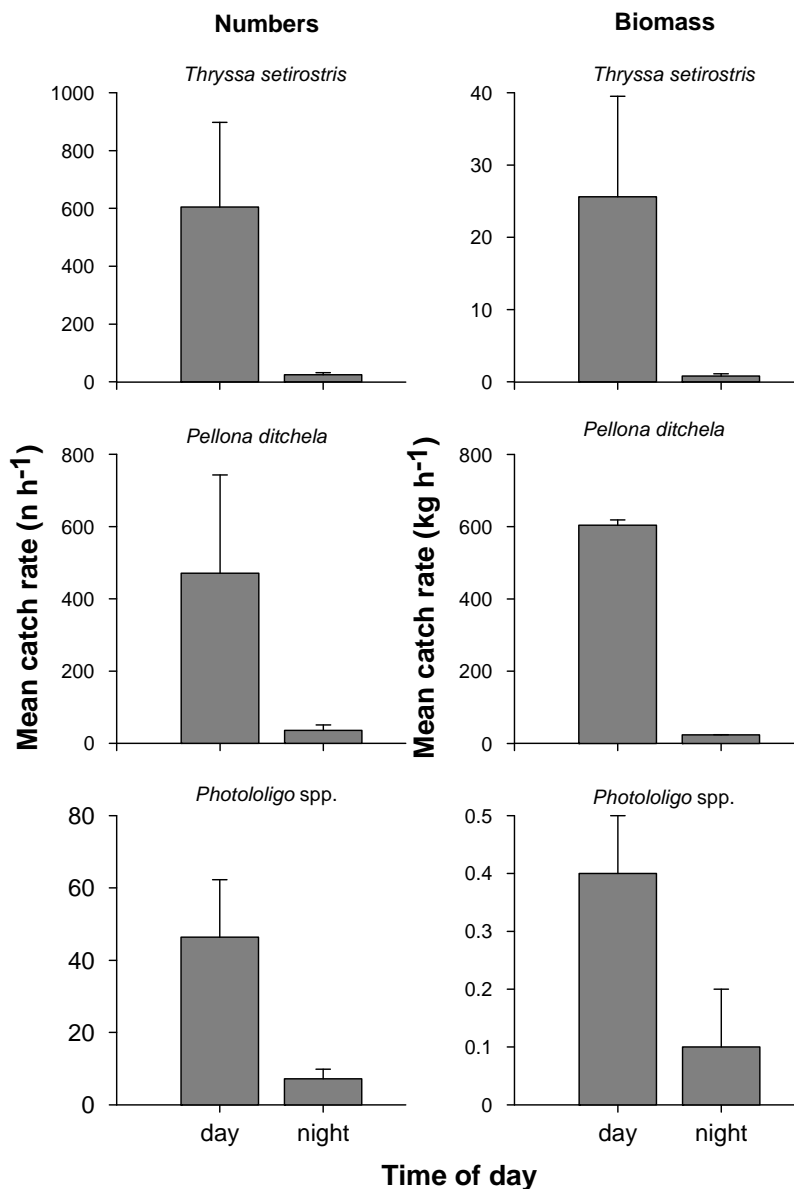


Figure 41. Bycatch species that had significantly different mean catch rates (\pm SE) between day and night, for numbers and biomass data.

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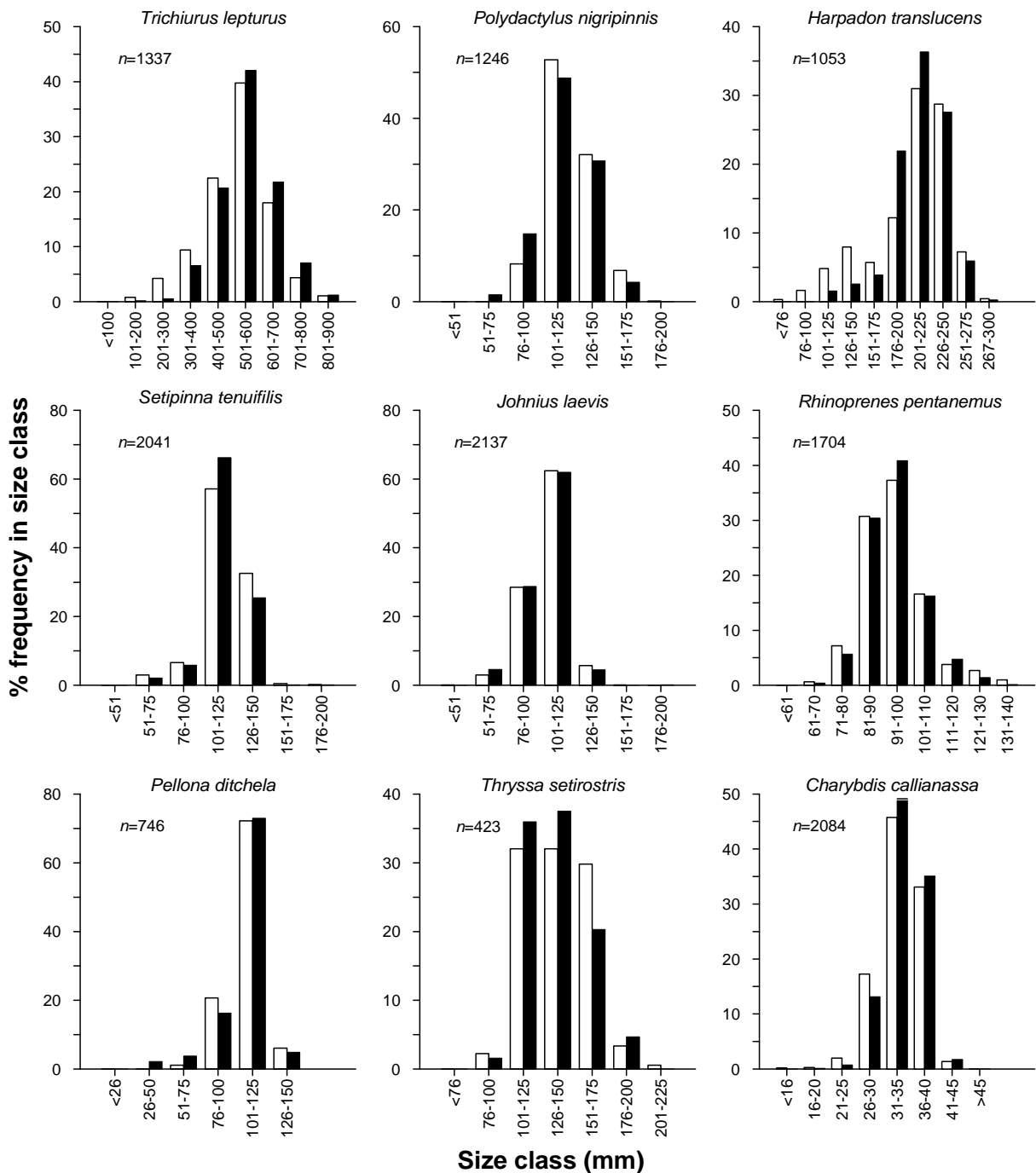


Figure 42. Percentage length–frequency distributions of the nine most abundant species caught during the day (white bars) and night (black bars). All measurements are standard length (SL), except for *T. lepturus* (total length) and *C. callianassa* (carapace width).

Discussion

This was the first comprehensive study of the banana prawn trawl bycatch from the Joseph Bonaparte Gulf. The bycatch of the JBG, like other tropical penaeid fisheries, has high diversity (Pender et al., 1992; Garcia-Caudillo et al., 2000; Stobutzki et al., 2001b). However, the high relative abundance of a few species is a unique characteristic of this region and the bycatch has a distinctly different species composition to other demersal communities in the NPF. For example, Stobutzki et al. (2001b) described two different bycatch assemblages in the NPF (Gulf of Carpentaria) that were associated with two of the fishery's target tiger prawn species (*Penaeus semisulcatus* and *Penaeus esculentus*). These assemblages were represented mainly by the teleost families Apogonidae, Synodontidae, Leiognathidae, Mullidae and Nemipteridae.

Our study indicates that the JBG bycatch should be regarded as a third major assemblage in the NPF. Its main teleost families are Synodontidae, Rhinoprenidae, Trichiuridae, Sciaenidae, Engraulidae and Polynemidae, which are represented by a suite of species that are not abundant in the Gulf of Carpentaria (in particular, *Harpadon translucens*, *Rhinoprenes pentanemus*, *Trichiurus lepturus*, *Polydactylus nigripinnis* and *Johnius laevis*). Furthermore, numerous species caught in the region have not been recorded in the prawn trawl bycatch from the Gulf of Carpentaria. These include the teleost species *Polydactylus nigripinnis*, *Setipinna paxtoni*, *Larimichthys pamoides*, *Benthoosema pterotum*, *Johnius laevis*, *Johnius cf trawavase*, *Lophichthys boschmai* and the invertebrate species (cephalopod) *Abralia amarta*. The composition of elasmobranchs in the JBG also appears to be distinctly different to other regions of the NPF. For example, the small stingray *Dasyatis annotata* was the most abundant in the JBG, occurring in 40% of samples, whereas the closely related *Dasyatis leylandi* was not recorded in our study but was the most abundant species in most other NPF regions (Stobutzki et al., 2002). Both of these are small species, with a disc width up to ~250 mm, so they are unlikely to have been excluded by TEDs.

It is estimated that an annual bycatch to prawn ratio of 13:1 exists for the JBG subfishery (calculated from Table 17). This falls between the 8:1 and 21:1 estimates provided previously for the NPF (Pender et al., 1992; Brewer et al., 1998). The teleost component of the bycatch biomass in the JBG approaches 91%, whereas it is 82% in other NPF regions (Stobutzki et al., 2001b). However, the teleost component of the Gulf of Mexico shrimp fishery bycatch is similar to the JBG at 92% (Garcia-Caudillo et al., 2000). Our data support anecdotal information from experienced fishers that JBG prawn trawls generally take large catches of fishes. The fishers believe an increase in prawn abundance coincides with a consistent and dramatic increase in pelagic and demersal aggregation of teleosts as the currents ease leading up to the neap tides (pers comm. NPF skippers M. Farrell, P. Williamson, C. Terjensen). As a result, fishing occurs close to these aggregations and therefore captures a large proportion of teleost species.

The JBG bycatch is also distinct from bycatch in other NPF regions, with the most abundant bycatch species being the portunid crab *Charybdis callianassa*. One trawl recorded an estimated 34,508 individuals h⁻¹ (biomass 211 kg h⁻¹). This portunid species often accounted for over 15% of the total bycatch biomass in individual trawls, and on two occasions contributed 33% and 60%. However, this species has not been recorded in large numbers in the Gulf of Carpentaria (Stobutzki et al., 2000). A closely related species, *Charybdis smithii*, also dominates catches in trawl fisheries along the Indian coast, averaging over 600 kg h⁻¹ (Thomas and Kurup, 2001). This oceanic species aggregates on the surface in October before

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migrating onto the continental shelf to reproduce during the night (Karcnik et al., 1997). Anecdotal evidence suggests that *C. callianassa* makes similar surface aggregations during September in the JBG (pers comm. M. Farrell; NPF skipper). The peak reproductive period for this species appears to be during September and October, with a higher proportion of large and berried crabs recorded than in April and May (CSIRO unpublished data). Very little is known about the biology and ecology of this species, but their large contribution to the biomass of the demersal species assemblage suggests they play a key ecological role in the region. Losse (1969) reported that swarming portunid crabs have a large predatory impact on the zooplankton, while Salini et al. (1994) and Griffiths et al. (2007) noted they are important prey for large predatory pelagic fishes in the NPF.

The differences in the species composition of the bycatch in the JBG and in other regions of the NPF may be in part explained by biophysical characteristics. According to a bioregionalisation study (IMCRA 1998), the JBG falls into the Bonaparte Gulf meso-scale, which is characterised by depths greater than 30 m and sediments dominated by biogenic gravels and sands grading to biogenic muds offshore. On the North-West Shelf of Australia, water depth and the particle size of sediments were found to be important factors in determining the abundance of decapod crustaceans (Ward and Rainer, 1988). The remainder of the NPF is categorised into another 13 meso-scale regions distinguished from each other, and the JBG, by factors that include depth, sediment type, tides, geology, biology and hydrology. These biophysical characteristics were found to influence bycatch composition across small spatial scales (Stobutzki et al., 2001b) and are likely to be similarly influential in determining the species composition of the JBG. The extremely large tidal range (~10 m spring tides) experienced in inshore areas of the JBG may also contribute to the dominance of a small number of teleosts species in the bycatch that can tolerate these conditions.

The JBG subfishery catches a significantly larger biomass and abundance of bycatch during spring (September–November) than in autumn (April–May). Several studies have also reported temporal changes in bycatch assemblages in tropical regions (Rainer and Munro, 1982; Wright, 1988; Blaber et al., 1990; Gallaway and Cole, 1999). For example, the composition of demersal trawl fauna on the Great Barrier Reef is influenced by ‘wet’ and ‘dry’ seasons (Watson et al., 1990), while variation in the composition of teleost bycatch in the southern Brazil pink shrimp fishery has been attributed to the seasonal movement of water masses (Vianna and Almeida, 2005).

In inshore waters of the Gulf of Carpentaria, seasonal variations in temperature and salinity may contribute to faunal assemblage structure (Rainer and Munroe, 1982). It is unknown whether temporal fluctuations of estuarine and inshore conditions in the JBG affect the species abundances there, or how they may be affected by temperature, salinity and tidal influences. In our study, *Rhinoprenes pentanemus*, *Trichiurus lepturus*, *Harpadon translucens* and *Polydactylus nigripinnis* accounted for the main differences in biomass between catches in spring and autumn. According to Carpenter and Niem (1998), these species are also known to exist in estuarine and inshore waters. Therefore, examining the influence of inshore abiotic factors on these species may help us to understand recruitment processes to offshore regions and explain the observed seasonal variations in bycatch.

The size structure of some bycatch species in the JBG also varied significantly between seasons. Many of the most abundant species had a higher proportion of smaller individuals in catches during autumn than in spring, suggesting recruitment is seasonal. However, *Rhinoprenes pentanemus* and *Harpadon translucens* were caught at similar sizes during both autumn and spring. This suggests they may spawn several times throughout the year; as does

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Harpadon nehereus, a closely related species to *H. translucens*, which spawns six times a year in Indian waters (Fernandez and Devaraj, 1996).

Our study did not find a difference between the bycatch biomasses caught at night and in the day. Other tropical studies have reported larger amounts of bycatch during daytime trawling (Blaber et al., 1990; Sivasubramaniam, 1990). These differences may be explained by the proportion of abundant species in the bycatch that exhibit diel changes in their vertical distribution. For example, Blaber et al. (1990) identified 15 species in the Gulf of Carpentaria that exhibited diel variation. These included a number of the more abundant species from the families Leiognathidae, Carangidae and Sciaenidae (Balan, 1967; Hobson, 1968; Potts, 1980; Brewer et al., 1989; Soares and Vazzoler, 2001). However, in our study, only two of the more abundant species *Thryssa setirostris* and *Pellona ditchela* had higher catch rates during the day. Squids were also significantly more abundant during the day; they, too, migrate to the surface at night (Yatsu et al., 1995; Carpenter and Niem, 1998), which makes them less vulnerable to demersal trawling.

There was less variation in the size composition of abundant bycatch species between the day and night, with the exception of *H. translucens* and *Trichiurus lepturus*, where smaller individuals were caught during the day. Nakamura and Parin (1993) demonstrated that adult and juvenile *T. lepturus* have alternating vertical diurnal feeding migrations. They found that large adults usually fed near the surface during the day and migrated to the bottom at night. In contrast, they found that the juveniles and small adults formed schools above the bottom during the day and then aggregated near the surface to feed at night. Diel feeding activity may also explain the size structure observed in day and night catches for the synodontid *H. translucens*. Yousif (2003) found that another synodontid *Saurida undosquamis*, in the Gulf of Suez shrimp trawl fishery, also exhibited similar diel size composition and suggested that feeding activity was a likely explanation.

Implications for bycatch management

From this study we were able to provide the first estimate of the level of fishing impact on the demersal communities of the JBG. Our estimate of around 5,000 t of annual bycatch shows that, despite the JBG occupying only 5% of the NPF managed area and having restricted fishing times (due to tidal influences on prawn abundance), it contributes about 17% to the 30,000 t of bycatch estimated to be taken each year in this fishery (Pender et al., 1992). This percentage is likely to be conservative considering the significant reduction in effort in other regions of the NPF since the 30,000 t estimate was made (Dichmont et al., in press), and the introduction of Bycatch Reduction Devices (BRDs) and Turtle Exclusion Devices (TEDs), which remove many of the larger animals from the catch (Brewer et al., 2006). This high bycatch take and the unique species assemblage with temporal variability is very different to other high trawl effort regions in the NPF, such as the Gulf of Carpentaria (see Stobutzki et al., 2001b) and will need to be considered independently within the long-term bycatch monitoring program currently being implemented in this fishery.

Conclusion

Trawl fisheries worldwide are being pressured to demonstrate greater ecological sustainability. This is a significant challenge for these fisheries whose bycatch is species-rich. For most trawl fisheries, there is little biological information and long-term catch data on each bycatch species to assess their population status by conventional population models. The practicality and high cost of monitoring the many species to evaluate their sustainability means that alternative methods, such as ecological risk assessment methods (Stobutzki et al.,

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2001a; Zhou and Griffiths, in press) are required. These methods are able to cope with the limited biological data relying more heavily on the distribution of impacted species and their susceptibility to capture in space and time. The information in the present study will be critical not only for northern Australian trawl fisheries, but also for other tropical trawl fisheries in the Indo–Pacific region.

Acknowledgements

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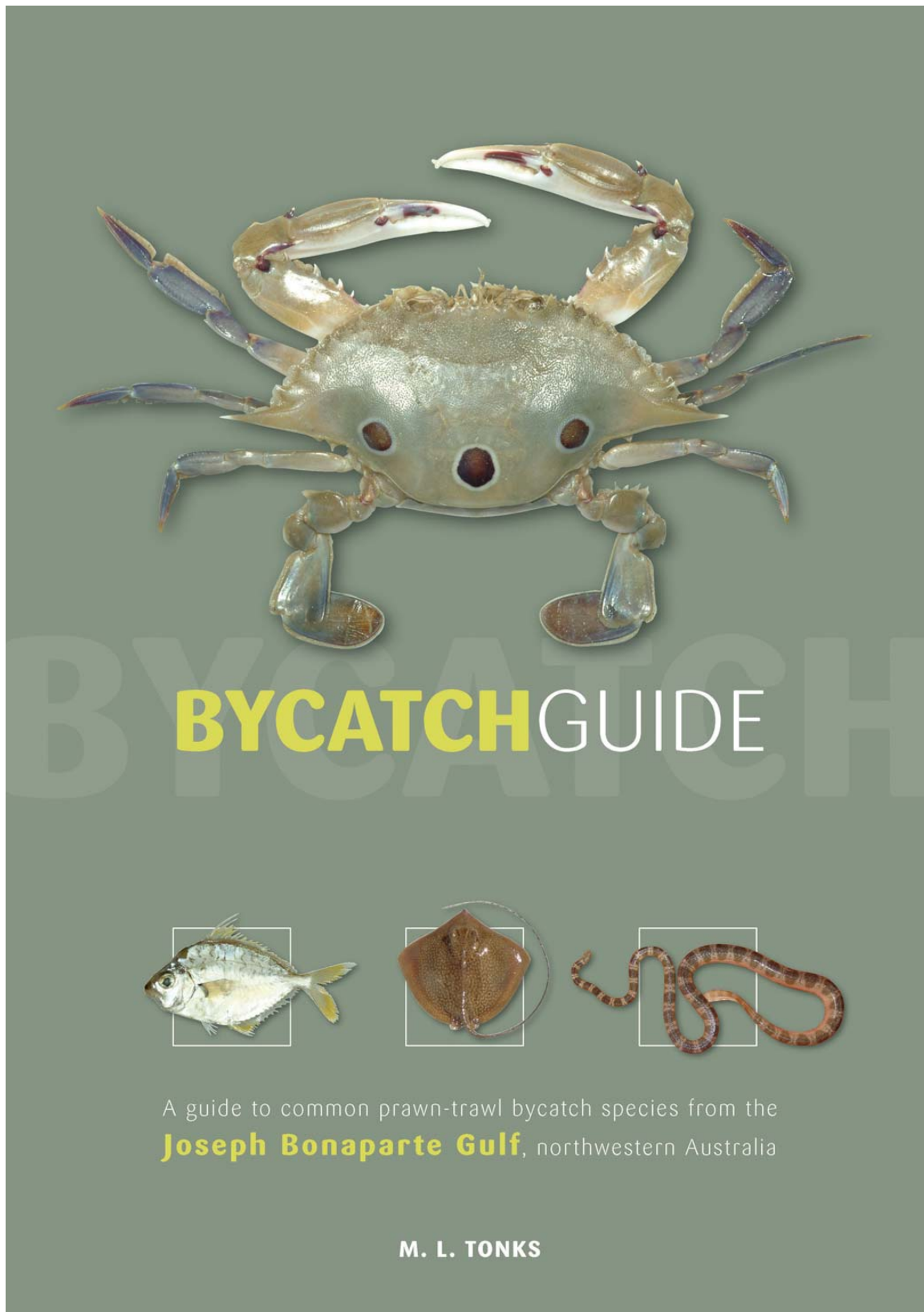
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5.5.3 JBG bycatch guide







BYCATCHGUIDE

A guide to common prawn-trawl bycatch species from
the **Joseph Bonaparte Gulf**, northwestern Australia

M. L. Tonks

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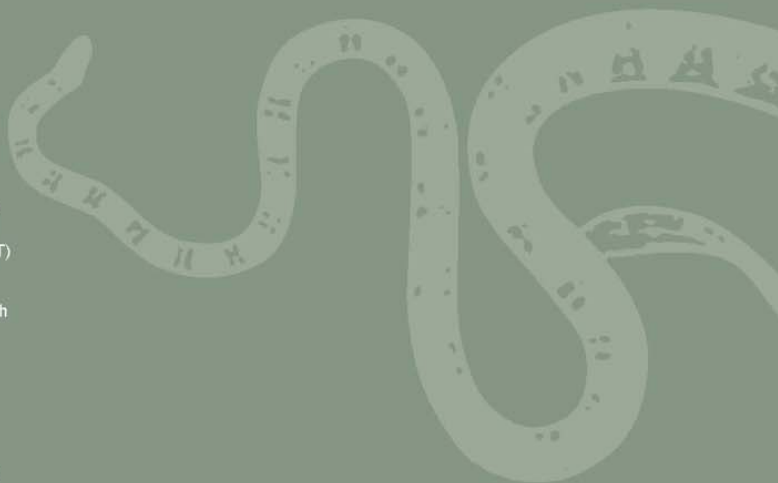
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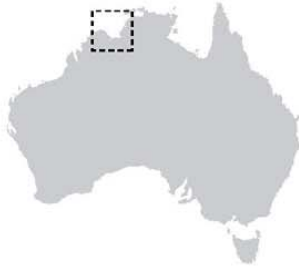
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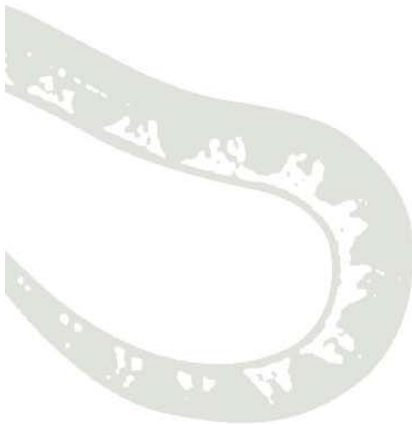
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Foreword



The Joseph Bonaparte Gulf (JBG) comprises a geographically small but important component of the Commonwealth-managed Northern Prawn Fishery (NPF). It is situated on the border of Western Australia and the Northern Territory in the Timor Sea. The JBG fishery targets the 'redleg' banana prawn and contributes about 65% of the annual NPF catch for this species.



The bycatch assemblage associated with this fishery is distinctly different from other NPF regions, particularly the Gulf of Carpentaria.

Bycatch is a major issue in all trawl fisheries. The Northern Prawn Fishery (NPF) is no exception. It is a tropical demersal trawl fishery that has significant bycatch. The NPF has already significantly reduced the amount of bycatch in the fishery through the development and compulsory use of Turtle Exclusion Devices (TEDs) and Bycatch Reduction Devices (BRDs). To this extent the catch of turtles and other large animals such as sharks and rays has decreased by over 97%. Bycatch reduction work in the fishery is ongoing, especially in relation to the smaller bycatch that is caught in conjunction with the target prawn species.

To reduce bycatch, it is necessary to first identify and quantify its composition over time and area. To assist in the gathering of this information, the fishery needs to accurately report what is being caught. Much of this responsibility lies with the NPF operators in completing their fishery logbooks. Of course fishers cannot accurately record all of their bycatch, especially some of the less common species; this is the role of observers and scientists.

This guide will assist industry, scientists, observers and managers to obtain more accurate information on the bycatch encountered in the JBG. It will improve our knowledge of this fishery and will allow for more informed management.

I applaud the development and production of this guide and am sure that it will be of great benefit to the fishery.

Wade Whitelaw

*Manager: Northern Prawn and Western Trawl Fisheries
Australian Fisheries Management Authority (AFMA)*

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 Cephalopods (cuttlefishes and squids) 49

Common name	Family	
Elasmobranchs		
butterfly rays	Gymnuridae	12
hammerhead sharks	Sphyrnidae	12
longtail carpet sharks	Hemiscylliidae	13
sawfishes	Pristidae	13
stingrays	Dasyatidae	14
weasel sharks	Hemigaleidae	15
wedgefishes	Rhinidae	16
whaler sharks	Carcharhinidae	16
Teleost fishes		
anchovies	Engraulidae	17
batfishes	Ephippidae	19
bombay ducks	Synodontidae	20
cardinal fishes	Apogonidae	20
codlets	Bregmaceroidae	22
cuskeels	Ophidiidae	22
eeltail catfishes	Plotosidae	23
flatheads	Platycephalidae	23
forktailed catfishes	Ariidae	24
goatfishes	Mullidae	25
grunter breams	Haemulidae	25
grunters	Terapontidae	26
hairtails	Trichiuridae	26
ilishas	Pristigasteridae	27
jewfishes	Sciaenidae	28
lanternfishes	Myctophidae	30
leatherjackets	Monacanthidae	30
lefteye flounders	Bothidae	31
lizardfishes	Synodontidae	31
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ocean basses	Acropomatidae	32
pikes	Sphyraenidae	33
ponyfishes	Leiognathidae	33
porcupinefishes	Diodontidae	34
rockcods	Serranidae	35
sand flounders	Paralichthyidae	35
searobins	Triglidae	36
toadfishes	Tetraodontidae	36
tongue soles	Cynoglossidae	37
threadfin breams	Nemipteridae	38
threadfin salmon	Polynemidae	38
trevallas	Centrolophidae	39
trevallies	Carangidae	39
tripodfishes	Triacanthidae	41

Common name	Family	
Reptiles		
sea snakes	Hydrophiidae	41
Crustaceans		
swimming crabs	Portunidae	42
prawns	Penaeidae, Solenoceridae	45
mantis shrimps	Squillaeidae	48
Cephalopods		
squids	Enoploteuthidae	49
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Introduction

The worldwide management of bycatch has become increasingly important in recent years. In Australia, the *Fisheries Management Act (1991)* and the *Environmental Protection and Biodiversity Conservation Act (1999)* have ensured that both the target species and the bycatch are now managed in an ecologically sustainable manner. Fishers, scientists and the general public alike understand the need to manage fisheries at this broader ecosystem level.

Recent bycatch research in the Northern Prawn Fishery (NPF) has focused on the prawn trawl bycatch of the Gulf of Carpentaria, at the expense of other NPF regions such as the Joseph Bonaparte Gulf (JBG). Prawn trawling in the JBG began in the early 1980s when commercial quantities of the 'redleg' banana prawn, *Penaeus indicus*, were discovered. This region provides a consistently high proportion of the total NPF banana prawn catch each year.

In order to address the regional imbalance in bycatch research within the NPF, the Fisheries Research and Development Corporation (FRDC) funded the project '**Design, trial and implementation of an integrated long-term bycatch monitoring program, road tested in the NPF**' (FRDC 2003 CMR154). One of the main objectives of this project was to describe the bycatch of the

JBG. As co-investigators, CSIRO and AFMA collected and analysed around 1500 kg of bycatch from 53 trawl samples. These samples were collected from depths that ranged between 45 and 68 metres. The information obtained has led to a much better understanding of the geographical distribution, relative abundance and quantities of bycatch species taken by the fleet. Such baseline information will be extremely useful for future bycatch management purposes.

I have produced this guide to assist other scientists in the identification of

the common bycatch species from this region, and also to assist fishers in the identification of the many interesting animals that they catch.

The guide provides colour photographs, important taxonomic features, size and distributional information for the common prawn trawl bycatch species. It also provides a checklist of all bycatch species collected during this project, including teleost fishes, crustaceans (prawns, crabs, mantis shrimps), reptiles (sea snakes) and elasmobranchs (sharks and rays).



glossary

Adipose eyelid – transparent fleshy tissue covering part of or the entire eye in some fishes.

Anal fin – unpaired median fin, supported by rays on the ventral tail behind the anus (Fig. 4).

Anterior – referring to the front portion.

Arborescent – branched; having shape like a tree.

Axil – angular area between pectoral fin and body, equivalent to armpit.

Barbel – elongate, fleshy, tentacle-like sensory projection, usually about mouth or head.

Branchiostegal rays – bony rays supporting the membranes inside the lower part of the gill cover

Breast – ventral surface of body between the isthmus and pectoral or pelvic fins.

CAAB – Codes for Australian Aquatic Biota.

Canine – slender conical tooth, often enlarged and elongate.

Carapace – a hardened encasing covering all or part of the body (refers to crustaceans).

Caudal fin – the tail fin (Fig. 4).

Caudal peduncle – posterior part of body between the rear parts of the dorsal and caudal fin.

Cheek – side of head below and slightly behind the eye.

Clupeiform – refers to fish like herrings, sardines and anchovies.

Crenate – with scalloped margin.

Ctenoid scale – scale with a spiny posterior margin (Fig.11).

Cusp – a projection (point) on a tooth (refers to sharks).

Cusplet – small cusp.

Cycloid scale – scales with smooth posterior margin, without spines on posterior margin (Fig. 11).

Denticles – the tooth-like scales of sharks and rays.

Digitated – having digits or fingerlike projections.

Distal – near outer edge; far end from point of attachment or centre of body.

Dorsal – back or upper body.

Dorsal fin – median fin supported by spines and/or rays; sometimes separated into 2 or more fins with the anteriormost fin designated the first.

Finlets – small, separate dorsal and anal fins.

Frontal bone – the superficial, paired dermal bone on top of the skull above the eyes.

Gill arch – bony angular skeleton that supports the gill filaments and gill rakers (Figs. 6 & 7).

Gill rakers – bony projections along the front edge of the gill (Figs. 6 & 7).

Hyomandibular pores – line of enlarged pores extending posteriorly from the mouth corners (refers to sharks).

Insertion – the posterior point of attachment of a fin to the body.

Interopercle – the lower anterior bone of the gill cover (Fig. 4).

Isthmus – the part of the underside of the head separating the gill openings.

Keel – sharp or strong ridge.

Labial furrows – shallow grooves around the lips (refers to sharks).

Lateral – the side or towards the side.

Lateral line – the line, or system of lines, of sensory structures along the head and sides of fishes.

Lunate – crescent-shaped, caudal-fin shape that is deeply emarginated with narrow lobes (Fig. 8).

Mantle – the fleshy (muscular) tubular or sac-like body of cephalopods.

Maxilla – the bone in the upper jaw behind the premaxilla.

Melanophore – cell carrying black or greyish pigments.

Mental pores – sensory pores on the chin.

Nape – the dorsal part of the body just behind the occiput or hard dorsal region of the skull.

Occiput – upper or back part of the skull.

Ocellus – a round, eye-like spot or marking with a marginal ring.

Opercle – the large posterior upper bone of the gill cover (Fig. 4).

Operculum – the gill cover composed of the preopercle, opercle, interopercle and subopercle.

Orbital – referring to the eye.

Origin – the anterior point of attachment of the fins to the body.

Palate – roof of the mouth.

Papillae – small fleshy projections.

Perciform – refers to perch-like fish.

Objective 4: Bycatch descriptions of banana prawn fisheries

Petasma – the male appendage for transferring sperm packets to the thelycum of the female. It looks like a penis and is formed from the inner parts of the first pair of pleopods.

Photophore – light-emitting organ or luminous spot.

Premaxilla – anterior bone in the upper jaw (Fig. 4).

Proximal – a point on an appendage closer to the main body.

Ray – supporting element of fins (Fig. 10).

Rugose – having wrinkles, creases or ridges.

Scute – a modified scale that can be enlarged, hardened, ridged, keeled, or spiny.

Serrate – with saw-like teeth along margin.

Skin folds – an area where skin is bent over upon itself, forming a fleshy ridge.

Soft dorsal fin – the portion of the dorsal fin supported by soft rays.

Spine – a fin support element that is unpaired laterally, unsegmented, unbranched and usually stiff and pointed, also refers to slender, sharply pointed bony processes not associated with fins.

Spinous dorsal fin – the anterior portion of the dorsal fin that is supported by spines.

Spiracle – a respiratory opening behind the eye (refers

to sharks and rays).

Subopercle – the lower, rear bone in the gill cover (Fig. 4).

Subquadrate – nearly or approximately square

Supramaxilla – small bone above the maxilla (Fig. 14).

Swimbladder – a gas-filled sac located in the dorsal areas of the abdominal cavity which is used by bony fish to regulate buoyancy.

Symphysis – the articulation between 2 bones; often refers to the anterior juncture between the 2 halves of either jaw.

Terminal – referring to the end, or situated at the end; a mouth position with the opening of the mouth forming the tip of the snout.

Thelycum – female external receptacle for sperm packets, situated between the fourth and fifth pair of legs.

Transverse septum – is the sheet-like tissue that separates the heart from the abdominal cavity.

Truncate – terminating abruptly in a square end, a caudal-fin shape with a vertically straight terminal border (Fig. 8).

Tuberculate – has bumps or projections.

Ventral – the bottom, lower surface, or abdominal part of the body.

Villiform – many small slender outgrowths, usually in a close-

set patch or carpet; often refers to slender teeth forming velvety bands (Fig. 12).

Vomer – An unpaired median bone on the roof of the mouth (Fig. 9).

List of measurement abbreviations

CL Carapace length (refer Fig. 17)

CW Carapace width (refer Fig. 18)

DW Disc width (refer Fig. 2)

FL Fork length (refer Figs. 1 & 3)

ML Mantle length (refer Fig. 25)

SL Standard length (refer Fig. 3)

SV Snout-vent length

TL Total length (refer Figs. 1 & 3)

Size used in this guide refers to the maximum size found for the species in this study.

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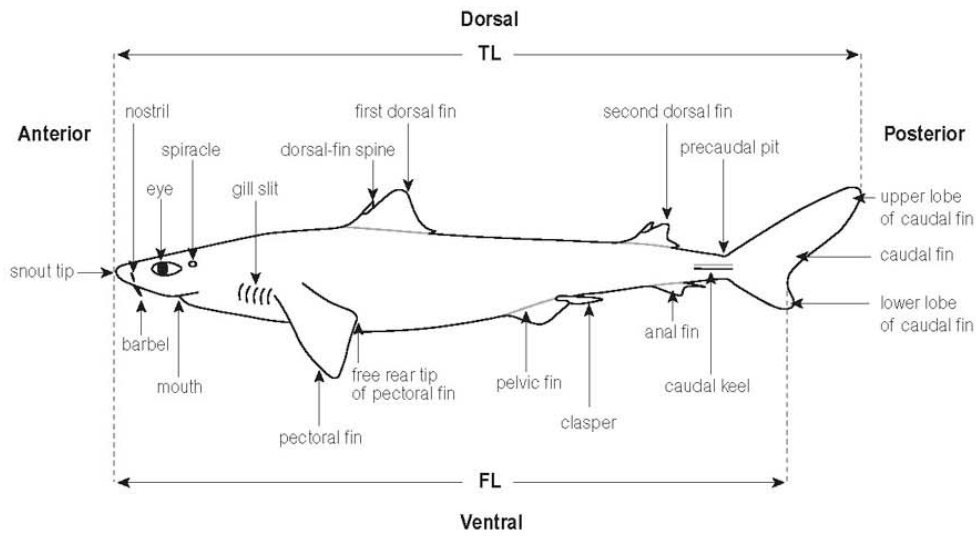


Figure 1: structural features of a generalised shark.

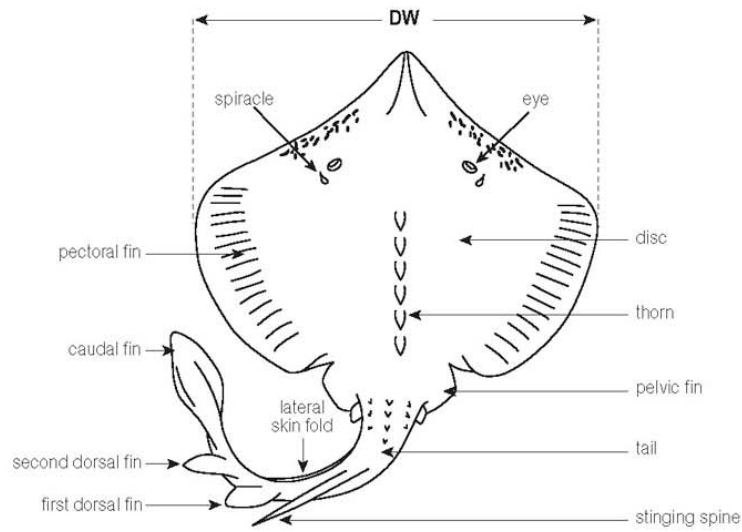


Figure 2: structural features of a generalised ray.

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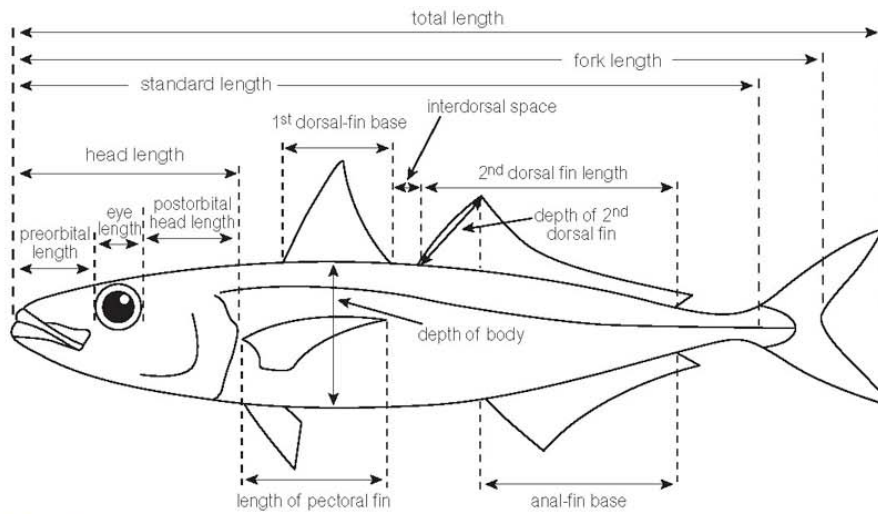


Figure 3: common external measurements of a finfish.

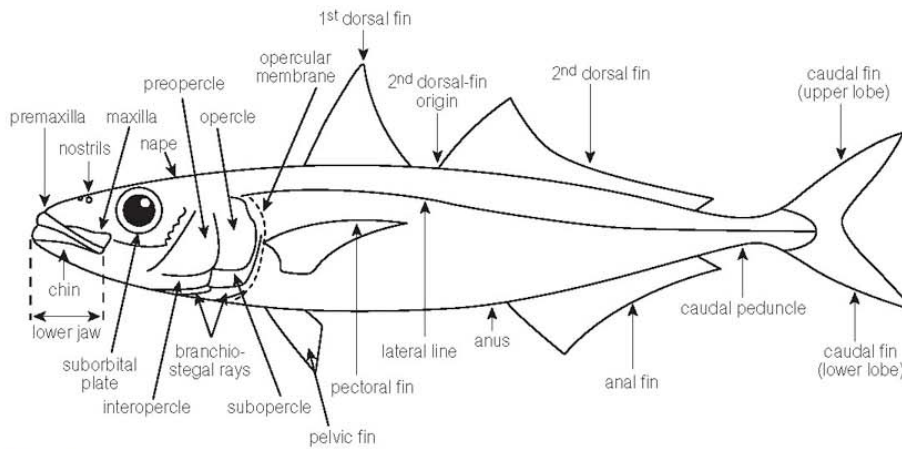


Figure 4: common external features of a finfish.

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Bycatch Guide 5

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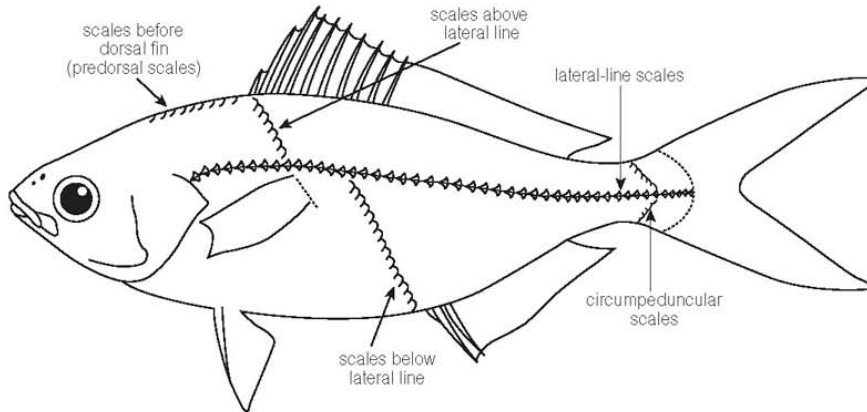


Figure 5: common scale counts on a finfish.

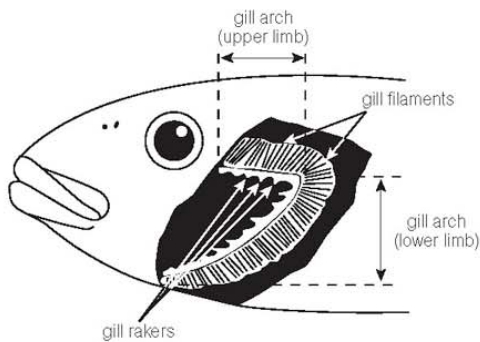


Figure 6: position of 1st left gill arch with gill cover removed.

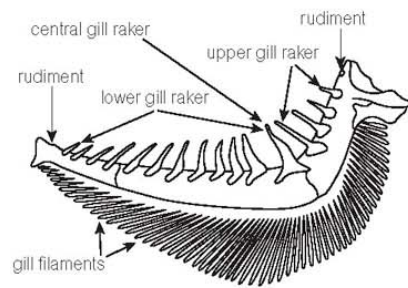


Figure 7: structures of 1st left gill arch.

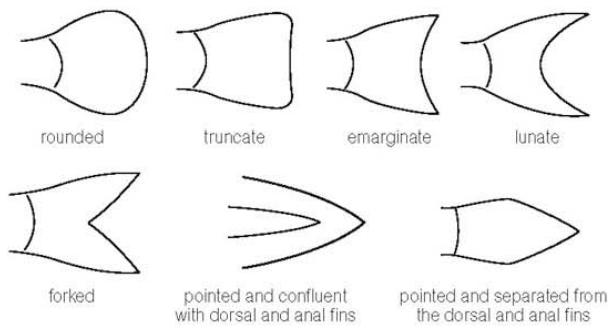


Figure 8: most common types of caudal fins.

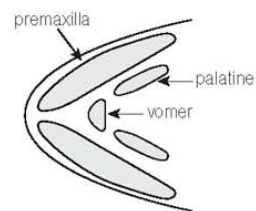


Figure 9: teeth-bearing bones in the roof of the mouth and upper jaw.

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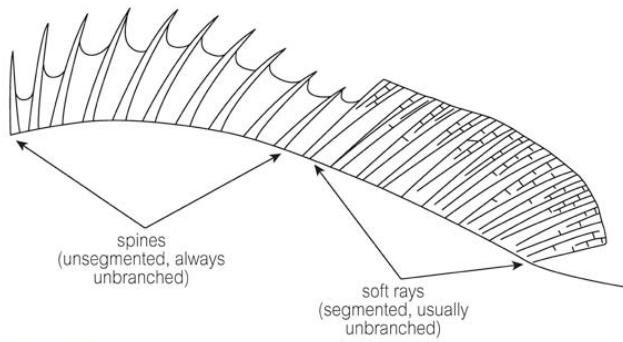


Figure 10: example of a continuous dorsal fin of a spiny-rayed fish.

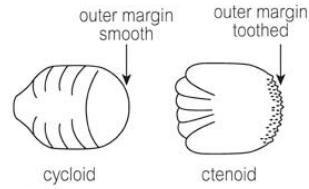


Figure 11: schematic examples of typical scales.

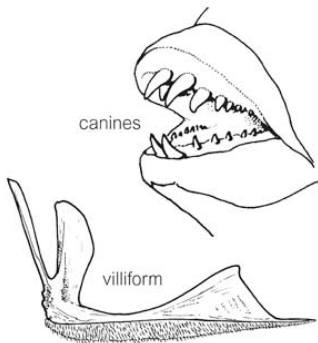


Figure 12: common types of teeth.

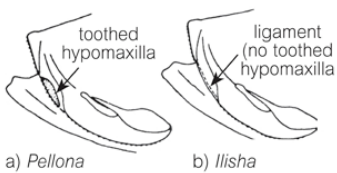


Figure 14: lateral view of mouth (Pristigasteridae).

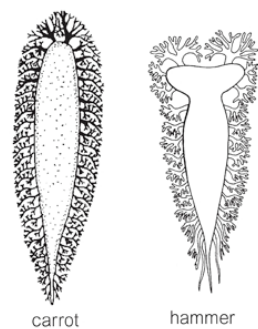


Figure 15: swimbladder shapes.

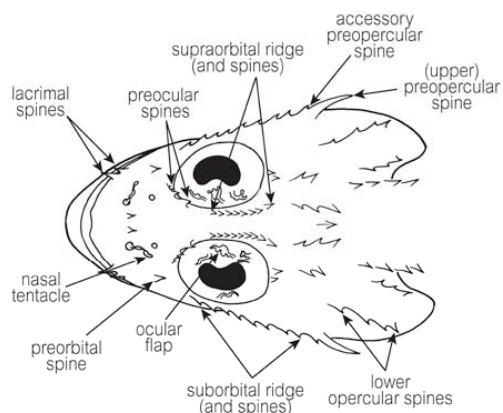


Figure 13: diagram of head spines for Platycephalidae.

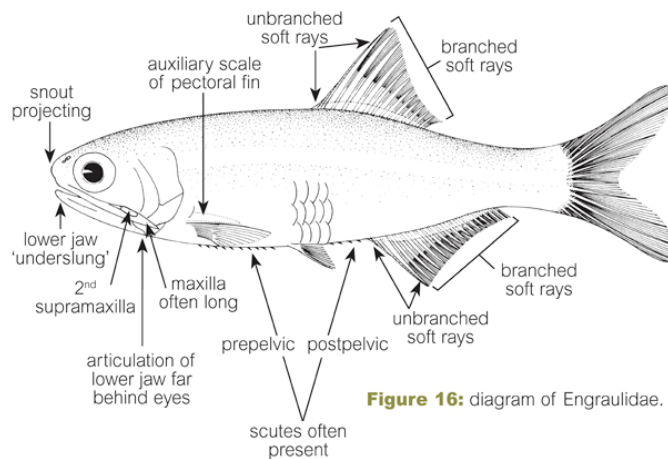


Figure 16: diagram of Engraulidae.

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Objective 4: Bycatch descriptions of banana prawn fisheries

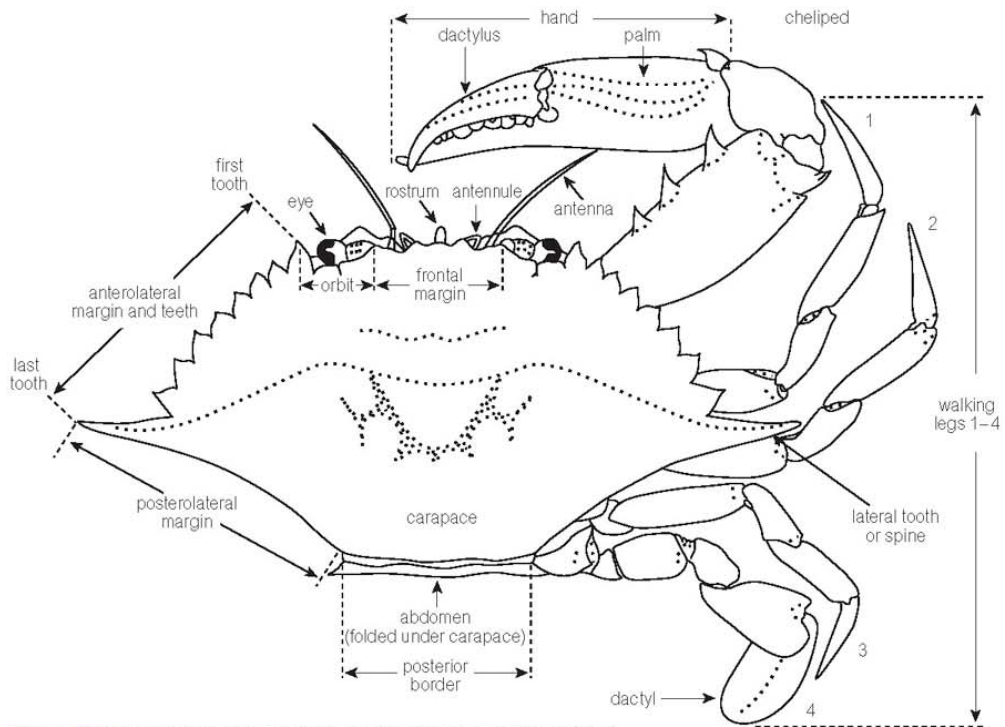


Figure 17: general shape (dorsal view) of a brachyuran crab (Portunidae).

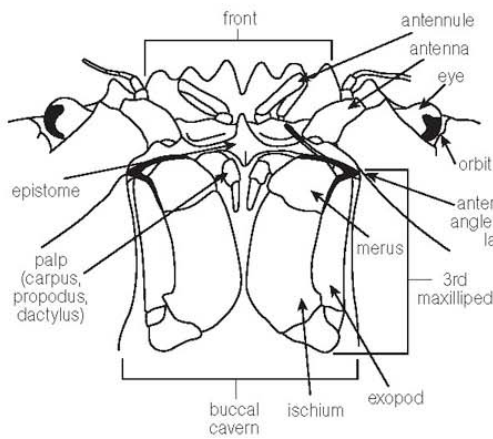


Figure 18: mouth of brachyuran crab.

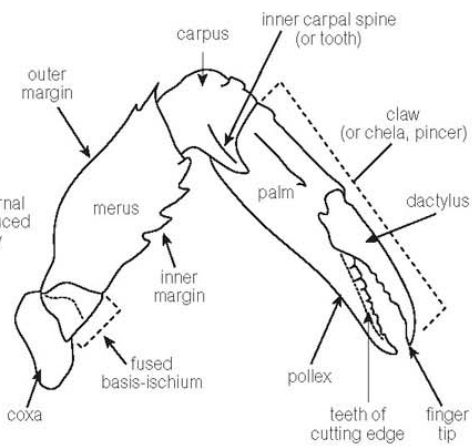


Figure 19: cheliped of brachyuran crab.

Objective 4: Bycatch descriptions of banana prawn fisheries

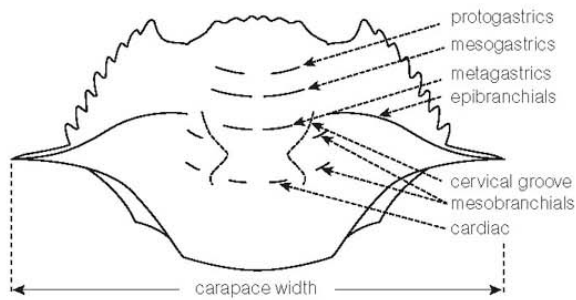


Figure 20: location of ridges and depressions of carapace in species of *Portunus* (also relates to species of *Charybdis*).

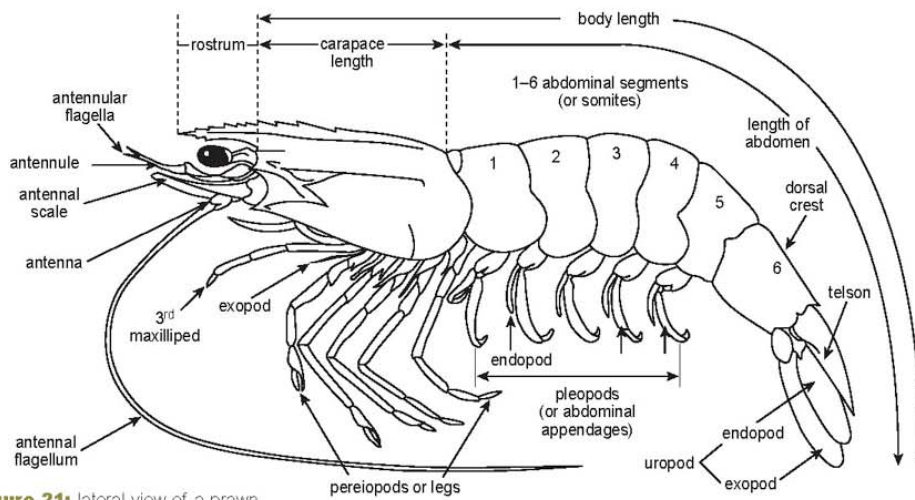


Figure 21: lateral view of a prawn.

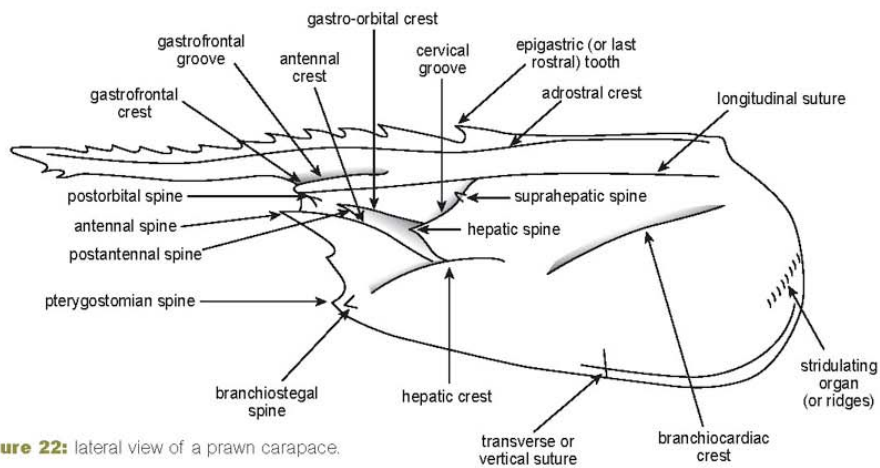


Figure 22: lateral view of a prawn carapace.

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Objective 4: Bycatch descriptions of banana prawn fisheries

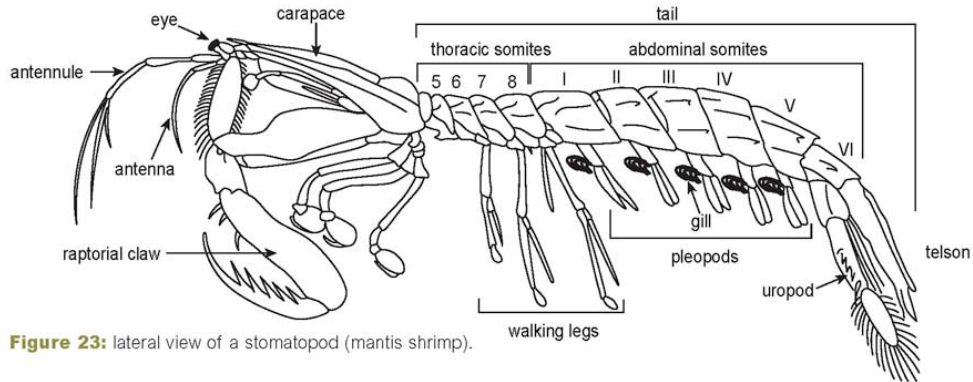


Figure 23: lateral view of a stomatopod (mantis shrimp).

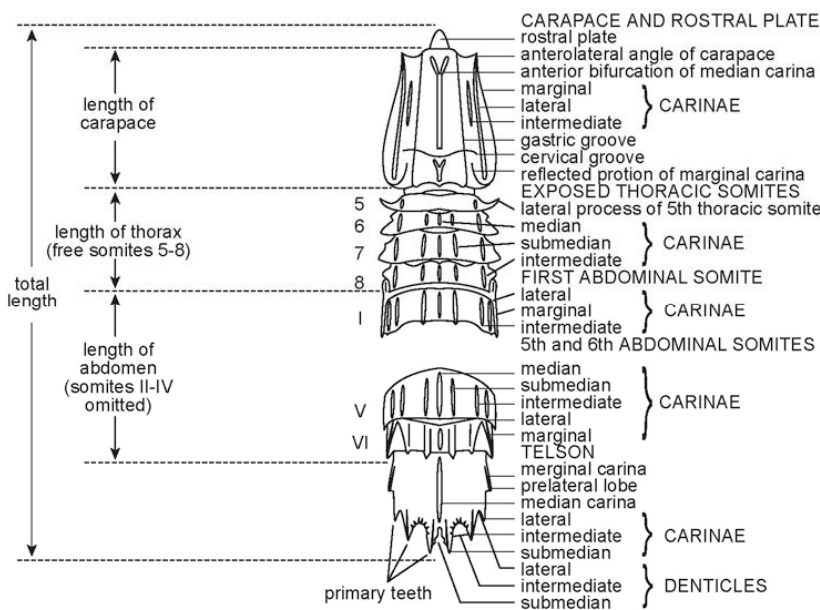


Figure 24: dorsal view of mantis shrimp (without head and extremities).

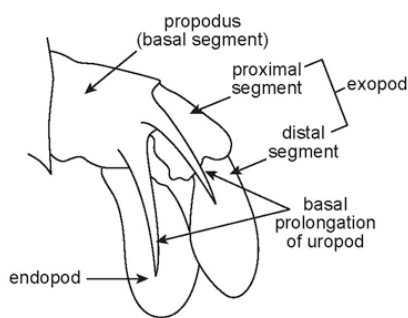


Figure 25: left uropod of mantis shrimp (ventral view).

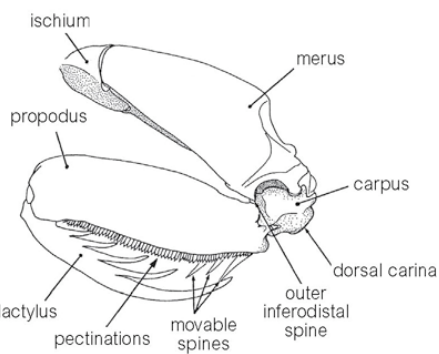


Figure 26: mantis shrimp raptorial claw.

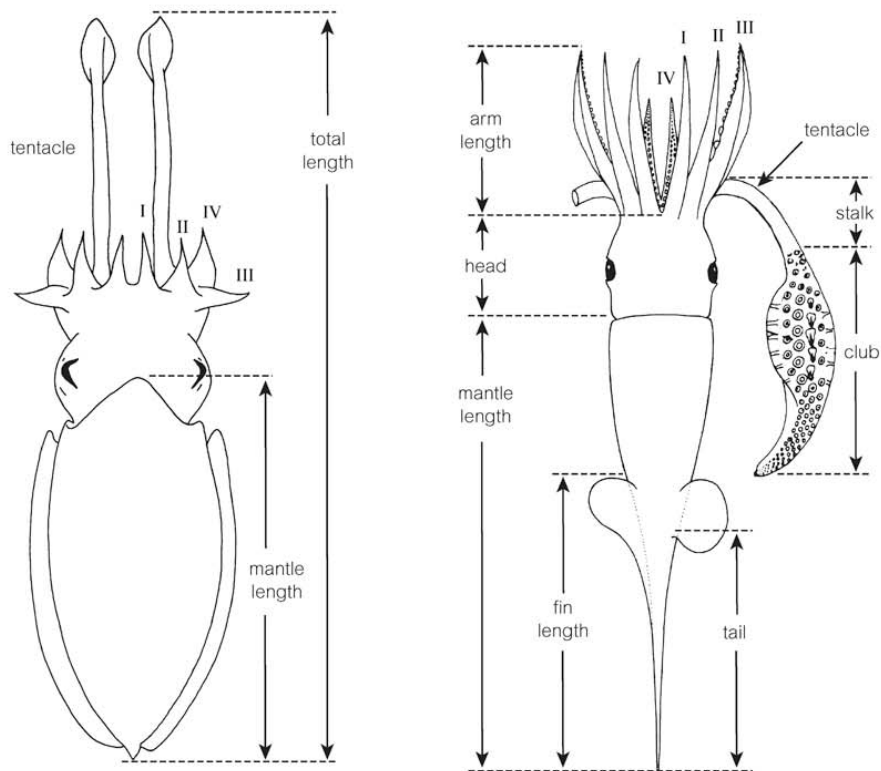


Figure 27: measurements, arm numbering, and major external features of cephalopods in dorsal view. Note: Orientation is relative to the resting animal, the arms and arm/tentacle crown being anterior. The body or mantle is considered posterior. The upper surface of the resting animal is considered dorsal and the underside ventral. Arms are numbered as left to right, commencing from the dorsal arms.

BUTTERFLY RAYS FAMILY: Gymnuridae

Australian Butterfly Ray *Gymnura australis*

CAAB: 37 037001

Specimen: 29.2 cm DW

Diagnostic characters

- Body much wider than long
- Usually no stinging spine
- Single small dorsal fin
- Tail extremely short and filamentous

Colour

- Mostly greenish grey above (sometimes greyish or yellowish) with a dense peppering of fine black spots over a delicate mosaic pattern; irregular blotch near each pectoral fin insertion
- Ventral surface white
- Tail with alternating black and white bands



Size

- Maximum disc width 73 cm (commonly caught in prawn trawls less than 40 cm DW)

Distribution

- Northern half of Australia and southern New Guinea
- Reference: Last and Stevens, 1994, p 443

HAMMERHEAD SHARKS FAMILY: Sphyrnidae

Winghead Shark *Eusphyra blochii*

CAAB: 37 019003

Specimen: 54.5 cm TL

Diagnostic characters

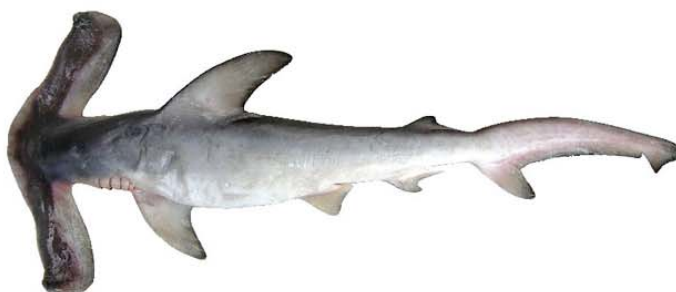
- Head shaped like a broad arrowhead or a pair of aircraft wings in dorsal or ventral view
- A shallow, but distinct indentation opposite each nostril, the edge of which has a row of low bumps; nostrils greatly elongated, wider than mouth
- First dorsal fin very high, strongly falcate

Colour

- Body grey or grey-brown above, paler below

Size

- Maximum total length 186 cm (commonly caught in prawn trawls less than 50 cm TL)



Distribution

- An Indo-West Pacific coastal species distributed from the Persian Gulf eastward to Pakistan, India, Sri Lanka, Thailand, Borneo, China, the Philippines, and northern Australia

Reference: Carpenter and Niem (eds), 1998, Vol 2, p 1363

LONGTAIL CARPET SHARKS FAMILY: Hemiscylliidae

Grey Carpetshark *Chiloscyllium punctatum*

CAAB: 37 013008

Specimen: 61 cm TL



Diagnostic characters

- Body moderately slender, without lateral ridges
- Two dorsal fins, somewhat larger than pelvic fins and with attenuated, projecting free rear tips; first dorsal fin over anterior halves of pelvic-fin bases
- Anal fin with its origin somewhat behind free rear tip of second dorsal fin

Colour

- Young with dark transverse bands
- Adults light brown, usually without a colour pattern

Size

- Maximum total length 100 cm (commonly caught in prawn trawls less than 70 cm TL)

Distribution

- Indo-West Pacific: India, Thailand, Malaysia, Singapore, Indonesia, Philippines, New Guinea, northern Australia, Vietnam, China, and Japan

Reference: Carpenter and Niem (eds), 1998, Vol 2, p. 1257

SAWFISHES FAMILY: Pristidae

Green Sawfish *Pristis zijsron*

CAAB: 37 025001

Specimen: unknown length



Diagnostic characters

- The saw has 24–28 pairs teeth on the rostrum, distinctly closer together near tip than near base
- Lower lobe of caudal fin very small

Colour

- Body greenish brown to olive above, pale whitish below

Size

- Maximum total length 500 cm

Distribution

- Widely distributed in the northern Indian Ocean, and off Indonesia and Australia

Reference: Last and Stevens, 1994, p. 366

Narrow Sawfish *Anoxypristis cuspidata*

CAAB: 37 025002

Specimen: 54.5 cm TL



Diagnostic characters

- The saw has 18–22 pairs of teeth with the basal third of rostrum without teeth
- Lower lobe of caudal fin more than half length of upper lobe

Colour

- Body greyish above, pale below
- Fins frequently pale
- Rostral teeth white

Size

- Maximum total length 103 cm

Distribution

- Indo-Pacific from the Red Sea to Australia, north to Japan

Reference: Last and Stevens, 1994, p 361

STINGRAYS FAMILY: Dasyatidae

Blackspotted Whipray *Himantura toshi*

CAAB: 37 035020

Specimen: 41 cm DW



Diagnostic characters

- Disc angular
- No caudal or dorsal fins
- No skin folds along dorsal or ventral surface
- Broad band of flat denticles along mid-disc
- Tail long, thin and whip-like
- Normally has a stinging spine

Colour

- Disc with light to medium covering of black spots (some variations have white spots)

Size

- Maximum disc width 74 cm (commonly caught in prawn trawls less than 50 cm DW)

Distribution

- New Guinea and tropical Australia, between Port Hedland (Western Australia) and Mackay (Queensland). Also off Indonesia

Reference: Last and Stevens, 1994, p 405

Plain Maskray *Dasyatis annotata*

Specimen: 22.8 cm DW

CAAB: 37 035012

Diagnostic characters

- A small stingray with rhomboidal disc
- Short thorns only on the midline of the disc and tail
- Prominent dorsal and ventral skin folds on the tail

Colour

- Dorsal surface dull greyish green with dark transverse bars about the eyes
- Ventral surface white
- Tail with variable black and white bands behind stinging spine; usually black at tip

Size

- Maximum disc width 35 cm

Distribution

- Eastern Indian Ocean: Timor Sea. Western Pacific: Arafura Sea and off northern Australia



Reference: Last and Stevens, 1994, p 386

WEASEL SHARKS FAMILY: Hemigaleidae

Weasel Shark *Hemigaleus australiensis*

Specimen: 46 cm TL

CAAB: 37 018020

Diagnostic characters

- Body elongate, trunk weakly compressed, oval in cross section at first dorsal-fin base; predorsal ridge absent; definite interdorsal ridge, extending just over half distance from first dorsal fin to second dorsal fin; post dorsal ridge absent; caudal peduncle is slender, cylindrical and tapering without lateral keels
- Second dorsal fin is about 2/3 the height of the first dorsal fin; posterior margin is deeply concave
- Origin of anal fin is slightly behind the origin of the second dorsal fin

- Precaudal pit is present
- Vertebral count 112–121

Colour

- Body light grey to greyish-brown above, whitish below
- Tips of second dorsal fin and upper caudal fin dark, fading in larger specimens; posterior margin of first dorsal fin pale

Size

- Maximum total length 110 cm (commonly caught in prawn trawls less than 100 cm TL)

Distribution

- Occurs off northern Australia from Geraldton (Western Australia) to Brunswick Heads (New South Wales)

Reference: White *et al.*, 2005, p 37



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WEDGEFISHES FAMILY: Rhinidae

Whitespotted Guitarfish *Rhynchobatus australiae*

CAAB: 37 026001

Specimen: 49 cm TL

Diagnostic characters

- Pectoral fins joined to head to form a wedge-shaped disc
- First dorsal fin above pelvic fins
- Pelvic fins separated by a narrow space from disc
- Lower lobe of caudal fin more than half length of upper lobe

Colour

- Upper surface, and dorsal and caudal fins, mostly greyish to yellowish brown
- 10–30 distinctive white spots (mostly less than half eye diameter in size and sometimes dark edged) extending from mid-pectoral fin along side of body to posterior tip of first dorsal fin; sometimes with additional white spots below first dorsal fin and usually with a prominent white spot on snout midline just in front of the eye



- 1–2 black spots above base of pectoral fin

Size

- Maximum total length 80 cm

Distribution

- Continental shelf of tropical and warm temperate Australia from Fremantle (Western Australia) to Coffs Harbour (New South Wales)

Reference: Last and Stevens, 1994, p 297

WHALER SHARKS FAMILY: Carcharhinidae

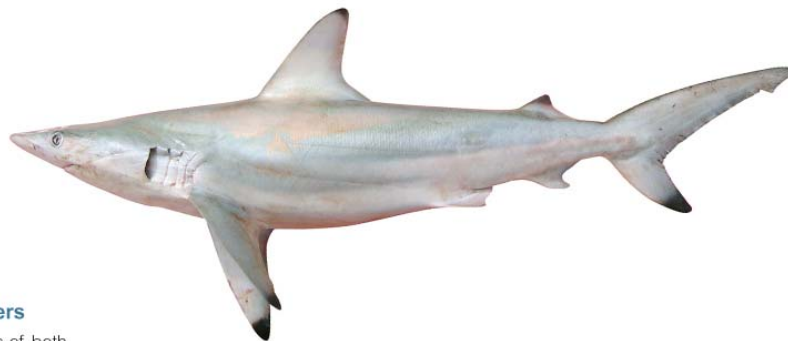
Australian Blacktip Shark *Carcharhinus tilstoni*

CAAB: 37 018014

Specimen: 86 cm TL

Diagnostic characters

- Black markings on tips of both dorsal fins, and the pectoral and lower caudal fins
- No ridge on back between dorsal fins
- Has a precaudal pit
- Upper teeth slender, erect and pointed



Colour

- Dorsal surface bronze, fading to grey after death; ventral surfaces pale. A pale stripe extends along each flank from the pelvic fin to below the first dorsal fin. All fins (except sometimes the pelvic and anal fins) black-tipped

Size

- Maximum total length 200 cm (commonly caught in prawn trawls less than 100 cm TL)

Distribution

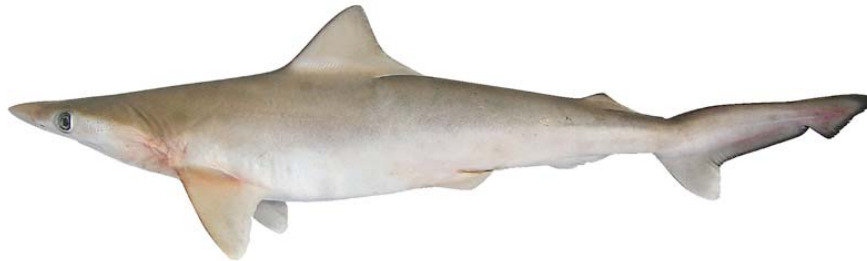
- Continental shelf of tropical Australia

Reference: Last and Stevens, 1994, p 265

Milk Shark *Rhizoprionodon acutus*

CAAB: 37 018006

Specimen: 41.8 cm TL



Diagnostic characters

- Small whaler shark with the second dorsal fin originating well behind the anal-fin origin
- Long labial furrows
- Usually more than 16 hyomandibular pores (total for both sides of the head)
- Teeth narrowly triangular, oblique and smooth-edged

Colour

- Dorsal surfaces bronze to greyish; ventral surface pale; pectoral, pelvic, anal and lower caudal-fin tips pale; pectoral-fin posterior margins pale; dorsal and upper caudal-fin tips dark in juveniles, sometimes dark-edged in adults

Size

- Maximum total length 100 cm (commonly caught in prawn trawls less than 100 cm TL)

Distribution

- Mainly tropical areas of the eastern Atlantic and Indo-West Pacific. Northern Australian waters from Fraser Island (Queensland) to Shark Bay (Western Australia)

Reference: Last and Stevens, 1994, p 265

ANCHOVIES FAMILY: Engraulidae

Common Hairfin Anchovy *Setipinna tenuifilis*

CAAB: 37 086008

Specimen: 126 mm SL

Diagnostic characters

- Body strongly compressed (depth 27–33% of standard length), abdomen with complete series of keeled scutes from isthmus to anus
- 18–20 (rarely 17 or 21) prepelvic and 7 (sometimes 8) postpelvic scutes; total number of scutes 25–27 (see fig. 14)
- A small spine-like scute just anterior to dorsal-fin origin
- First supramaxilla absent
- 13–17 lower gill rakers
- Dorsal fin moderately long, its origin about at body midpoint
- Anal fin long, with 3 unbranched and 46–56 branched fin rays
- Pectoral fins with first fin ray produced as a long filament that reaches posteriorly to the base of 9th to 21st anal-fin ray; total pectoral fin rays 11–13 (mostly 12)



- Pelvic fins not reaching posteriorly to anus
- No scales on dorsal or anal fins

Colour

- Head gold, body yellow and stippled above; silvery-white below
- Charcoal borders on scales along dorsal profile

- Dorsal and caudal fins yellow, stippled charcoal; heavy stippling on borders of fins; anal, pectoral and pelvic fins pale yellow

Size

- Maximum standard length 188 mm

Distribution

- Western Pacific (northern coast of Australia, Gulf of Papua). Indian Ocean (off northwestern Australia)

Reference: Carpenter and Niem (eds), 1998, Vol 3, p 1725

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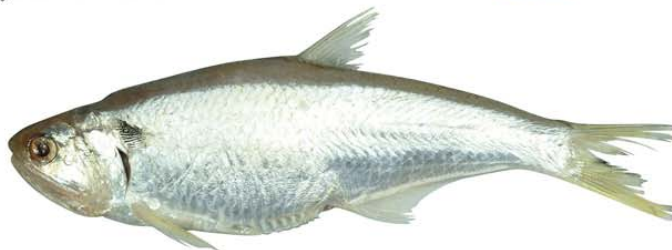
Hamilton's *Thryssa* *Thryssa hamiltonii*

CAAB: 37 086005

Specimen: 170 mm SL

Diagnostic characters

- Abdomen with 16–19 (rarely 15 or 20) prepelvic and 10–11 (rarely 9 or 12) postpelvic scutes; total number of scutes 26–31 (mostly 27 or 29)
- A small, spine-like scute present just anterior to dorsal-fin origin
- Tip of snout above level of eye centre, usually level with upper rim of eye
- Maxilla short to moderate, extending to or slightly beyond border of gill cover; first supramaxilla small and oval
- 12–14 lower gill rakers (less often 11 or 15)
- Anal fin relatively long, with 3 (rarely 4) unbranched and 32–39 (mostly 35 to 37) branched fin rays



Colour

- Head with gold tints, especially on maxilla and opercle
- Body silvery white, olive grey above, with pigment lines along dorsum and a dark blotch of horizontal, wavy black lines on shoulder just posterior to upper part of gill opening
- Dorsal fin dusky yellow, first dorsal fin ray and posterior border of fin charcoal; anal fin white; pectoral and pelvic fins pale yellow; caudal fin yellow, its upper and lower borders charcoal

Size

- Maximum standard length 202 mm

Distribution

- Widespread in western Pacific (Indonesia east to Papua New Guinea and northern coasts of Australia including the Gulf of Carpentaria to Gladstone, Queensland). Also found on shelf off northwestern Australia

Reference: Carpenter and Niem (eds), 1998, Vol 3, p 1748

Longjaw *Thryssa* *Thryssa setirostris*

CAAB: 37 086004

Specimen: 124 mm SL

Diagnostic characters

- Abdomen with 16–18 prepelvic and 9 or 10 postpelvic scutes; total 25–28 scutes
- A small, spine-like scute present just before dorsal-fin origin
- Maxilla very long (increases with size of fish), reaching posteriorly at least to tip of pectoral fins, usually to pelvic-fin base, or even to anal-fin origin
- 10–12 lower gill rakers
- Dorsal fin usually with 12 (rarely 10 or 11) branched fin rays
- Anal fin with 3 unbranched and 29–36 (mostly 32 to 35) branched fin rays



Colour

- Head with gold tints over pale blue
- Body silver white, bluish-grey above
- A diffuse patch of horizontal wavy grey or charcoal lines on shoulder posterior to upper part of gill opening
- Dorsal and caudal fins dusky to deep yellow, borders charcoal; anal fin deep yellow or white; pectoral and pelvic fins pale yellow

Size

- Maximum standard length 215 mm

Distribution

- Very widespread in western Pacific; northern Australia including Gulf of Carpentaria to Gladstone (Queensland) and on shelf off northwestern Australia

Reference: Carpenter and Niem (eds), 1998, Vol 3, p 1753

BATFISHES FAMILY: Ehippidae

Shortfin Batfish *Zabidius novemaculeatus*

CAAB: 37 362003

Specimen: 90 mm SL

Diagnostic characters

- Dorsal-fin spine with 9 spines and 27–29 rays, the anterior spines exposed, the last spine longest but only about half length of longest rays
- Soft dorsal fin rounded, with 4th–7th rays longest
- Anal fin with 3 spines and 20–22 soft rays

Colour

- Silvery with faint, dark horizontal lines between scale rows
- Brown band from top of head through eye to chest (fading with growth); another less distinct curved, dark band from nape across operculum and pectoral fin base to belly
- Median fins dusky or pale with dark margins



Size

- Maximum standard length 103 mm

Distribution

- North coast of Australia and south coast of New Guinea

Reference: Carpenter and Niem (eds), 1998, Vol 6, p 3622

Threadfin Scat *Rhinoprenes pentanemus*

CAAB: 37 364001

Specimen: 113 mm SL

Diagnostic characters

- Dorsal fin single, with 8 spines and 19 or 20 soft rays, the first spine greatly elongated, extending past caudal-fin base
- Tip of first dorsal-fin pterygiophore exposed as a T-shaped “procumbent dorsal spine” at front of dorsal fin
- Anal fin with 3 spines and 16–18 soft rays
- Pectoral fins with 19 or 20 rays, the upper 4 rays unbranched, and the fourth greatly elongated, reaching past base of anal fin
- Pelvic fins with 1 spine and 5 soft rays, the first ray elongate, extending to caudal fin, the other 4 rays rudimentary, the fin inserted well forward on chest (below preopercle)
- Caudal fin wedge-shaped, with 8 branched rays in dorsal portion and 7 in ventral portion



Colour

- Pinkish grey, darker dorsally
- Scales and fins speckled with minute melanophores
- Fins, including first dorsal-fin spine, blackish

Size

- Maximum standard length 186 mm

Distribution

- Known only from north coast of Australia and the Gulf of Papua

Reference: Carpenter and Niem (eds), 1998, Vol 6, p 3621

BOMBAY DUCKS FAMILY: Harpadontidae

Glassy Bombay Duck *Harpadon translucens*

CAAB: 37 119001

Specimen: 155 mm SL



Diagnostic characters

- Lateral-line scales enlarged, extending as a median lobe of caudal fin
- Primary caudal-fin rays without scales
- Body compressed
- Pectoral fins reaching well short of origin of dorsal fin

- 13–15 dorsal fin rays
- Pectoral fins reaching to more than 1/2 distance from snout to origin of dorsal fin

Colour

- Body translucent pale green, densely speckled with black

Size

- Maximum standard length 294 mm

Distribution

- Northwestern Australia, southern New Guinea, and northeastern Australia

Reference: Carpenter and Niem (eds), 1998, Vol 3, p 1936

CARDINAL FISHES FAMILY: Apogonidae

Creamspotted Cardinalfish *Apogon albimaculosus*

CAAB: 37 327014

Specimen: 61 mm SL



Diagnostic characters

- Preopercular margins smooth to weakly serrate, suborbital and preopercular ridge crenate or smooth
- Caudal fin rounded

Colour

- Distinct large yellow or cream spots in rows along body
- First dorsal fin yellowish-brown; second dorsal, anal and caudal fins checked brown over yellow; soft dorsal and sometimes anal fin with large black yellow-rimmed ocellus basally

Size

- Maximum standard length 68 mm

Distribution

- Indo-West Pacific: northwestern Australia and Papua New Guinea

Reference: Sainsbury *et al.*, 1985, p 148; Froese and Pauly (eds), 2005

Pearlyfin Cardinalfish *Apogon poecilopterus*

CAAB: 37 327026

Specimen: 98 mm SL

Diagnostic characters

- Preopercular margin smooth except at angle where it is crenate
- Preopercular and suborbital ridges smooth
- Caudal fin truncate or rounded
- First dorsal fin with 7 spines, second dorsal fin with 1 spine and 9 rays
- 11–12 gill rakers

Colour

- Body crossed by 9–10 light brown vertical bands, dark scale edges forming longitudinal rows in larger specimens
- Outer half of spinous dorsal fin black



Size

- Maximum standard length 103 mm

Distribution

- Indo-Pacific: as far north as Japan, southward to the Arafura Sea and northern Australia and westward to Madras, India

Reference: Sainsbury *et al.*, 1985, p 152; Froese and Pauly (eds), 2005

Striped Cardinalfish *Apogon fasciatus*

CAAB: 37 327008

Specimen: 75 mm SL

Diagnostic characters

- First dorsal fin with 7 spines, second dorsal fin with 1 spine and 9 rays
- 15 pectoral fin rays
- 27–28 lateral line scales

Colour

- Tan or whitish with a broad, blackish, midlateral stripe that extends to end of caudal fin, and one or two narrow, dark stripes above; no black spot at caudal fin base



Size

- Maximum standard length 91 mm

Distribution

- East Africa and Red Sea to northern Australia and north to Japan

Reference: Randall *et al.*, 1990, p 141

CODLETS FAMILY: Bregmacerotidae

Unicorn Codlet *Bregmaceros maclellandi*

CAAB: 37 225002

Specimen: 76 mm SL



Diagnostic characters

- Head relatively short, 6 or more times in total length
- Anal-fin origin under second dorsal-fin origin
- Second dorsal fin with 57–65 soft rays
- Lateral line scales less than 78

Colour

- Back and upper part of sides light brown, middle and lower sides and belly pale to silvery
- Upper part of pectoral fins black or rather dark

- Dark pigment usually present on caudal fin, and sometimes anterior and posterior lobes of anal fin (pigments often faint or absent in juveniles)

Size

- Maximum standard length 77 mm

Distribution

- Coastal waters from the west coast of India to the Gulf of Thailand; probably widespread in Southeast Asia and Indonesia, but taxonomically confused and distribution records must be verified

Reference: Carpenter and Niem (eds), 1998, Vol.3, p.1998

CUSKEELS FAMILY: Ophidiidae

Golden Cusk *Sirembo imberbis*

CAAB: 37 228005

Specimen: 110 mm SL



Diagnostic character

- Bands of fine teeth in jaws, those in outer row scarcely enlarged, not canine-like

Colour

- Body and head cream or silver, pale olive or golden above with 2–3 gold or tan longitudinal bands broken into dashes and smaller spots running along the body

- Dorsal fin pale gold with anterior and median large dark blotches; dark brown band along white anal fin

Size

- Maximum standard length 195 mm

Distribution

- Indo-West Pacific: Japan to Western Australia and Queensland, including the Arafura Sea

Reference: Sainsbury *et al.*, 1985, p.82; Froese and Pauly (eds), 2005

EELTAIL CATFISHES FAMILY: Plotosidae

Nakedhead Catfish *Euristhmus nudiceps*

CAAB: 37 192003

Specimen: 245 mm TL



Diagnostic characters

- Gill membranes not joined, separated by broad isthmus
- Body depth more than 10 times in standard length
- Two rows of teeth on vomer

Colour

- Body dark brown

Size

- Maximum standard length 290 mm

Distribution

- Indo-West Pacific: northwestern Australia, the Arafura Sea and Papua New Guinea

Reference: Gloerfelt-Tarp and Kailola, 1984, p 69

FLATHEADS FAMILY: Platycephalidae

Brownmargin Flathead *Cociella hutchinsi*

CAAB: 37 296018

Specimen: 149 mm SL



Diagnostic characters

(see Fig. 13 in Glossary for head spine positions)

- Upper preopercular spine long, nearly reaching to rear margin of opercle
- Preorbital spine slight, often lacking
- Suprarorbital ridge usually smooth over anterior quarter of eye
- Suborbital ridge bearing 2 spines below the eye, and 3–4 behind the eye
- Iris lappet simple, semicircular
- Interopercular flap absent

Colour

- Body brownish to tannish above; white below with specks of brown sparse on breast, more heavily stippled posteriorly
- Spinous dorsal fin dusky, with a broad, dark brown band along its outer third; soft dorsal fin pale with small brown spots on rays; anal fin white; pectoral fins dusky brown, with traces of vertical dark bands, lower margin white; pelvic fins with whitish base, stippled with brown,

with well-developed submarginal dark band; caudal fin dusky, usually with a row of dark streaks forming a submarginal dark band

Size

- Maximum standard length 257 mm

Distribution

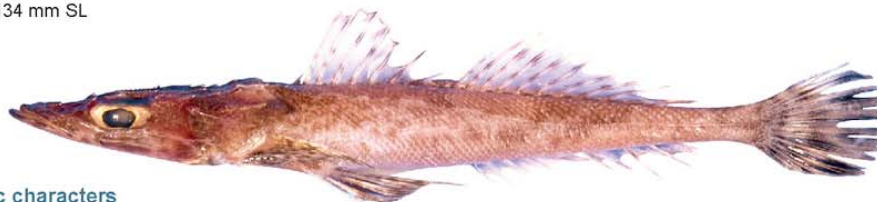
- Restricted to the Arafura and Timor seas off northern Australia

Reference: Carpenter and Niem (eds), 1998, Vol 4, p 2392

Harris' Flathead *Inegocia harrisii*

CAAB: 37 296010

Specimen: 134 mm SL



Diagnostic characters

(see Fig. 13 in Glossary for head spine positions)

- Upper preopercular spine short, sub-equal with next; accessory spine absent
- Suborbital ridge usually with 2 spines under eye and several more spines behind eye
- Interopercular flap absent
- 11 dorsal-fin rays
- 11 anal-fin rays
- 22–25 pectoral-fin rays (usually 23 or 24)

Colour

- Head and body grey or brown above, with small brown spots, whitish below with dark stippling, several vague dark bands crossing back
- Dorsal fins dusky, with small brown spots forming vertical bands; pelvic fins dusky, with a few dark spots; caudal fin dusky with prominent elongate dark blotches

Size

- Maximum standard length 198 mm

Distribution

- Northern Australia from northwestern shelf and Napier Broome Bay (Western Australia); Darwin (Northern Territory); Gulf of Carpentaria and Pine Peak (Queensland); the Arafura Sea; also Daru (Papua New Guinea)

Reference: Carpenter and Niem (eds), 1998, Vol 4, p 2401

FORKTAILED CATFISHES FAMILY: Ariidae

Shieldhead Catfish *Arius nella*

CAAB: 37 188013

Specimen: 142 mm SL



upper tooth patches



Diagnostic characters

- Head length 29–33% of standard length
- Mouth terminal, its width 30–39% of head length
- Granular posterior tooth patch on roof of mouth elongate, long axes slightly diverging distally (see Fig. above)
- Palate teeth stout and peg-like
- Total gill rakers on first gill arch 13–15, no rakers on hind aspect of first 2 gill arches

Colour

- Body reddish-brown to charcoal blue above, sides often scattered with small violet or olive-brown dots
- Pectoral fin dark bluish-brown above with remaining fins having dark brown margins

Size

- Maximum standard length 159 mm

Distribution

- Widespread from the east coast of India to Thailand, Singapore, Malaysia, Indonesia and the Philippines, to the southern coast of New Guinea and northern Australia from near Darwin (Northern Territory) to Townsville (Queensland)

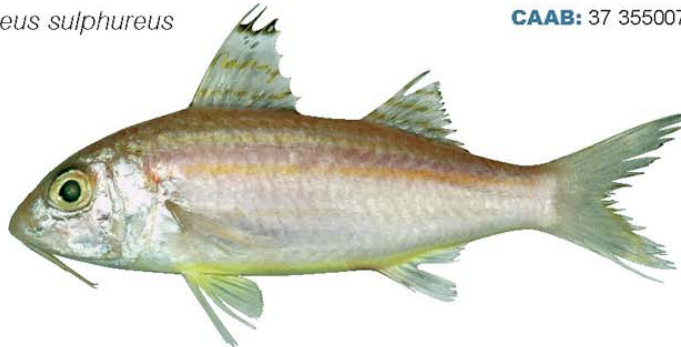
Reference: Carpenter and Niem (eds), 1998, Vol 3, p 1858

GOATFISHES FAMILY: Mullidae

Sunrise Goatfish *Upeneus sulphureus*

CAAB: 37 355007

Specimen: 110 mm SL



Diagnostic characters

- Chin with two slender white barbels that usually reach or extend past the rear margin of the preopercle, their length 1.25–1.7 times in head length (adults have relatively larger barbels than juveniles)
- Gill-rakers on first gill arch 26–30, with between 7 and 9 on the upper limb, and between 18 and 22 on the lower limb
- 14–17 pectoral-fin rays (rarely 14 or 17)
- 33–36 scales along lateral line (rarely 33 or 36)

Colour

- 2 narrow brassy yellow stripes on side of body
- Caudal fin without dark cross bands; first dorsal fin broadly tipped with black

Size

- Maximum standard length 133 mm

Distribution

- Southern Red Sea to southern Mozambique and Madagascar, east along continental shores to Indonesia and northern Australia, north to southern Japan. East in Oceania only to Fiji

Reference: Carpenter and Niem (eds), 1998, Vol 5, p 3197

GRUNTER BREAMS FAMILY: Haemulidae

Blotched Javelin *Pomadasys maculatus*

CAAB: 37 350002

Specimen: 101 mm SL



Diagnostic characters

- 2 pores and a median pit on chin
- Dorsal fin with 12 spines and 13–15 soft rays
- Anal fins with 3 spines and usually 7 soft rays
- 7–9 scales between lateral line and dorsal-fin origin
- 22–25 scales around caudal peduncle, 9 or 10 above lateral line and 11 or 12 below

Colour

- Silver grey with head brownish, nape and back with a series of incomplete cross bars (most obvious on nape)
- Spinous part of dorsal fin with a large black blotch; dorsal and caudal fins edged with black, other fins yellowish

Size

- Maximum standard length 140 mm

Distribution

- East coast of Africa, Madagascar, Red Sea, Gulf of Aden, Persian Gulf, Pakistan, India and Sri Lanka to northern half of Australia (from Shark Bay, Western Australia, to Moreton Bay, Queensland), New Guinea and Philippines to southern Japan

Reference: Carpenter and Niem (eds), 1998, Vol 5, p 2988

Bycatch Guide 25

GRUNTERS FAMILY: Terapontidae

Largescale Grunter *Terapon theraps*

Specimen: 83 mm SL

CAAB: 37 321003



Diagnostic characters

- Lower opercular spine very long and strong, extending well beyond margin of opercular lobe
- Post-temporal bone expanded, exposed and serrate posteriorly
- First gill arch with 6–8 gill rakers on upper limb and 14–17 on lower limb
- Second last dorsal-fin spine about half length of last spine
- Third anal-fin spine longest but shorter than the longest anal-fin ray
- Pored scales in lateral line 46–56

Colour

- 4 dark brown horizontal stripes on body
- Spinous part of dorsal fin with a black blotch dorsally on fin membranes between 3rd and 7th spines; both lobes on caudal fin with a dark transverse band

Size

- Maximum standard length 141 mm

Distribution

- Indo-West Pacific: East Africa, Madagascar, Seychelles, Red Sea, Arabian Peninsula, Persian Gulf to India and Andaman Islands; and southeast Asia. Reaches south to the Arafura Sea and northern Australia

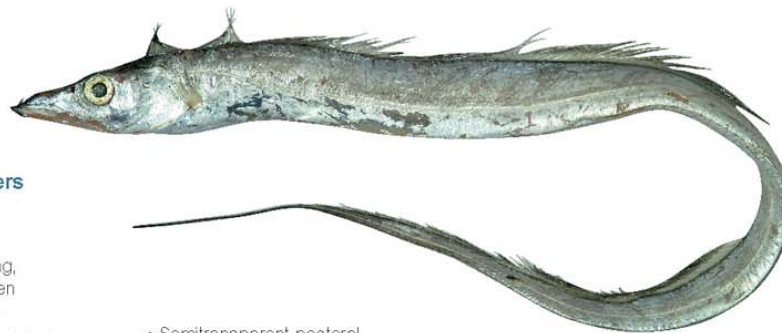
Reference: Carpenter and Niem (eds), 1998, Vol 5, p 3316

HAIRTAILS FAMILY: Trichiuridae

Largehead Hairtail *Trichiurus lepturus*

Specimen: 680 mm TL

CAAB: 37 440004



Diagnostic characters

- Lower hind-margin of gill cover concave
- Dorsal fin high and long, without a notch between spinous and soft parts
- Pectoral fins rather short, but extend beyond lateral line
- Pelvic fins absent

Colour

- Steel-blue body with a silvery reflection is noticeable in fresh specimens. After death, colour can become consistent silvery-grey

- Semitransparent pectoral fins and other fins can have a pale yellow tinge

Size

- Maximum total length 820 mm

Distribution

- Throughout tropical and temperate waters of the world

Reference: Carpenter and Niem (eds), 1998, Vol 6, p 3715

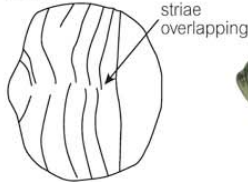
ILSHAS FAMILY: Pristigasteridae

Ditchelee *Pellona ditchela*

CAAB: 37 085009

Specimen: 125 mm SL

Scale



Diagnostic characters

- Upper jaw with a toothed hypomaxilla bone between posterior tip of premaxilla and lower bulge of maxilla blade
- Lower gill-raker count 22–27
- Vertical striae of scales slightly overlapping each other at centre of scales (see Fig. above)

Colour

- Head gold, snout and chin dusky; body dusky above; gold on flanks and silver below; faint humeral spot present
- Dorsal and caudal with dark stippling, other fins pale

Size

- Maximum standard length 138 mm

Distribution

- Tropical Indo-West Pacific from the western Gulf of Oman to Durban, South Africa; also Madagascar, Thailand, India to Indonesia, Papua New Guinea, the Philippines, and to north and southwestern Australia

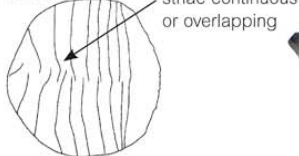
Reference: Carpenter and Niem (eds), 1998, Vol 3, p 1769

Longtail Ilisha *Ilisha lunula*

CAAB: 37 085012

Specimen: 170 mm SL

Scale



Diagnostic characters

- Body moderately slender, depth 32.8–39.2% standard length
- No toothed hypomaxilla between posterior tip of premaxilla and blade of maxilla; shaft of maxilla without distinct lobe or flange between posterior tip of premaxilla and blade of maxilla
- Frontal bones with 2 prominent ridges
- Lower gill-raker count 18–20
- Anal fin with 41–46 fin rays

- Pelvic fins present
- Vertical striae on scales continuous or overlapping across centre of scale (see Fig. above)
- Caudal fin deeply forked, caudal-fin lobes slender, tips attenuated such that expanded fin is strongly lunate

Colour

- Dorsal tip and margin of entire caudal fin dark brown or charcoal

Size

- Maximum standard length 172 mm

Distribution

- Indo-West Pacific, from Timor Sea near northwestern Australia, through Arafura Sea, off northern Australia, Gulf of Carpentaria, Gulf of Papua, and southward along Queensland coast at least to Fitzroy River mouth

Reference: Carpenter and Niem (eds), 1998, Vol 3, p 1761

JEWFISHES FAMILY: Sciaenidae

Orange Jewfish *Atrobuca brevis*

CAAB: 37 354012

Specimen: 166 mm SL

Diagnostic characters

- Has a moderately long caudal peduncle (22–27% of standard length)
- Mental pores in 3 pairs, the first small, at front of chin, separated by symphysis
- Dorsal fin with 29–32 soft rays
- Pectoral fins short, 20–24% of standard length
- Swimbladder is carrot shaped, with 27–30 pairs of branched appendages, each with a well-developed dorsal and ventral limb, regularly arranged so that terminal branches of dorsal limb point backwards and those of ventral limb downward (see Fig. on right)



Colour

- Brown or copper-coloured above, pale orange or white below

Size

- Maximum standard length 240 mm

Distribution

- Along north western and northern coast of Australia and southern Papua New Guinea



Swimbladder appendages

Reference: Carpenter and Niem (eds), 1998, Vol 5, p 3131

River Jewfish *Johnius borneensis*

CAAB: 37 354007

Specimen: 120 mm SL

Diagnostic characters

- Snout pointed, but not swollen or projecting
- Mouth large, oblique
- Upper jaw extending backward below posterior half of eye
- No barbel on chin
- Teeth in narrow bands, well differentiated into large and small in both jaws, but none canine-like
- Gill rakers slender, about 1/2 length of gill filaments at angle of arch, 9–15 on lower limb of first gill arch
- Anal fin with 2 spines, second spine moderately long, 24–42% of head length
- Scales on head cycloid (see Fig. 11); some scales present on soft parts of dorsal and anal fins



- Swimbladder hammer-shaped, with 14–17 pairs of arborescent appendages, the first pair entering head beyond transverse septum

Colour

- Head and body dusky mauve above, silvery-white below
- 2 dusky streaks usually present along mid-sides
- Fins pale yellow or cream, outer two-thirds of spinous dorsal fin black, sometimes a dusky streak along soft dorsal fin

Size

- Maximum standard length 172 mm

Distribution

- Indo-West Pacific west to Persian Gulf, east to southern China, Taiwan Province of China and northern and northeastern Australia and New Guinea

Reference: Carpenter and Niem (eds), 1998, Vol 5, p 3154

Objective 4: Bycatch descriptions of banana prawn fisheries

Smooth Jewfish *Johnius laevis*

CAAB: 37 354004

Specimen: 124 mm SL



Diagnostic characters

- No barbel on chin
- Teeth differentiated into large and small in upper jaw only; lower jaw with band of villiform teeth
- Gill rakers slender, about 1/2 length of gill filaments at angle of arch, 10–12 on lower limb of first gill arch
- Dorsal fin with 29–34 soft rays
- Anal fin with 2 spines, second spine moderately long, stiff, 29–52% of head length
- Scales large, those on flanks much larger than those on lateral line, 5 or 6 scale rows above lateral line to origin of dorsal fin, 8–10 scale rows below lateral line to origin of anal fin

- Scales cycloid on head and throat, weakly ctenoid on other parts of body, ctenii (small spines along posterior margin of scales) poorly developed, smooth to touch
- Small scales present on soft parts of dorsal and anal fins
- Swimbladder hammer-shaped with 11–14 pairs of arborescent appendages, the first pair entering head beyond transverse septum

Colour

- Head and upper body iridescent mauve or bronze, lower part of body silvery-white
- Fins yellow, black margins to dorsal and caudal fins

Size

- Maximum standard length 154 mm

Distribution

- Northern Australia and southern Papua New Guinea

Reference: Carpenter and Niem (eds), 1998, Vol 5, p 3147

Southern Yellow Jewfish *Larimichthys pamoides*

CAAB: 37 354026

Specimen: 141 mm SL



Diagnostic characters

- Mental pores in 3 pairs, the first open at front of chin, separated by symphysis
- Gill rakers long, slender, about 1/2 times longer than gill filaments at angle of arch, 13–17 on lower limb of first arch
- Dorsal fin with 9 spines, followed by a notch, second part of fin with 1 spine and 33–36 soft rays
- Anal fin with 2 spines, second spine short and slender, 23–33% of head length
- Pectoral fins long, 25–27% of standard length
- Scales on head and body cycloid except on belly and underside of caudal peduncle where it's weakly covered with ctenoid scales; small scales present on a basal quarter of soft parts of dorsal and anal fins

- Swimbladder carrot shaped, with about 26 pairs of arborescent appendages along its sides, each appendage with a well-developed dorsal and ventral limb, the first pair entering head beyond transverse septum

Colour

- Dorsal surface and side of body dusky (densely covered with melanophores); ventral surface paler with yellow pigmentation

- The chin, lower jaw, premaxilla and branchiostegal rays with yellow pigmentation
- Caudal, anal, first dorsal and a majority of second dorsal fin dusky; pectoral and pelvic fins yellow

Size

- Maximum standard length 147 mm

Distribution

- Along coasts of northwestern Australia and southern Papua New Guinea

Reference: Carpenter and Niem (eds), 1998, Vol 5, p 3158

Bycatch Guide 29

LANTERNFISHES FAMILY: Myctophidae

Opaline Lanternfish *Benthosema pterotum*

CAAB: 37 122079

Specimen: 31 mm SL



Diagnostic characters

- Prc2 much higher than Prc1, lying twice its own diameter or less below the lateral line (see top Fig. opposite)
- Small simple teeth on premaxillae and dentaries
- Primary photophore locations (see bottom Fig. opposite)

- All fins colourless except for the caudal fin where small black spots become more frequent posteriorly

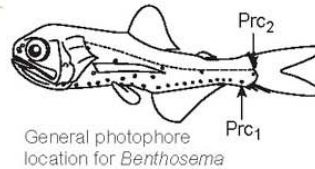
Size

- Maximum standard length 46 mm

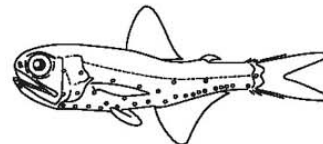
Distribution

- Tropical at least in the Indian and far Western Pacific Oceans

Reference: Carpenter and Niem (eds), 1998, Vol 3, p 1962



General photophore location for *Benthosema*



Photophore location for *Benthosema pterotum*

LEATHERJACKETS FAMILY: Monacanthidae

Threadfin Leatherjacket *Paramonocanthus filicauda*

CAAB: 37 465024

Specimen: 82 mm SL



Diagnostic characters

- Caudal peduncle unarmed or occasionally with a single spine
- Bony rudiment of ventral fin elongate, slender and moveable
- Caudal fin almost truncate, upper 2 rays often produced as a filament

Colour

- Snout and cheeks with yellow tints
- Charcoal blotch on back below anterior of soft dorsal fin
- Spinous dorsal fin fawn with 3–4 black spots along posterior of first spine; anal fin pale blue with 5–6 undulating yellow bands; caudal fin pale blue with 3–4 oblique series of black checks

Size

- Maximum standard length 105 mm

Distribution

- Northern half of Australia to southern New Guinea

Reference: Carpenter and Niem (eds), 1998, Vol 6, p 3937

LEFTEYE FLOUNDERS FAMILY: Bothidae

Waite's Flounder *Arnoglossus waitei*

CAAB: 37 460045

Specimen: 65 mm SL

Diagnostic characters

- Eyes separated by bony ridge
- Scales on eyed side of body with short ctenii or scales cycloid
- Teeth in both jaws small and closely spaced
- Body depth 2.2–2.3 times in standard length
- Dorsal fin rays 94–106; anal fin rays 76–82

Colour

- Body brown, mottled; dark spots on dorsal, anal and caudal fins, sometimes a dark bar or 2 dark blotches near base of caudal fin



Size

- Maximum standard length 78 mm

Distribution

- Known from Arafura Sea and southern Queensland

Reference: Carpenter and Niem (eds), 1998, Vol 6, p 3831

LIZARDFISHES FAMILY: Synodontidae

Shortfin Saury *Saurida argentea*

CAAB: 37 118005

Specimen: 183 mm SL

Diagnostic characters

- Lateral-line scales not enlarged, nor extending beyond base of caudal fin, procurrent and primary caudal-fin rays with scales
- Body cylindrical
- Lower jaw shorter than or equal to upper jaw, not visible from above when mouth is closed
- Longest ray of dorsal fin more than 3 times as long as last ray
- Scale in axil of pectoral fins long and pointed
- Fins (except first dorsal and caudal-fin rays in some species) without dark rays or spots

- Pectoral fins short, not reaching to or extending a little beyond base of pelvic fins, their length less than 20% of standard length
- Outer bands of palatine teeth in 3 or more rows anteriorly
- Pectoral fins not reaching base of pelvic fins
- Lateral-line scales 56–58
- Predorsal scales 20–21

Colour

- Upper half of inside of pectoral fins dusky

- 9 or 10 faint blotches along lateral line, sometimes traces of very indistinct cross bars on back

Size

- Maximum standard length 300 mm

Distribution

- Widespread in the Western Pacific from the Gulf of Thailand to northeastern Australia

Reference: Carpenter and Niem (eds), 1998, Vol 3, p 1937



MULLETS FAMILY: Mugilidae

Longfinned Mullet *Valamugil perusii*

CAAB: 37 381023

Specimen: 130 mm SL



Diagnostic characters

- Snout short, 17–21% of head length
- Teeth on both lips minute and villiform or absent
- Maxilla slender and weakly curved down at posterior tip, which is only just posteroventral to corner of mouth and partially or completely concealed
- Serrate anteroventral edge of preorbital very weakly concave, not kinked; posteroventral tip narrow but not pointed
- Adipose eyelid reasonably well developed, covering most of iris posteriorly and partly anteriorly
- Origin of first dorsal fin midway between tip of snout and base of caudal fin, or slightly nearer latter
- Origin of fully erected second dorsal spine on vertical through about 3rd soft ray of anal fin; both fins moderately to well scaled on all parts
- Pectoral fin just reaching level of origin of first dorsal fin or extending beyond this
- Scales with a membranous, digitated hind margin; thoracic and abdominal scales more distinctly ctenoid; 31–34 in longitudinal series, 10 1/2–11 1/2 in transverse series; 16 scales in transverse series around caudal peduncle; predorsal scales to level of posterior nostril

Colour

- Pectoral fins with dark spot dorsally at origin

Size

- Maximum standard length 156 mm

Distribution

- Indo-West Pacific from Africa to Marianas Islands

Reference: Carpenter and Niem (eds), 1998, Vol 4, p 2106

OCEAN BASSES FAMILY: Acropomatidae

Sharptooth Seabass *Synagrops philippinensis*

CAAB: 37 311028

Specimen: 63 mm SL



Diagnostic characters

- Pelvic spine with distinct serration on the anterior edge
- 2 spines and 7 soft rays in anal fin
- 16 rays in pectoral fin

Colour

- Body blackish-brown dorsally, paler ventrally

Size

- Maximum standard length 81 mm

Distribution

- Indo-West Pacific: India east to Japan and south to the Philippines, the Arafura Sea and northern Australia

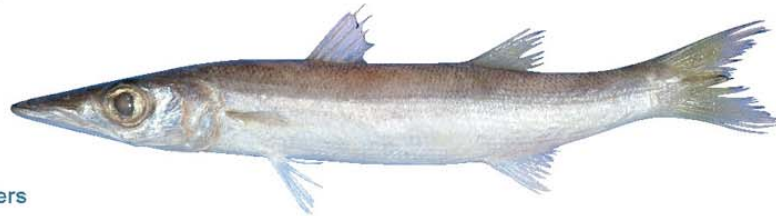
Reference: Sainsbury *et al.*, 1985, p 136

PIKES FAMILY: Sphyraenidae

Striped Barracuda *Sphyraena obtusata*

CAAB: 37 382001

Specimen: 178 mm SL



Diagnostic characters

- First gill arch with 2 prominent gill rakers
- Tip of pectoral fins reaching past origin of first dorsal fin

Colour

- Body without bars, stripes, and blotches, but a faint broad yellowish midlateral band on body (in very fresh specimens)
- Caudal fin yellowish

Size

- Maximum standard length 225 mm

Distribution

- Indo-Pacific: Red Sea and East Africa to Samoa north to the Ryukyu Islands, south to Lord Howe Island; Kapingamarangi and Marianas in Micronesia. Migrated to eastern Mediterranean from the Red Sea via the Suez Canal

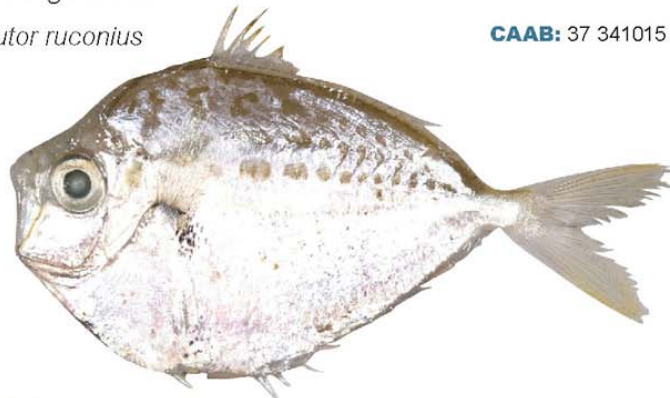
Reference: Carpenter and Niem (eds), 1998, Vol 6, p 3695

PONYFISHES FAMILY: Leiognathidae

Snubnose Ponyfish *Secutor ruconius*

CAAB: 37 341015

Specimen: 42 mm SL



Diagnostic characters

- Body oval, compressed and very deep, its depth between 1.4–1.7 times in standard length
- Upward pointing mouth when protracted
- Scales on body are relatively large with 10–16 rows between pectoral and pelvic fin bases; cheek has scales; totally scaly on breast including isthmus
- Lateral line terminates below middle of soft dorsal fin, with a count of 28–32 tubed scales at this point. Total count of scales to caudal peduncle 54–60

Colour

- Silvery body with around 10 bluish vertical stripes on back extending just under lateral line

Size

- Maximum standard length 76 mm

Distribution

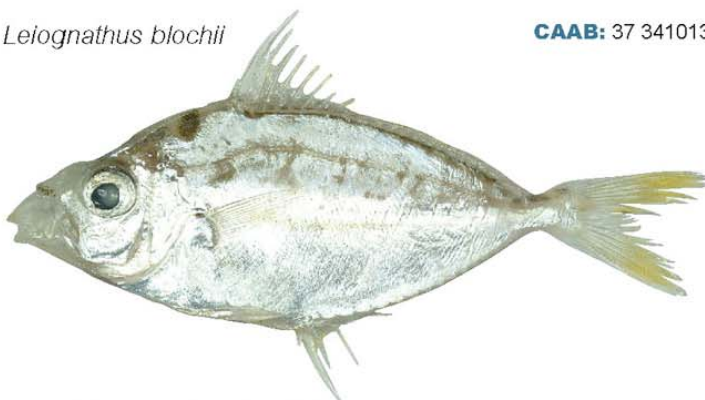
- Northern Australia to Taiwan Province of China, southward to Philippines, Indonesia, Thailand, coasts of Malaysia to India, the Red Sea, oceanic islands of Seychelles, Reunion, Mauritius and to East Africa

Reference: Carpenter and Niem (eds), 1998, Vol 5, p 2822

Twoblotch Ponyfish *Leiognathus blochii*

CAAB: 37 341013

Specimen: 54 mm SL



Diagnostic characters

- Body depth between 2.3 to 3.2 times in standard length
- Conspicuous scales present on breast

Colour

- Belly silvery
- Back is light brown with dark, irregular vertical lines extending down to midline
- A brown blotch on nape that becomes diffuse on preservation in formalin
- Tip of snout, head and ventral half of body with fine black dots

- Spinous part of the dorsal fin is traversed by a yellow streak at midheight, with the apex of the fin hyaline; soft part of dorsal and anal fins, as well as caudal fin, yellow with grey edges; pelvic and pectoral fins colourless, underside of pectoral-fin base dotted black

Size

- Maximum standard length 61 mm

Distribution

- Northern Australia from Broome to Cape York

Reference: Carpenter and Niem (eds), 1998, Vol 5, p 2803

PORCUPINEFISHES FAMILY: Diodontidae

Plain Porcupinefish *Cyclichthys hardenbergi*

CAAB: 37 469008

Specimen: 81 mm SL



Diagnostic characters

- Spines on top of head and on belly fixed and in an erect position
- No spines wholly on caudal peduncle
- Few black spots on body, those present at base of spines dorsally and dorsolaterally
- Dorsal, anal and caudal fins with dusky distal margins
- 2 spines over eye, 2 spines between nostrils – 1 immediately adjacent to each nostril

Colour

- Body and head olive brown, with white below

Size

- Maximum standard length 113 mm

Distribution

- Tropical Australia west of Cape York and southern New Guinea

Reference: Carpenter and Niem (eds), 1998, Vol 6, p 3962

ROCKCODS FAMILY: Serranidae

Sixbar Grouper *Epinephelus sexfasciatus*

CAAB: 37 311017

Specimen: 116 mm SL



Diagnostic characters

- Body depth 2.7–3.2 times in standard length
- Caudal peduncle depth 2.6–3.4 times in head length
- Preopercle with 2–4 greatly enlarged serrae at the angle; upper edge of operculum straight
- First gill arch with 20–23 gill rakers, of which 7 or 8 are on upper limb and 13–15 on lower limb

Colour

- 5 dark brown bars on body and 1 on nape (dark bars usually more or less divided by a narrow pale bar)

- Scattered pale spots may be present on body, and some faint small brown spots are often visible on the edges of the dark bars
- Soft dorsal, caudal, and pelvic fins dusky grey, the pectoral fins greyish or dusky orange-red
- Jaws and ventral parts of head sometimes pale reddish brown

Size

- Maximum standard length 181 mm

Distribution

- Tropical western Pacific Ocean: Indonesia, Singapore, Malaysia, Thailand, Vietnam, Philippines, Papua New Guinea, Louisiade Archipelago, and Australia (north coast from Western Australia to Queensland)

Reference: Carpenter and Niem (eds), 1998, Vol 4, p 2522

SAND FLOUNDERS FAMILY: Paralichthyidae

Large-tooth flounder *Pseudorhombus arsius*

CAAB: 37 460009

Specimen: 192 mm SL



Diagnostic characters

- Some pairs of moderately large canines in anterior part of both jaws, 6–13 lateral teeth in lower jaw, stronger and more widely spaced than those of upper jaw
- Gill rakers pointed, longer than broad

Colour

- Body greenish or pale brownish; a dark blotch at junction of straight and curved parts of lateral line, and a smaller blotch half-way to caudal-fin base

Size

- Maximum standard length 203 mm

Distribution

- Widespread from the east coast of Africa to Australia, and tropical and subtropical areas of the western Central Pacific

Reference: Carpenter and Niem (eds), 1998, Vol 6, p 3847

SEAROBINS FAMILY: Triglidae

Smooth Gurnard *Lepidotrigla russelli*

CAAB: 37 288016

Specimen: 95 mm SL



Diagnostic characters

- Rostral processes curved with many short, rounded spines
- Pectoral fin reaching to below 4th or 5th dorsal-fin ray

Colour

- Most of inner pectoral fin uniform smokey dark green; base of lower rays cream; leading edge pink; diffuse patches of blue on lower portions of fin and over membrane between first and second rays

Size

- Maximum standard length 142 mm

Distribution

- Northern Australia from Barrow Island (Western Australia) to the Gulf of Carpentaria (Queensland)

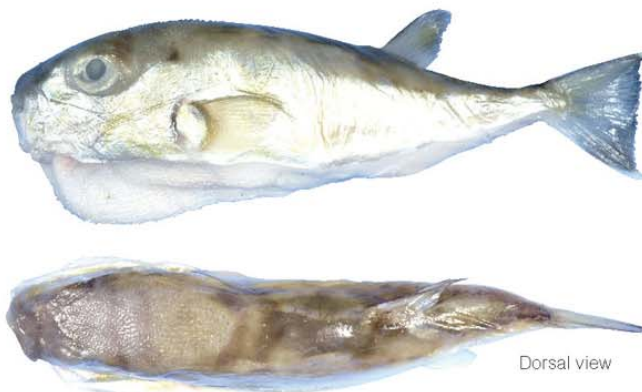
Reference: Sainsbury *et al.*, 1985, p102 (as *Lepidotrigla* sp. 1)

TOADFISHES FAMILY: Tetraodontidae

Brownback Toadfish *Lagocephalus spadiceus*

CAAB: 37 467017

Specimen: 92 mm SL



Diagnostic characters

- Body naked except for a patch of spines on back (between snout and half way to dorsal fin origin)
- Teeth in jaws modified to form a powerful beak of 4 teeth, 2 above and 2 below
- Caudal fin slightly lunate

Colour

- Head and body dusky green above with 3 broad dark cross-bands; flanks silver-yellow, belly white
- Pectoral fin yellow; anal fin white; dorsal and caudal fins dusky with white tips

Size

- Maximum standard length 140 mm

Distribution

- Indo-West Pacific: Red Sea, South Africa and Australia

Reference: Sainsbury *et al.*, 1985, p 320

TONGUE SOLES FAMILY: Cynoglossidae

Ogilby's Tongue Sole *Cynoglossus ogilbyi*

CAAB: 37 463017

Specimen: 125 mm SL



Diagnostic character

- Scales ctenoid on eyed side of body, cycloid on blind side

Colour

- No obviously pigmented spots on eyed side of body; eyed side of body tan
- Blind side pale
- Fin rays are dusky yellow at the base and become whiter towards the tip, less so towards caudal fin

Size

- Maximum standard length 180 mm

Distribution

- Western Central Pacific: southern Queensland, Australia, and Papua New Guinea

Reference: Froese and Pauly (eds), 2005

Pinocchio Tongue Sole *Paraplagusia longirostris*

CAAB: 37 463002

Specimen: 210 mm SL



Diagnostic characters

- Lips on eyed side of head distinctly fringed with labial papillae; labial papillae unbranched
- Caudal vertebrae 44 or more
- Two lateral lines on eyed side of body
- Dorsal fin rays 133–143; anal fin rays 102–112

Colour

- No obviously pigmented spots on eyed side of body
- Eyed side of body tan

Size

- Maximum standard length 278 mm

Distribution

- Indo-West Pacific: northern Australia and southern coast of Papua New Guinea

Reference: Carpenter and Niem (eds), 1998, Vol 6, p 3894

THREADFIN BREAMS FAMILY: Nemipteridae

Ornate Threadfin Bream *Nemipterus hexodon*

CAAB: 37 347014

Specimen: 163 mm SL

Diagnostic characters

- Lower margin of eye is located above a line from the snout tip to upper base of pectoral fins
- Long pectoral fins between 1 and 1.4 times length of head, extending to or beyond level of anus
- Long pelvic fins also reach to, or just beyond, level of anus
- Forked caudal fin with a slightly shorter lower lobe than the upper lobe

Colour

- Upper part of body pinkish, paling to silvery white on ventral surface
- 6–8 pale yellow stripes below the lateral line
- Blood red, ovoid spot below origin of the lateral line, bordered below by bright yellow



- Yellow stripe on either side of ventral midline, from isthmus to lower caudal-fin base
- Dorsal fins translucent whitish, with yellow margin; a narrow yellow stripe beginning anteriorly near base of fin and extending backwards to just above midposterior margin, this stripe bordered on either side by a translucent interspace that is edged by a narrow pale bluish white stripe; caudal fin pinkish with an upper lobe that is tipped with yellow; anal fin translucent; a narrow yellow

stripe running from near anterior base of anal fin to midposterior margin, the base of the fin beneath this stripe is pale bluish white

Size

- Maximum standard length 183 mm

Distribution

- Indo-West Pacific from the Andaman Sea to the Solomon Islands including northern Australia

Reference: Carpenter and Niem (eds), 1998, Vol 5, p 3066

THREADFIN SALMONS FAMILY: Polynemidae

Blackfin Threadfin *Polydactylus nigripinnis*

CAAB: 37 383001

Specimen: 111 mm SL

Diagnostic characters

- Posterior margin of preopercle with less than 10 serrations
- Gill rakers on first gill arch 24–27
- 6 pectoral filaments with the longest extending to the origin of the anal fin

Colour

- Body dusky dorsally, silvery-golden below; spot immediately above origin of pectoral fin
- Unpaired fins yellow with melanophores; pectoral fins, except uppermost ray and free filaments, intensely black on lateral surface



Size

- Maximum standard length 178 mm

Distribution

- Gulf of Papua, specifically Orokolo Bay; Cambridge Gulf (Western Australia) to Van Diemen Gulf (Northern Territory)

Reference: Carpenter and Niem (eds), 1998, Vol 5, p 3109

TREVALLAS FAMILY: Centrolophidae

Blackspot Butterfish *Psenopsis humerosa*

CAAB: 37 445007

Specimen: 110 mm SL



Diagnostic characters

- Dorsal-fin spines increase in length from front to back
- The last spine and 1st ray of dorsal fin of the same length; no impression of 2 separate fins
- More than 25 dorsal-fin rays

Colour

- Silvery-grey to charcoal above
- A large diffuse spot on shoulder
- Operculum and inside of mouth stippled charcoal

Size

- Maximum standard length 165 mm

Distribution

- Occurs in oceanic waters and well offshore in eastern Australian waters. Northern Australia to New Guinea and eastern Indonesia

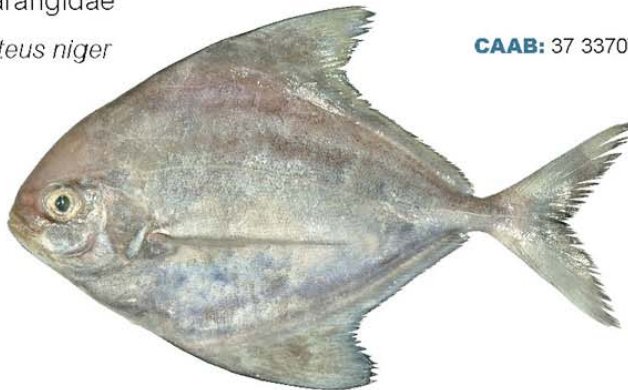
Ref: Carpenter and Niem (eds), 1998, Vol 6, p.3770

TREVALLIES FAMILY: Carangidae

Black Pomfret *Parastromateus niger*

CAAB: 37 337072

Specimen: 145 mm SL



Diagnostic characters

- Dorsal fin with 5 or 6 short spines (embedded and not apparent in adults) followed by 1 spine and 41–44 soft rays
- Profile of second dorsal and anal fins nearly identical, with elevated, broadly rounded anterior lobes; pelvic fins absent in specimens larger than about 10 cm fork length, and, in juveniles, positioned distinctly anterior to a vertical line through pectoral-fin base
- Scales small and deciduous, and almost completely covering dorsal and anal fins

Colour

- In life, adults uniformly silvery grey to bluish brown (yellowish brown when deciduous scales missing)
- Juveniles with dark, vertical bars and long, black, jugular pelvic fins

Size

- Maximum standard length 187 mm

Distribution

- Pelagic on the continental shelf from South Africa, Mozambique, Kenya, the Arabian Sea, Bay of Bengal, Indonesia, the Philippines, China, southern Japan, and Australia

Reference: Carpenter and Niem (eds), 1998, Vol 4, p.2729

Finny Scad *Megalaspis cordyla*

CAAB: 37 337028

Specimen: 120 mm SL



Diagnostic characters

- Caudal peduncle strongly compressed with a marked medial keel
- Second dorsal fin with the posterior 7–9 rays consisting of detached finlets
- Anal fin with posterior 8–10 rays consisting of detached finlets
- Lateral line strongly arched anteriorly, with junction of curved and straight parts below vertical from 4th or 5th spine of dorsal fin

Colour

- Head and body bluish grey to green dorsally, sides and belly silvery
- Large black opercular spot
- Dorsal and anal fins pale to yellow, distally dusky; pectoral and pelvic fins pale, with upper half dusky; caudal fin dark, especially leading and trailing edges of fin

Size

- Maximum standard length 160 mm

Distribution

- Broadly distributed throughout the Indian Ocean; elsewhere in the Indo-West Pacific from Japan to Australia and eastward to Fiji

Reference: Carpenter and Niem (eds), 1998, Vol 4, p 2726

Malabar Trevally *Carangoides malabaricus*

CAAB: 37 337005

Specimen: 122 mm SL

Diagnostic characters

- Gill rakers (including rudiments) on first gill arch 32–38
- Breast naked ventrally to distinctly behind pelvic fins, often to origin of second anal fin; laterally, naked area of breast extends diagonally to naked base of pectoral fins, including small area anteriorly just above pectoral-fin base

Colour

- In life, generally silvery with bluish grey above, silvery white below; opercle with a small black spot on upper margin; tongue greyish brown to brown; caudal fin, soft dorsal and anal fins pale greenish-yellow to dusky; interradial membranes of soft anal-fin rays often with a white spot basally

Size

- Maximum standard length 146 mm



Distribution

- In the western Indian Ocean broadly distributed in coastal waters, from south Africa to Sri Lanka, including the west coast of the Malagasy Republic; elsewhere in the Indo-West Pacific known from Straits of Malacca, Gulf of Thailand, Okinawa, Japan (rare), Indonesia, and Australia

Reference: Carpenter and Niem (eds), 1998, Vol 4, p 2704

TRIPODFISHES FAMILY: Triacanthidae

Blacktip Tripodfish *Trixiphichthys weberi*

CAAB: 37 464001

Specimen: 113 mm SL



Diagnostic characters

- Scale-covered ventral surface of pelvis almost as wide anteriorly as posteriorly, not distinctly tapered to a point
- Postorbital distance (eye to upper end of gill opening) short (5.2–6.8% of standard length)
- Snout long (21.2–27.7% of standard length)

Colour

- Basal third of first dorsal-fin spine much paler than distal portion

Size

- Maximum standard length 170 mm

Distribution

- Tropical western Pacific from the Philippines through Indonesia to northwestern Australia and on both sides of the Bay of Bengal in the Indian Ocean

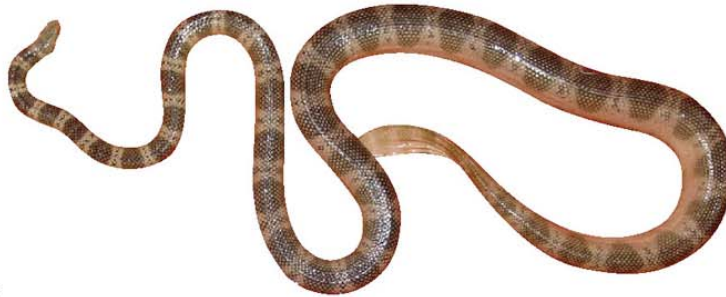
Reference: Carpenter and Niem (eds), 1998, Vol 6, p 3910

SEA SNAKES FAMILY: Hydrophiidae

Elegant Sea Snake *Hydrophis elegans*

CAAB: 39 125021

Specimen: 1210 mm SV



Diagnostic characters

- Head and body shape have some variation, but generally it's a long, thin sea snake with a relatively small head

Colour

- Juveniles: pale brown with black head and strongly banded body, the 35–55 bands widest on the mid-dorsal and mid-ventral lines, narrowest laterally where all or some of the bands may be broken; usually at least some indication of secondary spots or narrow bands in the paler interspaces between the primary bands

- Adults: bands reduced to a series of transverse vertebral bars or blotches and two series of spots on the flanks; a mid-lateral series on which each spot lies below a pale interspace, and a lower lateral series in which each spot lies below a dark dorsal bar. In addition there may be narrow secondary cross-bars or rows of dark scales. Ventral scales usually blackish on the throat and front part of the body

Size

- Maximum snout-vent length 1528 mm

Distribution

- New South Wales around New Guinea to Western Australia. It is the most common species caught in prawn trawls in the NPF

Reference: Cogger, 1992, p 716

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Olive-headed Sea Snake *Disteira major*

Specimen: 930 mm SV

CAAB: 39 125011



Diagnostic characters

- Large head and neck with a stout body

Colour

- Head olive or brown above with darker flecks
- Body pale grey above, with a series of 25–30 dark dorsal blotches or cross-bars (the first on the nape), each slightly narrower than the paler interspaces, extending about halfway down each side on to the cream or yellowish lower half of the body. A narrow, dark bar, usually no more than one scale in width, in the middle of each pale dorsal interspace.

A series of large, dark, lateral blotches, each below the narrow interspace bar, and a series of lower lateral blotches, each tending to lie below a large dorsal blotch. Most ventral scales dark grey, or tipped darker. Juveniles have a more conspicuous pattern than adults

Size

- Maximum snout–vent length 930 mm

Distribution

- Coastal waters of northern Australia from northwestern Western Australia and the Arafura Sea to eastern Queensland. Also found in southern New Guinean waters. Probably the second most common species caught in prawn trawls in the NPF. Appears to prefer more open, muddy habitats

Reference: Cogger, 1992, p 707

SWIMMING CRABS FAMILY: Portunidae

Cornflake Crab *Charybdis callianassa*

Specimen: 36 mm CW

CAAB: 28 911037



Diagnostic characters

- 6 anterolateral teeth of which 5 are large
- No distinct cardiac ridges on carapace (see Fig. 18)
- First anterolateral tooth truncate
- Anterior border of arm of cheliped with 2 spines
- Hand of cheliped swollen

Size

- Maximum carapace width 48 mm

Distribution

- Northern Territory (Gulf of Carpentaria, north coast), Western Australia (central west coast, northwest coast); Indo-west Pacific Oceans (Pakistan to Japan and Philippines)

Reference: Stephenson, 1972; Davie, 2002

Blunt-toothed Crab *Charybdis truncata*

CAAB: 28 911015

Specimen: 29 mm CW



Diagnostic characters

- 6 or 7 teeth on each anterolateral margin
- Posterior border of carapace forming an angular junction with posterolateral border
- Merus of cheliped without distal spine on posterior border

Size

- Maximum carapace width 45 mm

Distribution

- Northern Territory (north coast), Queensland (central east coast), Western Australia (central west coast, northwest coast); Indo-west Pacific Oceans (Madagascar to Japan and Philippines)

Reference: Stephenson, 1972; Davie, 2002

Swimming Crab *Portunus hastatoides*

CAAB: 28 911030

Specimen: 44 mm CW



Diagnostic characters

- Hand is long and slender, less massive than arm
- Posterior-postlateral junction of carapace has a conspicuous spine
- Frontal margin has 4 acute but somewhat rounded lobes, of which the outer are much larger than the inner
- Posterodistal end of the arm has 2 distinctive spines
- Posterodistal border of merus on the 4th walking leg is serrated
- Anteroexternal angle of merus of 3rd maxilliped strongly produced laterally

Size

- Maximum carapace width 51 mm

Distribution

- Northern Territory (north coast), Queensland (central east coast, northeast coast), Western Australia (central west coast, northwestern coast); Indo-west

Pacific Oceans (South Africa to Japan, Philippines south to Chesterfield Reefs, Coral Sea)

Reference: Stephenson and Campbell, 1959; Davie 2002

Swimming Crab *Portunus gracillimanus*

CAAB: 28 911027

Specimen: 41 mm CW



Diagnostic characters

- Hand of cheliped extremely slender, much less massive than arm (merus)
- 4-6 spines on anterior border of arm
- Mesogastric region of carapace with transverse granulated ridges
- Lateral frontal teeth almost the same width as medians

Size

- Maximum carapace width 49 mm

Distribution

- Northern Territory (north coast), Queensland (central east coast, northeastern coast); Indo-West Pacific Oceans (India to Hong Kong, Philippines, and New Guinea)

Reference: Stephenson, 1972; Davie, 2002

Three-spotted Crab *Portunus sanguinolentus*

CAAB: 28 911006

Specimen: 112 mm CW



Diagnostic character

- Posterior of carapace with 3 large red spots

Size

- Maximum carapace width 161 mm

Distribution

- East Africa to Japan and Australia. Within Australia from Exmouth Gulf, Geraldton (Western Australia); from Kennedy Sound (North Queensland) south to Botany Bay (New South Wales)

Reference: Stephenson, 1972; Davie, 2002

PRAWNS FAMILY: Penaeidae

Orange Prawn *Atypopenaeus formosus*

CAAB: 28 711003

Specimen: 28 mm CL



Diagnostic characters

- 4th and 5th segments of the abdomen with distinct dorsal ridges
- Rostrum long and slender, up-curved with widely spaced teeth

Colour

- Body generally bright pink to orange-pink

Size

- Females to 100 mm total length, males to 80 mm

Distribution

- Indo-West Pacific. Australia: northern coastline from Darwin (Northern Territory) through Queensland to northern New South Wales, including the Gulf of Carpentaria

Reference: Grey *et al.*, 1983, p 106

Northern Velvet Prawn *Metapenaeopsis novaeguineae*

CAAB: 28 711016

Specimen: 24 mm CL



Diagnostic characters

- Saw tooth ridge (stridulating organ) present on posterior sides of carapace (see Fig. 20)
- Rostrum slightly upcurved
- Dorsal ridge of third segment of abdomen flat or convex on top
- Body with a velvet texture

Colour

- Irregular mottling that is a dark reddish brown; colour between mottling is mild brown above fading to light brown below

Size

- Females to 100 mm total length, males to 70 mm

Distribution

- New Guinea and northern Australia. Australia: between Moreton Bay (Queensland) and Exmouth Gulf (Western Australia)

Reference: Grey *et al.*, 1983, p 70

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Southern Velvet Prawn *Metapenaeopsis palmensis*

CAAB: 28 711017

Specimen: 21 mm CL



Diagnostic characters

- Body with velvet texture
- Groove along dorsal ridge of third abdominal segment wide and shallow

Colour

- Translucent to pinkish with irregular dark pink or occasionally dark red mottling

Size

- Females to 105 mm total length, males to 85 mm

Distribution

- Indo-West Pacific. Found in tropical and warm temperate Australian waters (Shark Bay, Western Australia, and west to Sydney, New South Wales)

Reference: Grey et al, 1983, p 72

Coral Prawn *Solenocera australiana*

CAAB: 28 714011

Specimen: 29 mm CL



Diagnostic characters

- Antennular flagella flattened with lower one distinctly channeled
- Postrostral ridge distinct but not markedly raised

Colour

- Carapace and abdomen is pinkish-red to orange
- Telson, uropods and pleopods are dark red
- Antennular flagella are bright red

Size

- Females to 120 mm total length, males to 90 mm

Distribution

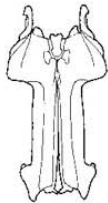
- Northern Australia

Reference: Grey et al, 1983, p 44

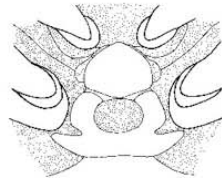
Northern Rough Prawn *Trachypenaeus anchoralis*

CAAB: 28 711054

Specimen: 24 mm CL



Petasma



Thelycum

note: shaded region depressed



Diagnostic characters

- Deep upward curving rostrum
- Hepatic crest present (see Fig. 20)
- Distinctive petasma and thelycum (Figs. above)
- Pleopods, telson and uropods pinkish-red, the later with yellow margins
- Legs yellowish-brown

Colour

- Body and abdomen uniform pale yellow-brown

Size

- Females to 100 mm total length, males to 70 mm

Distribution

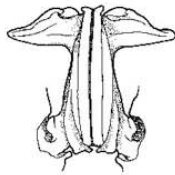
- Restricted to northern Australia between Shark Bay (Western Australia) and Keppel Bay (Queensland)

Reference: Grey *et al.*, 1983, p 122

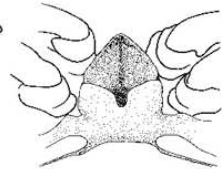
Southern Rough Prawn *Trachypenaeus curvirostris*

CAAB: 28 711055

Specimen: 27 mm CL



Petasma



Thelycum

note: shaded region depressed



Diagnostic characters

- Body colouration
- Presence of hepatic groove
- Distinctive petasma and thelycum (Figs. above)

Colour

- Body pink to reddish-brown with whitish legs; red pleopods and uropods

Size

- Females to 100 mm total length, males to 80 mm

Distribution

- Indo-West Pacific: from Japan through China, Malaysia, India, East Africa, Madagascar; Red Sea through Suez to the Mediterranean to Egypt, Israel and

Turkey; Indonesia, New Guinea; Australia: from Shark Bay (Western Australia) through northern waters to Sydney (New South Wales)

Reference: Grey *et al.*, 1983, p 124

Objective 4: Bycatch descriptions of banana prawn fisheries

Rough Prawn *Trachypenaeus gonospinifer*

CAAB: 28 711057

Specimen: 14 mm CL



Diagnostic characters

- Rostrum slender and curving downwards
- Body relatively slender with robust cuticle

Colour

- Body translucent, faintly pink
- Appendages, telson, uropods and rostrum transparent

Size

- Females to 80 mm total length, males to 50 mm

Distribution

- Northern Australian to southern New Guinean waters

Reference: Grey *et al.*, 1983, p 116

MANTIS SHRIMP FAMILY: Squillidae

Dictyosquilla tuberculata

CAAB: 28 051030

Specimen: 19 mm CL, 93 mm TL



Diagnostic characters

- Mid-dorsal surface of carapace and thorax, and median surface of abdomen covered with irregular reticulated carinae
- Raptorial claw dactylus with 6 teeth, its proximal margin is not swollen
- 5th thoracic somite lateral process anterior lobe is a short, narrow spine directed forward; posterior lobe is short, usually triangular and flattened, apex usually blunt, directed laterally
- 8th thoracic somite lateral process is subquadrate with a blunt apex

- 1st–5th abdominal somites with lateral margins below intermediate carinae distinctly tuberculate
- Dorsolateral carinae of telson slender, approximately half the width of the intervening space between adjacent carinae

Colour

- Tan dorsally with a light brown longitudinal stripe down length of body

- Distal 2/3 of uropod brownish
- Carinae on telson brownish

Size

- Maximum carapace length 22 mm

Distribution

- From northeast Queensland coast (Townsville), in the Gulf of Carpentaria and across to Joseph Bonaparte Gulf (Western Australia)

Reference: Ahyong, 2001, p 244

Objective 4: Bycatch descriptions of banana prawn fisheries

Oratosquilla graveri

Specimen: 18 mm CL, 97 mm TL

CAAB: 28 051050



Diagnostic characters

- Dorsal surface is coarsely and irregularly pitted and rugose
- 1st abdominal somite dorsal processes apices angular to sharp
- Rostral plate broader than, to as long as, broad, usually appearing elongate; without dorsal carina
- Carapace with branches of anterior bifurcation of median carina distinct
- Raptorial claw dactylus with 6 teeth; carpus dorsal carina undivided
- 6th thoracic somite lateral process anterior lobe is broad, trianguloid, apex rounded to angular, blunt; posterior lobe is broad and triangular with its anterior margin straight

Colour

- Light grey brown dorsally
- Carinae and grooves of carapace, submedian carinae and posterior margins of body somites dark red
- Telson with median carina and carinae of primary teeth red; median carina with a dark red-maroon spot proximally and a narrow dark patch on posterior third
- Uropodal protopod with red terminal spines
- Endopod with black on distal half

- Exopod with pink movable spines
- Exopod distal segment yellowish with inner proximal third to half dark, diffuse distally

Size

- Maximum carapace length 23 mm

Distribution

- Vietnam to the Philippines, New Caledonia and Australia. Australia: southern Queensland to the Gulf of Carpentaria and Arafura Sea

Reference: Ahyong, 2001, p 288

SQUIDS FAMILY: Enoploteuthidae

Firefly Squid *Abralia armata*

Specimen: 38 mm ML

CAAB: 23 621009



Diagnostic characters

- Mantle is short and conical, the anterior half with nearly parallel sides, the posterior half is cone-shaped
- Fins are short and wide, their anterior margins slightly convex, the posterior margins are concave and roundly pointed laterally
- Arms 1, 2 and 4 are not keeled, 3 is keeled for most of its length
- Most of each arm has hooks in two rows, 12–16 on 1, 12–16 on 2, 14–16 on 3 and 15–20 on 4. Beyond the hooks are 6–12 moderately large suckers, in 2 rows, with 4–6 teeth on the upper border and entirely on the bottom; beyond the larger suckers are two rows of minute suckers reaching to the distal extremity
- Tentacles are long and oval in cross section

- Tentacular club is slightly expanded with the tentacular hooks and their associated suckers occupying about 80% of the club (pictured); proximal region of club has a carpal cluster that consists of about 5 small cups and 5 buttons; distal of the carpal cluster are 6 long widely open slender hooks (the proximal and distal hooks are smaller than the other 4, which are subequal); dorsal to the hooks is a distinct

double row of small suckers, one pair to each hook; distal of the hooks lie 4 rows of small suckers which extend to the distal extremity

Size

- Maximum mantle length 42 mm

Distribution

- Philippines and Joseph Bonaparte Gulf

Reference: Voss, 1963, p 92

CUTTLEFISHES FAMILY: Sepiidae

Ovalbone Cuttlefish *Sepia elliptica*

CAAB: 23 607003

Specimen: 85 mm ML



Dorsal view

Diagnostic characters

- Cuttlebone outline oval
- Inner cone limbs broadening posteriorly, not thickened, outer margin of inner cone forms a raised flattened anteriorly directed ledge

Colour

- Pale pinkish purple

Size

- Maximum mantle length 99 mm

Distribution

- Northern Australia and New Guinea, South China Sea, possibly Philippines



Ventral view
cuttlebone

Reference: Carpenter and Niem (eds), 1998, Vol 2, p 741

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nomenclatural issues

The scientific and common names used for fish, follow the Australian standard maintained by CSIRO Marine and Atmospheric Research, and Australia's Fish Names Committee. There are, however, a few exceptions: *Valamugil perusii*, *Secutor ruconis* and *Apogon fasciatus*. Australian specimens of these three species are being reviewed and did not have an Australian standard name assigned at the time of publishing. For these species, I have used common names from other references. Similarly, Australian standard names have not yet been assigned to invertebrates. Their common names herein have also been derived from other sources. Several crab and mantis shrimp species, in fact, have never been designated a common name. The 'Cornflake Crab' assigned to *Charybdis callianassa*, is a common name used by NPF fishers.

Objective 4: Bycatch descriptions of banana prawn fisheries

Appendix A: List of taxa caught as bycatch in 53 trawls in the Joseph Bonaparte Gulf between May 2003 and September 2005. The contribution by each taxa to the catch in terms of percentage occurrence, biomass and number is shown. The catch rates were determined for vessels towing twin rig configuration (mean net size 21.2 ± 0.46 m). Teleost species in bold without catch rates were recorded from samples collected opportunistically from various sampling methods. Other taxa with asterisk in n h⁻¹ column indicate that the number of individuals was not recorded. 'Elasmobranchs (b)' species were not recorded from subsamples but are caught in JBG prawn trawls.

Bycatch group	Common name	Taxa	Family level (unless stated)	% occurrence (n=53)	Catch rate (kg h ⁻¹) Mean ± (se)	Catch rate (n h ⁻¹) Mean ± (se)
Teleosts	Largehead Hairtail	<i>Trichiurus lepturus</i>	Trichiuridae	98	59.665 (12.308)	878.859 (183.003)
	Blackfin Threadfin	<i>Polydactylus nigripinnis</i>	Polynemidae	94	44.983 (22.587)	957.055 (515.174)
	Smooth Jewfish	<i>Johnius laevis</i>	Sciaenidae	94	43.823 (12.997)	1872.039 (586.200)
	Common Hairfin Anchovy	<i>Setipinna tenuifilis</i>	Engraulidae	94	29.763 (5.624)	1569.985 (311.598)
	Threadfin Scat	<i>Rhinoprenes pentanemus</i>	Rhinoprenidae	92	67.402 (19.585)	2442.553 (723.690)
	River Jewfish	<i>Johnius borneensis</i>	Sciaenidae	89	4.078 (0.840)	150.578 (31.102)
	Pinochio Tongue Sole	<i>Paraplagusia longirostris</i>	Cynoglossidae	89	3.861 (0.990)	87.316 (18.923)
	Shortfin Saury	<i>Saurida argentea</i>	Synodontidae	85	4.716 (1.205)	53.020 (15.095)
	Brownmargin Flathead	<i>Cociella hutchinsi</i>	Platycephalidae	81	3.299 (0.668)	79.324 (14.311)
	Glassy Bombay Duck	<i>Harpadon transluens</i>	Synodontidae	79	69.935 (28.916)	812.557 (383.286)
	Ditchelee	<i>Pellona ditchela</i>	Clupeidae	79	13.161 (7.945)	548.522 (301.193)
	Orange Jewfish	<i>Atrubucca brevis</i>	Sciaenidae	77	1.461 (0.372)	46.121 (10.868)
	Longjaw Thryssa	<i>Thryssa setirostris</i>	Engraulidae	72	14.391 (7.746)	341.547 (163.816)
	Pearlyfin Cardinalfish	<i>Apogon poecilopterus</i>	Apogonidae	68	0.791 (0.177)	154.375 (34.097)
	Largescale Grunter	<i>Terapon theraps</i>	Terapontidae	64	2.696 (0.961)	103.712 (36.98)
	Striped cardinalfish	<i>Apogon fasciatus</i>	Apogonidae	60	0.586 (0.175)	83.718 (25.292)
	Nakedhead Catfish	<i>Euristhmus nudiceps</i>	Plotosidae	58	0.976 (0.327)	27.660 (9.736)
	Sunrise Goatfish	<i>Upeneus sulphureus</i>	Mullidae	55	1.608 (0.636)	42.894 (16.302)
	Opaline Lanternfish	<i>Benthosema pterotum</i>	Myctophidae	53	0.130 (0.048)	248.728 (75.214)
	Blackspot Butterfish	<i>Psenopsis humerosa</i>	Centrolophidae	49	1.445 (0.442)	49.013 (17.114)
	Sharptooth Seabass	<i>Synagrops philippinensis</i>	Percichthyidae	45	0.325 (0.106)	56.163 (16.175)
	Deep pugnose ponyfish	<i>Secutor ruconius</i>	Leiognathidae	43	0.179 (0.089)	64.781 (30.591)
	Harris' Flathead	<i>Inegocia harrisi</i>	Platycephalidae	42	1.049 (0.506)	27.944 (11.515)
	Smooth Gurnard	<i>Lepidotrigla russelli</i>	Triglidae	42	0.247 (0.092)	19.666 (11.709)
	Threadfin Leatherjacket	<i>Paramonacanthus filicauda</i>	Monacanthidae	38	0.134 (0.059)	13.828 (7.288)
	Striped Barracuda	<i>Sphyræna obtusata</i>	Sphyræniidae	36	0.893 (0.276)	13.597 (4.193)
	Ornate Threadfin Bream	<i>Nemipterus hexodon</i>	Nemipteridae	34	1.064 (0.396)	17.222 (6.537)
	Brownback Toadfish	<i>Lagocephalus spadiceus</i>	Tetraodontidae	34	0.177 (0.049)	6.242 (1.575)
	Longtail Illisha	<i>Ilisha lunula</i>	Clupeidae	30	1.549 (0.907)	27.842 (16.483)
	Blotched Javelin	<i>Pomadasyd maculatus</i>	Haemulidae	25	0.335 (0.19)	6.741 (3.119)
	Twoblotch Ponyfish	<i>Leiognathus blochii</i>	Leiognathidae	25	0.056 (0.026)	17.667 (8.148)
	Waite's Flounder	<i>Amoglossus waitei</i>	Bothidae	25	0.011 (0.005)	5.481 (2.985)
	Hamilton's Thryssa	<i>Thryssa hamiltonii</i>	Engraulidae	23	1.291 (0.685)	14.762 (8.332)
	Sixbar Grouper	<i>Epinephelus sexfasciatus</i>	Serranidae	23	0.304 (0.135)	3.920 (1.636)
	Southern Yellow Jewfish	<i>Larimichthys pamoides</i>	Sciaenidae	21	1.240 (0.733)	41.575 (22.535)
	Blacktip Tipodfish	<i>Trixipichthys weberi</i>	Triacanthidae	21	0.260 (0.140)	11.785 (6.569)
	Shieldhead Catfish	<i>Arius nella</i>	Ariidae	19	0.628 (0.404)	8.910 (4.794)
	Ogilby's Tongue Sole codlets	<i>Cynoglossus ogilbyi</i>	Cynoglossidae	19	0.102 (0.037)	4.467 (1.541)
		Bregmacerotidae	Bregmacerotidae	19	0.012 (0.008)	8.918 (4.714)
	Shortfin Battfish	<i>Zabidus novemaculeatus</i>	Ephippidae	17	0.200 (0.144)	5.910 (4.102)
	Plain Porcupinefish	<i>Cyclichthys hardenbergi</i>	Diodontidae	15	0.747 (0.430)	8.859 (4.745)
	Malabar Trevally	<i>Carangoides malabaricus</i>	Carangidae	15	0.372 (0.265)	9.368 (5.665)
	Longfinned mullet	<i>Valamugil perusii</i>	Mugilidae	15	0.302 (0.230)	5.198 (3.921)
	Black Pomfret	<i>Parastromateus niger</i>	Carangidae	15	0.277 (0.157)	3.718 (2.012)
	Golden Cusk	<i>Srembo imberbis</i>	Ophidiidae	15	0.022 (0.010)	1.969 (0.945)
	Creamspotted Cardinalfish	<i>Apogon albimaculosus</i>	Apogonidae	15	0.013 (0.006)	2.033 (1.018)
	Unicorn Codlet	<i>Bregmaceros maclellandi</i>	Bregmacerotidae	15	0.009 (0.005)	5.035 (3.378)
	Finny Scad	<i>Megalaspis cordyla</i>	Carangidae	13	0.297 (0.146)	10.259 (4.917)
	Largescale Saury	<i>Saurida undosquamis</i>	Synodontidae	13	0.190 (0.127)	7.398 (4.155)
	Australian Halibut	<i>Psettodes erumei</i>	Psettodidae	11	0.236 (0.110)	0.917 (0.468)
	Large-tooth Flounder	<i>Pseudorhombus arsius</i>	Paralichthyidae	11	0.200 (0.098)	1.972 (1.000)

Objective 4: Bycatch descriptions of banana prawn fisheries

Bycatch group	Common name	Taxa	Family level (unless stated)	% occurrence (n=53)	Catch rate (kg h ⁻¹) Mean ± (se)	Catch rate (n h ⁻¹) Mean ± (se)
	Bluespotted Trevally	<i>Caranx bucculentus</i>	Carangidae	11	0.152 (0.081)	1.741 (0.884)
	Dwarf Flathead	<i>Elates ransonnetii</i>	Platycephalidae	11	0.041 (0.025)	3.161 (2.041)
	Common Pike Eel	<i>Muraenesox bagio</i>	Muraenesocidae	9	0.129 (0.072)	0.824 (0.467)
	Anchovy	<i>Stolephorus</i> spp	Engraulidae	9	0.012 (0.006)	0.982 (0.520)
	Yellow Waspfish	<i>Cottapistus cottoides</i>	Tetrarogidae	9	0.008 (0.004)	0.847 (0.416)
	Trewavas' Croaker	<i>Johnius cf trewasasae</i>	Sciaenidae	8	1.508 (1.499)	69.402 (68.940)
	Mouth Mackerel	<i>Rastrelliger kanagurta</i>	Scombridae	8	0.188 (0.103)	1.368 (0.743)
	Smallspeckled Herring	<i>Herkotsichthys lippa</i>	Clupeidae	8	0.049 (0.031)	1.653 (1.001)
	Longfin Waspfish	<i>Apistus carinatus</i>	Apistidae	8	0.037 (0.027)	1.533 (1.107)
	Pugnose ponyfish	<i>Secutor insidiator</i>	Leiognathidae	8	0.029 (0.018)	3.576 (2.254)
	Razorbelly Trevally	<i>Caranx kleinii</i>	Carangidae	8	0.026 (0.015)	0.714 (0.362)
	Flagfin Cardinalfish	<i>Apogon truncatus</i>	Apogonidae	8	0.012 (0.010)	3.023 (2.471)
	Dusky Velvetfish	<i>Aploactis aspera</i>	Aploactinidae	8	0.010 (0.008)	1.233 (0.767)
	Rough Golden Toadfish	<i>Lagocephalus lunaris</i>	Tetraodontidae	6	0.694 (0.521)	0.709 (0.524)
	Humpback Hairfin Anchovy	<i>Setipinna paxtoni</i>	Engraulidae	6	0.432 (0.358)	21.569 (16.199)
	Silvermouth Trevally	<i>Ulua aurochs</i>	Carangidae	6	0.079 (0.050)	1.100 (0.736)
	Orangefin Ponyfish	<i>Leiognathus bindus</i>	Leiognathidae	6	0.033 (0.024)	15.394 (12.727)
	Indian Scad	<i>Decapterus russelli</i>	Carangidae	6	0.033 (0.022)	0.794 (0.516)
	Threadfin Silverbiddy	<i>Gerres filamentosus</i>	Gerreidae	6	0.022 (0.014)	0.820 (0.519)
	Giant Sea Catfish	<i>Arius thalassinus</i>	Ariidae	6	0.021 (0.012)	0.430 (0.263)
	Black Jewfish	<i>Protonibea diacanthus</i>	Sciaenidae	6	0.012 (0.007)	0.224 (0.129)
	Crescent Grunter	<i>Terapon jarbua</i>	Terapontidae	6	0.010 (0.005)	0.224 (0.129)
	Plumbstriped Stingfish	<i>Minous versicolor</i>	Synanceiidae	6	0.008 (0.005)	0.517 (0.319)
	Humpback Anglerfish	<i>Tetrabrachium ocellatum</i>	Tetrabrachiidae	6	0.002 (0.002)	0.394 (0.243)
	Smallhead Flounder	<i>Laeops parviceps</i>	Bothidae	6	0.002 (0.001)	0.748 (0.408)
	Barred Javelin	<i>Pomadasys kaakan</i>	Haemulidae	4	0.197 (0.169)	0.181 (0.127)
	Lemon Tongue Sole	<i>Paraplagusia bilineata</i>	Cynoglossidae	4	0.124 (0.121)	1.195 (1.087)
	Smallmouth Scad	<i>Alepes apercna</i>	Carangidae	4	0.059 (0.056)	0.460 (0.379)
	Shorthose Boxfish	<i>Rhynchostracion nasus</i>	Ostraciidae	4	0.050 (0.050)	0.369 (0.313)
	Longtail Catfish	<i>Euristmus lepturus</i>	Plotosidae	4	0.044 (0.031)	0.150 (0.107)
	Whitetongue Trevally goatfish	<i>Carangoides talamparoides</i>	Carangidae	4	0.026 (0.024)	0.587 (0.417)
	lizardfish	Mullidae	Mullidae	4	0.007 (0.006)	0.235 (0.172)
	Saurida spp	<i>Saurida</i> spp	Synodontidae	4	0.003 (0.002)	0.249 (0.183)
	Whitsunday Wire Eel	<i>Saurenchelys finitimus</i>	Nettastomatidae	4	0.002 (0.002)	0.470 (0.330)
	Flathead Dragonet	<i>Repomucenus belcheri</i>	Callionymidae	2	<0.001 (0.000)	0.279 (0.279)
	Greedy Gaper	<i>Champsodon vorax</i>	Champsodontidae	2	<0.001 (0.000)	0.153 (0.153)
	worm eel	<i>Muraenichthys</i> spp	Ophichthidae	2	<0.001 (0.000)	0.224 (0.224)
	Blue Threadfin	<i>Eleutheronema tetradactylum</i>	Polynemidae	2	0.090 (0.090)	0.341 (0.341)
	Needleskin Queenfish	<i>Scomberoides tol</i>	Carangidae	2	0.080 (0.080)	0.102 (0.102)
	Epulette Trevally	<i>Carangoides humerosus</i>	Carangidae	2	0.041 (0.041)	0.341 (0.341)
	Longfin Silverbiddy	<i>Pentaprion longimanus</i>	Gerreidae	2	0.018 (0.018)	2.928 (2.928)
	Purplespotted Bigeye	<i>Priacanthus tayenus</i>	Priacanthidae	2	0.016 (0.016)	0.153 (0.153)
	Sharksucker	<i>Echeneis naucrates</i>	Echeneidae	2	0.014 (0.014)	0.217 (0.217)
	Sevenband Cardinalfish	<i>Apogon septemstriatus</i>	Apogonidae	2	0.012 (0.012)	3.829 (3.829)
	Jewfish	Sciaenidae	Sciaenidae	2	0.010 (0.010)	0.341 (0.341)
	Dusky Tongue Sole	<i>Paraplagusia sinerama</i>	Cynoglossidae	2	0.008 (0.008)	0.137 (0.137)
	Threespine Frogfish	<i>Batrachomoeus trispinosus</i>	Batrachoididae	2	0.007 (0.007)	0.093 (0.093)
	Yellowtail Jewfish	<i>Austronibea oedogenys</i>	Sciaenidae	2	0.007 (0.007)	0.137 (0.137)
	Saddletail Snapper	<i>Lutjanus malabaricus</i>	Lutjanidae	2	0.005 (0.005)	0.088 (0.088)
	Razor Moonfish	<i>Mene maculata</i>	Menidae	2	0.005 (0.005)	0.217 (0.217)
	Silver Toadfish	<i>Lagocephalus sceleratus</i>	Tetraodontidae	2	0.004 (0.004)	0.224 (0.224)
	White Sardinella	<i>Sardinella albella</i>	Clupeidae	2	0.004 (0.004)	0.217 (0.217)
	Slender Sardine	<i>Dussumieria elopsoides</i>	Clupeidae	2	0.004 (0.004)	0.164 (0.164)
	Shaggy Anglerfish	<i>Antennarius hispidus</i>	Antennariidae	2	0.004 (0.004)	0.133 (0.133)
	Plaintail Lionfish	<i>Pterois russelli</i>	Pteroidae	2	0.003 (0.003)	0.086 (0.086)
	Blackspot Waspfish	<i>Liocranium praepositum</i>	Tetrarogidae	2	0.003 (0.003)	0.088 (0.088)
	Luminous Conger	<i>Lumiconger arafura</i>	Congridae	2	0.003 (0.003)	0.246 (0.246)
	Muller's Coralfish	<i>Chelmon muelleri</i>	Chaetodontidae	2	0.002 (0.002)	0.093 (0.093)

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Objective 4: Bycatch descriptions of banana prawn fisheries

Bycatch group	Common name	Taxa	Family level (unless stated)	% occurrence (n=53)	Catch rate (kg h ⁻¹) Mean ± (se)	Catch rate (n h ⁻¹) Mean ± (se)
	Fringefin Trevally	<i>Pantolabus radiatus</i>	Carangidae	2	0.002 (0.002)	0.090 (0.090)
	Yellow Largescale Flounder	<i>Brachypleura novaezeelandiae</i>	Citharidae	2	0.002 (0.002)	0.153 (0.153)
	Slender Conger	<i>Uroconger lepturus</i>	Congridae	2	0.001 (0.001)	0.098 (0.098)
	Smooth Golden Toadfish	<i>Lagocephalus inermis</i>	Tetraodontidae	2	0.001 (0.001)	0.070 (0.070)
	Longfin Saury	<i>Saurida longimanus</i>	Synodontidae	2	0.001 (0.001)	0.308 (0.308)
	Monster Cardinalfish	<i>Apogon melanopus</i>	Apogonidae	*	* *	* *
	Boschma's Anglerfish	<i>Lophichthys boschmai</i>	Lophichthyidae	*	* *	* *
	Pinkbreast Siphonfish	<i>Siphamia roseigaster</i>	Apogonidae	*	* *	* *
Elasmobranchs (a)						
	Plain Maskray	<i>Dasyatis annotata</i>	Dasyatidae	40	1.141 (0.266)	6.731 (1.374)
	Milk Shark	<i>Rhizoprionodon acutus</i>	Carcharhinidae	9	0.289 (0.126)	1.013 (0.484)
	Winghead Shark	<i>Eusphyrus blochii</i>	Sphyrnidae	8	0.163 (0.095)	0.517 (0.307)
	Australian Butterfly Ray	<i>Gymnura australis</i>	Gymnuridae	6	1.166 (0.955)	0.771 (0.483)
	Blackspeckled Whipray	<i>Himantura toshi</i>	Dasyatidae	6	0.651 (0.615)	0.724 (0.670)
	Weasel Shark	<i>Hemigaleus australiensis</i>	Hemigaleidae	2	0.240 (0.240)	0.740 (0.740)
% occurrence (n=23)						
Elasmobranchs (b)						
	Whitespotted Guitarfish	<i>Rhynchobatus australiae</i>	Rhynchobatidae	30	* *	* *
	Australian Blacktip Shark	<i>Carcharhinus tilstoni</i>	Carcharhinidae	17	* *	* *
	Grey Carpetshark	<i>Chiloscyllium punctatum</i>	Hemiscylliidae	17	* *	* *
	Narrow Sawfish	<i>Anoxypristis cuspidata</i>	Pristidae	9	* *	* *
	Whitecheek Shark	<i>Carcharhinus dussumieri</i>	Carcharhinidae	4	* *	* *
	Green Sawfish	<i>Pristis zijsron</i>	Pristidae	4	* *	* *
Invertebrates						
	Cornflake Crab	<i>Charybdis callianassa</i>	Portunidae	98	22.555 (5.629)	3205.650 (848.980)
	Rough Prawn	<i>Trachypenaeus gonospinifer</i>	Penaeidae	91	2.136 (0.859)	1264.131 (465.979)
	Northern Velvet Prawn	<i>Metapenaeopsis novaeguineae</i>	Penaeidae	91	1.560 (0.285)	328.146 (72.638)
	Coral Prawn	<i>Solenocera australiana</i>	Solenoceridae	81	1.341 (0.304)	196.382 (40.374)
	Southern Rough Prawn	<i>Trachypenaeus curvirostris</i>	Penaeidae	74	0.763 (0.221)	160.276 (52.042)
	Mantis Shrimp	<i>Oratosquilla gravieri</i>	Squillidae	66	0.255 (0.055)	38.383 (9.208)
	Ovalbone Cuttlefish	<i>Sepia elliptica</i>	Sepiidae	62	0.585 (0.120)	26.414 (6.773)
	Northern Rough Prawn squid	<i>Photololigo</i> spp	Loliginidae	58	0.512 (0.134)	57.344 (18.254)
	swimmer crab	<i>Portunus hastatooides</i>	Portunidae	51	0.368 (0.182)	133.541 (67.653)
	swimmer crab	<i>Portunus gracilimanus</i>	Portunidae	43	0.301 (0.151)	59.929 (31.044)
	Blunt-toothed Crab	<i>Charybdis truncata</i>	Portunidae	34	0.352 (0.207)	58.094 (35.72)
	Firefly Squid	<i>Abralia armata</i>	Enoplateuthidae	34	0.108 (0.036)	28.946 (9.768)
	hydroids	Hydroida	Order Hydroida	34	0.059 (0.026)	* *
	carid prawn	<i>Pontocaris arafurae</i>	Crangonidae	34	0.007 (0.002)	9.461 (3.748)
	carid prawn	<i>Procladius levicarina</i>	Pandalidae	30	0.014 (0.009)	13.573 (8.862)
	Southern Velvet Prawn	<i>Metapenaeopsis palmensis</i>	Penaeidae	28	0.107 (0.049)	24.167 (11.238)
	mantis shrimp	<i>Dictyosquilla tuberculata</i>	Squillidae	26	0.023 (0.008)	3.315 (1.101)
	jellyfish	Scyphozoa	Class Scyphozoa	23	0.335 (0.176)	4.568 (1.743)
	Three-spotted Crab	<i>Portunus sanguinolentus</i>	Portunidae	23	0.322 (0.131)	2.775 (1.006)
	Orange Prawn	<i>Atypopenaeus formosus</i>	Penaeidae	21	0.026 (0.011)	6.970 (3.164)
	Mudbug	<i>Thenus indicus</i>	Scyllaridae	19	0.145 (0.061)	2.135 (0.968)
	crab	<i>Jonas leuteanus</i>	Corystidae	17	0.012 (0.005)	1.650 (0.547)
	hermit crabs	Paguroidea	Superfamily Paguroidea	15	0.003 (0.002)	1.505 (0.786)
	zoanthid anemone	<i>Sphenopus marsupialis</i>	Spenopidae	13	0.078 (0.064)	2.029 (1.487)
	feather stars	Crinoidea	Class Crinoidea	11	0.062 (0.042)	0.567 (0.264)
	sea urchin	<i>Chaetodiadema granulatum</i>	Diademidae	9	0.017 (0.009)	0.529 (0.242)
	Crucifix Crab	<i>Charybdis feriata</i>	Portunidae	8	0.127 (0.105)	1.644 (1.154)
	sponges	Porifera	Phylum Porifera	8	0.064 (0.047)	* *
	octopus	<i>Octopus</i> spp	Octopodidae	8	0.018 (0.010)	0.850 (0.475)

Objective 4: Bycatch descriptions of banana prawn fisheries

Bycatch group	Common name	Taxa	Family Level (unless stated)	% occurrence (n=23)	Catch rate (kg h ⁻¹) Mean ± (se)	Catch rate (n h ⁻¹) Mean ± (se)
	slipper lobster	<i>Biarctus sordidus</i>	Scyllaridae	8	0.009 (0.007)	2.064 (1.802)
	algae	Sargassaceae	Sargassaceae	8	0.007 (0.004)	* *
	penaeid prawn	<i>Metapenaeopsis</i> spp	Penaeidae	8	0.003 (0.002)	6.355 (3.325)
	swimmer crab	<i>Charybdis yaldwyni</i>	Portunidae	8	0.003 (0.002)	0.396 (0.204)
	dumpling squid	<i>Euprymna cf tasmanica</i>	Sepiidae	8	0.003 (0.001)	0.617 (0.331)
	soft corals	Nephtidae	Nephtidae	6	0.027 (0.014)	0.093 (0.093)
	sea pen	Pennatulidae	Pennatulidae	6	0.011 (0.010)	1.017 (0.823)
	penaeid prawn	Penaeidae	Penaeidae	6	0.004 (0.003)	7.506 (5.964)
	Mud Spiny Lobster	<i>Panulirus polyphagus</i>	Palinuridae	4	0.096 (0.074)	0.493 (0.387)
	mantis shrimp	<i>Oratosquilla interrupta</i>	Squillidae	4	0.012 (0.009)	0.597 (0.460)
	Mantis Shrimp	<i>Harpiosquilla harpax</i>	Squillidae	4	0.012 (0.009)	0.294 (0.222)
	fire worm	<i>Chloela flava</i>	Amphinomidae	4	0.004 (0.003)	0.335 (0.260)
	swimmer crab	<i>Charybdis hellerii</i>	Portunidae	4	0.004 (0.002)	0.159 (0.114)
	spider crab	<i>Phalangipus longipes</i>	Majidae	4	0.003 (0.003)	0.711 (0.619)
	spider crab	<i>Phalangipus australiensis</i>	Majidae	4	0.003 (0.002)	0.911 (0.756)
	Papuan Cuttlefish	<i>Sepia papuensis</i>	Sepiidae	4	0.002 (0.001)	0.163 (0.118)
	swimmer crab	<i>Lupocyclus rotundatus</i>	Portunidae	4	0.001 (0.001)	0.277 (0.203)
	brittle stars	Ophiuroidea	Class Ophiuroidea	2	<0.001 (0.000)	0.093 (0.093)
	crab	<i>Cryptoltea</i> sp.	Pilumnidae	2	<0.001 (0.000)	0.056 (0.056)
	crab	<i>Harrovia tuberculata</i>	Pilumnidae	2	<0.001 (0.000)	0.370 (0.370)
	swimmer crab	<i>Lissocarcinus arkati</i>	Portunidae	2	<0.001 (0.000)	0.086 (0.086)
	crab	<i>Atergatopsis tweediei</i>	Xanthidae	2	<0.001 (0.000)	0.090 (0.090)
	sea urchins	Echinoidea	Class Echinoidea	2	0.024 (0.024)	1.850 (1.850)
	Mantis Shrimp	<i>Erugosquilla woodmasoni</i>	Squillidae	2	0.015 (0.015)	0.433 (0.433)
	Web's Cuttlefish	<i>Sepiella weberi</i>	Sepiidae	2	0.013 (0.013)	0.740 (0.740)
	sea cucumber	Stichopodidae	Stichopodidae	2	0.011 (0.011)	0.170 (0.170)
	sea cucumber	Holothuridae	Holothuridae	2	0.010 (0.010)	0.433 (0.433)
	sea squirts	Ascididae	Ascididae	2	0.010 (0.010)	0.246 (0.246)
	swimmer crab	<i>Thalassidroma sima</i>	Portunidae	2	0.008 (0.008)	0.459 (0.459)
	Arafura Prawn	<i>Parapenaeopsis arafurica</i>	Penaeidae	2	0.004 (0.004)	0.349 (0.349)
	crab	<i>Carcinoplax purpurea</i>	Goneptacidae	2	0.004 (0.004)	0.919 (0.919)
	swimmer crab	<i>Lupocyclus tugelae</i>	Portunidae	2	0.003 (0.003)	0.220 (0.220)
	swimmer crab	<i>Charybdis jaubertensis</i>	Portunidae	2	0.002 (0.002)	0.206 (0.206)
	Pharaoh Cuttlefish	<i>Sepia pharaonis</i>	Sepiidae	2	0.001 (0.001)	0.088 (0.088)
	Slow Prawn	<i>Axiopsis consobrina</i>	Axiidae	2	0.001 (0.001)	0.900 (0.900)
	hairy crab	<i>Glabropilumnus seminudus</i>	Pilumnidae	2	0.001 (0.001)	0.278 (0.278)
	penaeid prawn	<i>Parapenaeus</i> spp	Penaeidae	2	0.001 (0.001)	0.450 (0.450)
	Hardback Prawn	<i>Trachypenaeus granulatus</i>	Penaeidae	2	0.001 (0.001)	0.089 (0.089)
				% occurrence (n=27)		
Reptiles (sea snakes)	elegant seasnake	<i>Hydrophis elegans</i>	Hydrophidae	22	* *	* *
	olive-headed seasnake	<i>Distenia major</i>	Hydrophidae	11	* *	* *
	spine-bellied seasnake	<i>Lapemis hardwickii</i>	Hydrophidae	4	* *	* *

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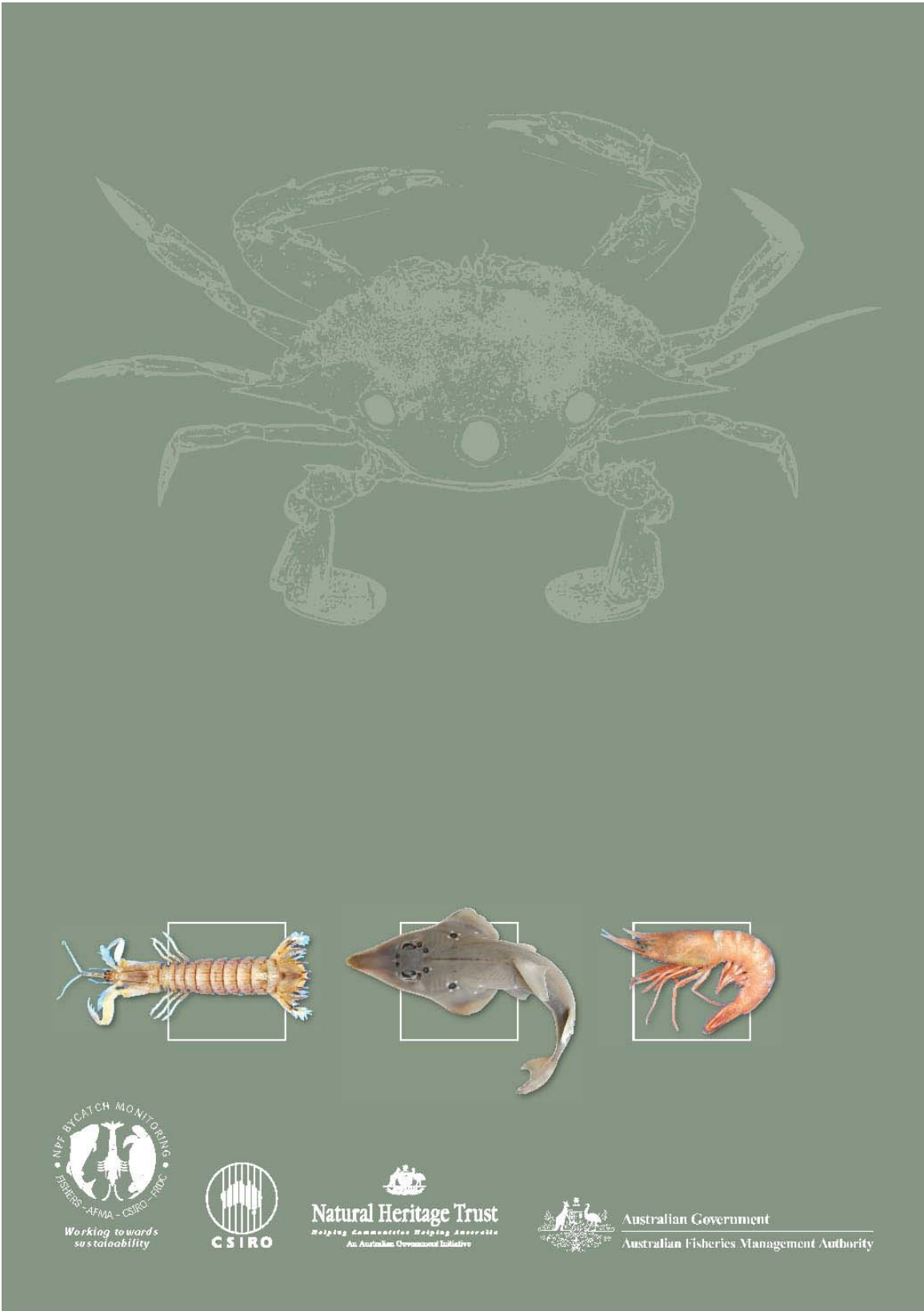
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5.5.4 Description of the Gulf of Carpentaria banana prawn fishery bycatch and a contrast with the tiger prawn fishery bycatch

Q Dell, D. Brewer, D. Heales, S. Griffiths and M. Tonks

Abstract

The bycatch of a unique tropical Australian banana prawn trawl fishery (*Penaeus merguensis*) is described for the first time and contrasted with the closely related tiger prawn trawl fishery (*Penaeus semisulcatus* and *Penaeus esculentus*), that operates in the same managed region with the same vessels and skippers. The bycatch of the banana prawn fishery was obtained in 2004 and 2005 by scientific observers aboard commercial vessels. The bycatch from the fishery was previously thought to be negligible in comparison to other prawn trawl fisheries, but it makes up a mean 43.5% of the total catch per trawl. It consists of small teleosts (75.3%), small to medium-sized elasmobranchs (23.1%) and invertebrates (1%). While the overall bycatch was diverse (218 spp.), just over three quarters of the similarity between catches was accounted for by three species (*Polydactylus multiradiatus*, *Caranx bucculentus* and *Rhizoprionodon acutus*). Together, these three species accounted for 51.4% of the total bycatch weight (42.8% accounted for by *P. multiradiatus*). The bycatch assemblage structure of the banana prawn fishery was significantly different to that of the adjoining tiger prawn fishery. The banana prawn fishery bycatch had a higher mean bycatch catch rate from shorter duration trawls, although a lower estimated total bycatch (1,502.1 t yr⁻¹) than the longer fishing season of the tiger prawn fishery (20,073.4 t yr⁻¹). This study highlights that management and long-term monitoring goals for bycatch need to consider all of the fishing components; particularly fisheries with multiple target species, and those with large spatial and temporal distributions.

Keywords: Prawn (shrimp) trawling, Tropical, Bycatch, *Penaeus merguensis* de Man, Monitoring, Assemblage structure

Introduction

The effects of fishing effort on bycatch (non-target species) are a global concern, particularly in trawl fisheries that typically use non-selective methods (Kennelly, 1995). The small-mesh nets of prawn (shrimp) trawls can catch many non-target species (Harris and Poiner, 1990; Ye et al., 2000; Sánchez et al., 2004; Harrington et al., 2005); and the catch in tropical regions usually includes a wide variety of species and habitats including benthopelagic fish, reptiles and sessile benthic communities (Pitcher et al., 2000; Stobutzki et al., 2001 *a*; Kaiser et al., 2002; Burrige et al., 2003; Brewer et al., 2006 *a*). However, the type and degree of effect depends on many factors such as the trawl gear used (Revell et al., 2006), the area fished (Zeller and Pauly, 2005) or the time of day for fishing operations (Blaber et al., 1994).

In many countries, changes to fishing practices have been influenced by informed science, legislation, conservation bodies, market drivers and codes of conduct (FAO, 1995; Hall and Mainprize, 2005). Fisheries management is moving toward demonstrating the sustainability of all species and communities, via processes such as ecosystem-based fisheries management and independent accreditation (Garcia and Staples, 2000; Sinclair et al., 2002). These approaches require intricate knowledge of the species involved, their inter-specific relationships and the effects of fishing on each species. Effective management in this context requires a significant increase in information, research capacity and scope (Garcia and Staples, 2000). Unfortunately this is considerably difficult when there are rarely encountered, low-value and data-limited bycatch species; particularly where hundreds of species are

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involved. While demonstrating sustainability in this context is difficult, some new ecological risk assessment methods have gone some way to solving this problem (e.g. Stobutzki et al., 2001 *b*; Griffiths et al., 2006; Zhou and Griffiths, in press). However, for fisheries to comply with increasingly stringent legislation, both new risk assessment methods and management rely on accurate information on the impacts of all mortality sources.

Australia's tropical Northern Prawn Fishery (NPF) is in a remote and vast area, adjacent to around 6,000 km of the northern coastline of the continent (Figure 43). Here, the fishers target several penaeid species in various regions over two seasons (e.g. Stobutzki et al., 2001 *a*; Kenyon et al., 2004). This study concentrates on the Gulf of Carpentaria region of this fishery, where two temporally distinct penaeid trawl fisheries operate. These are the 'banana prawn' trawl fishery which primarily targets *Penaeus merguensis* in the first half of the year (typically April to May), and the mixed species 'tiger prawn' trawl fishery which primarily targets *Penaeus semisulcatus* and *Penaeus esculentus* in the second half of the year (typically August to November).

Bycatch studies in the NPF have previously focused on the tiger prawn trawl fishery (e.g. Ramm et al., 1990; Pender et al., 1992; Stobutzki et al., 2001a, Brewer et al., 2006a), which is a longer fishing season interacting with more than 500 bycatch species. The information from that fishery has been used as a proxy for all prawn trawl bycatch occurring in the NPF. For the banana prawn fishery there is no information specifically on the bycatch species and therefore there has been no opportunity to discriminate between the bycatch of the two fisheries.

The white banana prawn, *P. merguensis*, is widely distributed and fished throughout the Indo-Pacific. Economically it is one of the most important penaeid species in this region, harvested by trawl and artisanal fleets from the Persian Gulf to northern Australia. It is the basis of extensive commercial trawl fisheries particularly in Australia, New Guinea, Indonesia and the Philippines (Grey et al., 1983; Evans and Wahju, 1996; Chan, 1998). The NPF yielded an annual mean of 3,946 tonnes of *P. merguensis* between 2000 and 2005, comprising about 60% of the total NPF penaeid catch for that period (the majority of *P. merguensis* catch being from the Gulf of Carpentaria) (Haine and Garvey, 2005). It is a unique penaeid fishery in that aggregations of these prawns typically occupy an area up to several thousand square metres. Most of the catch comes from dense isolated aggregations which can yield several tonnes or greater of *P. merguensis* (Lucas et al., 1979).

The first step for investigating any impact of fishing is to have a reliable description of the magnitude and composition of the bycatch (Andrew and Pepperell, 1992). Our study provides the first description of *P. merguensis* trawl bycatch from the Gulf of Carpentaria. It also contrasts the bycatch assemblage structure and fishery operations from two major penaeid fisheries operating in the same region. This (i) allows differentiation of the fisheries, (ii) provides broader knowledge of the effects of all prawn trawl fisheries on bycatch for a region, and (iii) provides data for a long-term bycatch monitoring program being developed for penaeid trawling in northern Australia.

Methods

Collection of bycatch samples from the banana prawn fishery

The banana prawn trawl bycatch was subsampled by scientific observers aboard two commercial trawlers during 2004 and 2005 in the Gulf of Carpentaria (Figure 43). As the first two–three weeks of the season attract the highest fishing effort for *Penaeus merguensis*,

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sampling was undertaken in this period (commencing 15th April 2004 and 9th April 2005). A total of 93 trawls were sampled in the standard fishing areas for *P. merguensis* amongst the rest of the commercial fleet—as it is normal that aggregations of these penaeids are fished simultaneously by a number of vessels (Lucas et al., 1979).

The bycatch subsamples were collected from one randomly chosen net (of the two nets towed per vessel). They were obtained before the catch went into a seawater processing hopper to eliminate bias brought about by the hopper (see Heales et al., 2003 a). For each trawl at least 10% of the catch was subsampled, which provides an adequate representation of the species composition (Heales et al., 2003 a, 2003 b). All subsamples were later processed in the laboratory. Taxa were identified to species level where possible; if not, they were identified to the lowest taxonomic level.

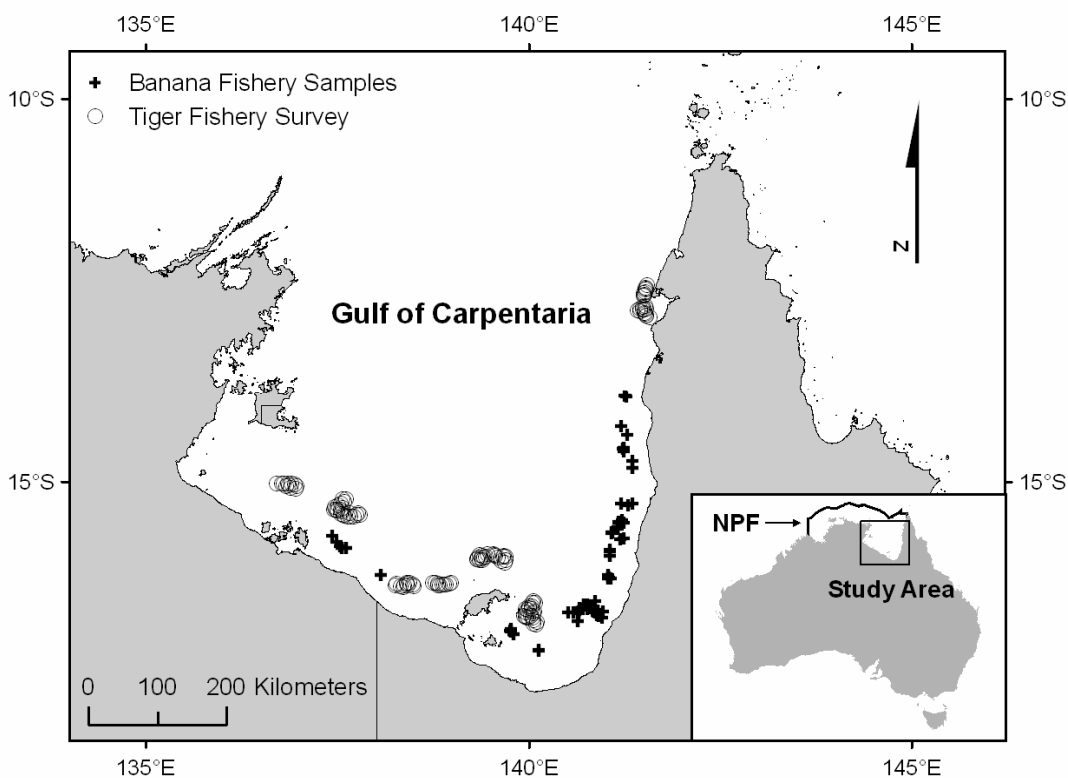


Figure 43. The Gulf of Carpentaria study region showing locations of sampled trawls in the banana prawn fishery and the survey sites of the tiger prawn fishery selected for this study. The Northern Prawn Fishery managed area is shown in the inset as ‘NPF’.

The fishing operations in the banana prawn fishery were of three types based on the fishing behaviour. A ‘prawn school’ trawl had a short duration (~10–30 mins) when the skipper located a school of *P. merguensis* ‘banana prawns’. The fishers located prawn aggregations by the echo-sounder, visual observation of the sea surface, spotter plane (as per Die and Ellis, 1999), or activity of other trawlers. The ‘fish school’ trawl was either schooling fish species that were mistaken for a school of *P. merguensis*, or occasional targeted trawls near fish schools, since fishers sometimes associate fish aggregations (located on their echo-sounders) with the presence of commercial prawns. A ‘fish school’ trawl is also short in duration (~10–30 min) but the bycatch usually comprises most of the catch. The third fishing behaviour (termed ‘dispersed prawn’) occurs when fishers target sparsely distributed *P. merguensis*. This is typically due to the prawns being dispersed as a result of the trawling activity, or

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occasionally much smaller schools of prawns are spread over a greater area. These trawls can be either long or short in duration (~20 min–2 h) and may cover a larger swept area than the other two types of trawling.

Threatened, Endangered and Protected species in the banana prawn fishery

Turtles, syngnathids, sawfish and sea snakes are listed in either the International Union for the Conservation of Nature and Natural Resources (IUCN) Red List of Threatened Species (IUCN, 2004), or under the Australian Environment Protection and Biodiversity Conservation (EPBC) Act (EPBC, 1999), or are listed in both. Taxonomic and size data for all individuals caught from these groups were recorded onboard from both nets; except for syngnathids, which had to be sorted from bycatch subsamples taken from each trawl due to their small size.

Sea snakes were photographed at sea with a 100 mm scale bar and then released. Digital images of each sea snake were later identified and their total length was measured with 'ImageJ' software (Abramoff et al., 2004). Weights were estimated from length–weight relationships calculated for northern Australian species (Fry et al., 2001; Milton, 2001).

Data for the tiger prawn trawl fishery

The banana and tiger prawn fisheries both operate and are both legislated within the same Northern Prawn Fishery managed area. They use similar (though varied) trawl fishing gear and are both fished by the same vessels with the same skippers. In order to assess the impacts of prawn trawling on bycatch in this region, a differentiation of the impacts of the two fisheries needs to be examined.

The bycatch data from the tiger prawn fishery was collected from a survey of high–effort areas in the NPF tiger prawn fishery in 1997 by Stobutzki et al. (2001 *a*). Stobutzki et al. (2001 *a*) sampled throughout the entire NPF and Torres Strait, however, only data from 113 trawls conducted in similar geographical regions to the banana prawn fishing grounds were used (Figure 43). The trawls in that study were deployed at night during the tiger prawn fishery to replicate commercial fishery operations, using NPF industry standard tiger prawn net and gear arrangements (Figure 44a).

The tiger prawn fishery survey was conducted before the use of Turtle Excluder Devices (TEDs) and Bycatch Reduction Devices (BRDs) in the NPF. The species known to be excluded (see Brewer et al., 2006 *b*) were removed from the tiger prawn fishery survey dataset in order to contrast with the banana fishery bycatch (as TEDs and BRDs have been compulsory in the NPF since 2000).

Trawl gear

The banana prawn fishing vessels in our study used the industry standard modified 'Florida Flyer' style nets of a mean 21.2 m \pm 1.1 S.E (11.6 fathom) headrope size with No. 9 Bison otterboards arranged in an industry standard twin–net configuration (Figure 44b). All trawl nets had 50 mm (two inch) stretch–mesh net with codends of 150 mesh long x 150 mesh round.

The current gear arrangement for both fisheries appears similar. They are fished by the same fleet of vessels and use the same fishing gear components (e.g. trawl boards, TEDs, BRDs, trawl net material and codends). However, because of the different prawn behaviour and distribution, there are important differences in the gear arrangement for each fishery. In comparison to the tiger prawn fishery net (Figure 44a), the banana prawn fishery net had the

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addition of extra netting (extra 150–240 net meshes deep) in the front (mouth) of the nets' wingend panel for a greater canopy or ballooning–open effect (Figure 44b). This assists vertical opening in the net mouth, in conjunction with large floats on the headline rope which are towed by long extra–light bridles or flywires (10–15 metres long). The lateral middle/belly section of the banana prawn net is also supported by strengthened seams (belly ropes), which were shackled directly to the top of each trawl board where the headrope is normally attached for a tiger prawn net (Figure 44).

Data analysis

Bycatch to prawn ratios for the banana prawn fishery were calculated from the total subsample weight (with *P. merguensis*) minus the subsample weight without *P. merguensis*. The total bycatch weight of a trawl was extrapolated using the total prawn catch weight and measured bycatch:prawn ratio (in some instances, where the total *P. merguensis* weight could not be measured for a trawl, experienced skippers estimated the total catch).

The catch of each bycatch species was scaled up to estimate the total weight or number caught per tow (two nets), then standardised by the trawl duration. Catch rates are thus expressed as kilograms (kg h^{-1}) and numbers ($n \text{ h}^{-1}$) per hour of trawling for the standard industry twin–net arrangement.

The Gulf of Carpentaria trawl area for *P. merguensis* was classified as east of 135 degrees 24 minutes longitude, south of 12 degrees latitude, for the years 2001–2005. The estimated total bycatch of the banana prawn fishery for this region was calculated from catch rates while trawling (daily catch rate multiplied by trawl effort in days). This excludes 'search time' which is search–only effort. While search time represents a considerable proportion of the fisher's time at sea (as per Lucas et al. 1979; Die and Ellis, 1999), during this time the gear is not deployed at all, hence the catch and bycatch when searching is zero. Trawl and search effort were extracted from the Australian Fisheries Management Authority (AFMA) commercial logbook database for banana prawn trawling operations. We calculated a mean annual estimate of bycatch using the mean fleet trawl effort for the period 2001–2005, multiplied by the 2004 and 2005 mean catch rates for bycatch.

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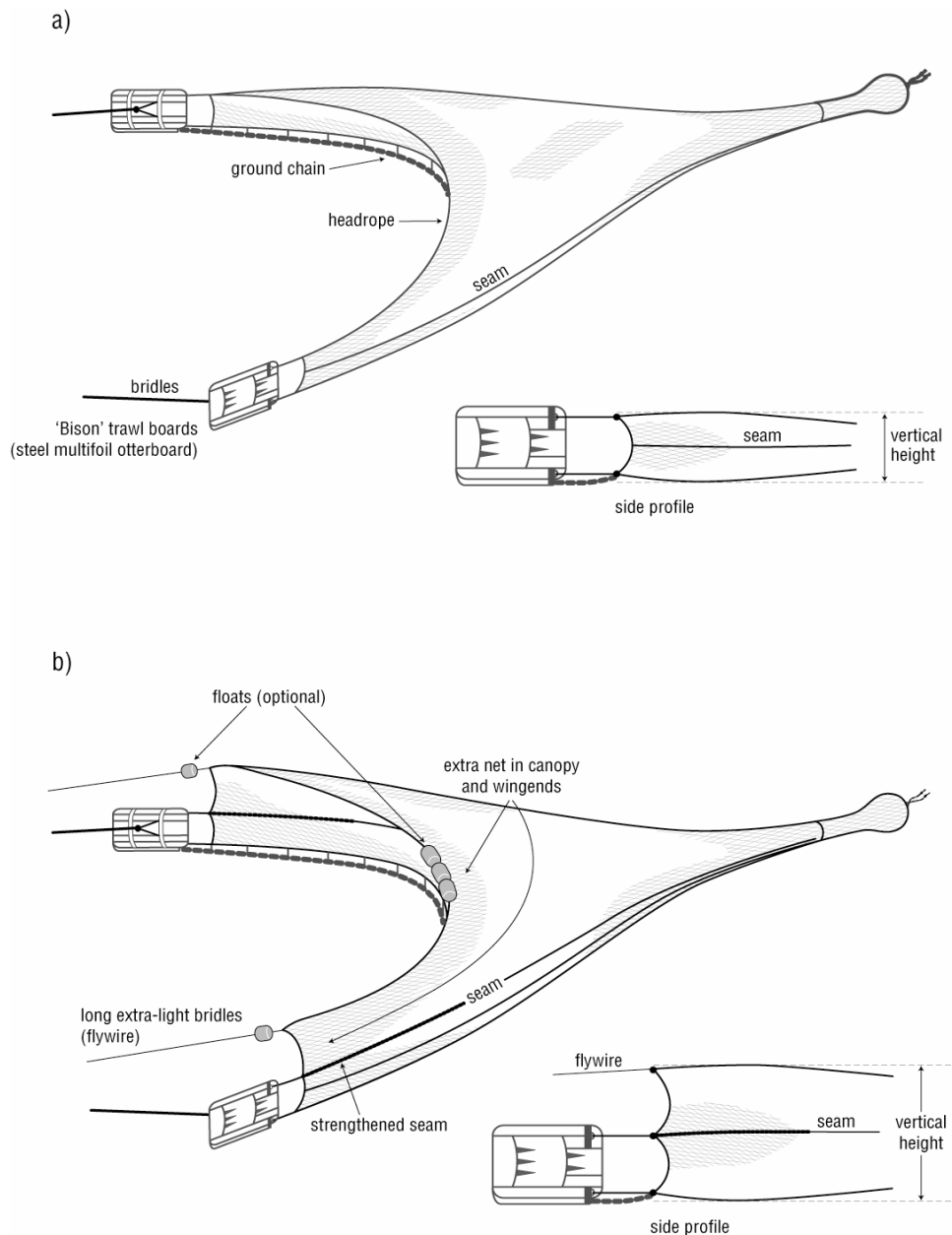


Figure 44. The Northern Prawn Fishery trawl gear arrangement for: (a) tiger prawn trawl fishery net and (b) banana prawn trawl fishery net. The tiger prawn net figure is adapted from an illustration by Gary Day, courtesy of the Australian Maritime College. Turtle Exclusion Devices (TEDs) and Bycatch Reduction Devices (BRDs) are not illustrated.

The estimated total bycatch from the tiger prawn fishery for the Gulf of Carpentaria was calculated using AFMA commercial logbook effort data for the tiger prawn season (for the period 2001–2005). The fleet effort of the tiger prawn fishery for each year was multiplied by the tiger prawn fishery bycatch catch rate; the mean value was then calculated as the estimate for total bycatch.

Non-metric multidimensional scaling (nMDS) was used to explore differences in the bycatch assemblage structure between the banana prawn fishery samples from 2004 and 2005, and between the banana and tiger prawn fisheries in the Gulf of Carpentaria. The analyses incorporated both weight (kg h^{-1}) and numerical abundance ($n \text{ h}^{-1}$) data for each species recorded. Data were fourth-root transformed to reduce the influence of highly numerous or

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heavy taxa and similarity matrices were constructed for both weight and abundance using the Bray–Curtis similarity measure.

Analysis of similarities (ANOSIM) was used to test whether there were differences in bycatch assemblage structure between years sampled in the banana prawn fishery (2004 and 2005) and between the banana and tiger prawn trawl fisheries in the Gulf of Carpentaria. Each ANOSIM comparison involved 9,999 random permutations of the data to test that the observed difference between factors (either fishery or year) did not arise by chance (Clark and Warwick, 2001).

Similarity percentages (SIMPER) were used to identify the species responsible for significant differences between a priori groups (of fishery and year) as determined by ANOSIM. All multivariate analyses were undertaken with PRIMER software (Version 5.2.2, PRIMER–E Ltd., Plymouth, UK).

A Kruskal–Wallis nonparametric test was used to test whether there were any differences between factors (fishing behaviour) in the banana prawn fishery. A Wilcoxon (two–sample) nonparametric test was then used to determine significantly different types of fishing behaviour. Both nonparametric tests were undertaken with ‘R’ statistical software (Version 2.4.0, R Foundation for Statistical Computing, Vienna, Austria).

Results

Composition of the banana prawn fishery’s bycatch

In the Gulf of Carpentaria region the banana prawn trawl fishery caught an estimated 884.1 tonnes of bycatch in 2004 and 1136.3 tonnes in 2005. The estimated annual mean total bycatch for this region in the period 2001 to 2005 was calculated to be 1,502.1 t (± 288.4 S.E). The calculated total annual bycatch taken by the banana prawn fishery during the two–year study consisted of 760.8 t of teleosts (mean 75.3% ± 2.7 S.E of weight per trawl), 233.6 t of elasmobranchs (mean 23.1% ± 2.7 S.E of weight per trawl), and 9.6 t of invertebrates (mean 1% ± 0.2 S.E of weight per trawl). Overall, the banana prawn fishery bycatch made up a mean 43.5% (± 2.6 S.E) of the total catch per trawl; equating to a bycatch to prawn ratio of 0.8:1.

A total of 218 taxa from 80 families—and six higher level taxonomic classifications—comprised the bycatch of the banana prawn fishery (Table 22). These included 150 species of teleosts represented by 51 families, 18 species of elasmobranchs (9 families), 7 species of sea snakes (Hydrophidae), and 43 invertebrate taxa (19 families and six higher level taxonomic classifications). Of the 80 families recorded, 44 were represented by ≤ 2 species. ANOSIM confirmed that there were no significant differences in the bycatch assemblage structure between 2004 and 2005 banana prawn fishery samples.

Objective 4: Bycatch descriptions of banana prawn fisheries

Table 22. Composition and abundance data for families and species of bycatch caught in the banana prawn trawl fishery. Families are represented as the percentage each contributed to the total numbers and weight of all bycatch, species occurrence as a percentage of all trawls sampled.

Family	Family Number (%)	Family Weight (%)	Species Represented per Family	Species Occurrence (%)	Mean Catch Rate n h^{-1} (\pm S.E)	Mean Catch Rate kg h^{-1} (\pm S.E)
<i>TELEOSTS (n = 82 samples)</i>						
Apistidae	<0.1	<0.1	<i>Apistus carinatus</i>	23.2	2.829 (3.801)	0.041 (0.057)
			<i>Apistops caloundra</i>	3.7	0.643 (0.726)	0.012 (0.014)
Apogonidae	<0.1	<0.1	<i>Apogon poecilopterus</i>	8.5	3.778 (3.841)	0.026 (0.026)
			<i>Apogon fasciatus</i>	3.7	0.437 (0.520)	0.004 (0.005)
Ariidae	0.1	0.1	<i>Arius bilineatus</i>	15.9	10.578 (11.587)	0.339 (0.395)
			<i>Arius thalassinus</i>	8.5	2.826 (3.026)	0.081 (0.091)
			<i>Arius argyropleuron</i>	4.9	2.794 (3.072)	0.081 (0.089)
Bathysauridae	0.4	0.8	<i>Saurida argentea</i>	75.6	13.102 (17.325)	1.207 (1.383)
			<i>Saurida undosquamis</i>	56.1	11.067 (12.666)	0.779 (0.830)
Callionymidae	0.1	<0.1	<i>Repomucenus sublaevis</i>	14.6	3.771 (6.259)	0.072 (0.104)
			<i>Repomucenus belcheri</i>	6.1	0.728 (0.808)	0.010 (0.011)
			<i>Calliurichthys grossi</i>	2.4	0.165 (0.171)	0.005 (0.005)
			<i>Repomucenus meridionalis</i>	1.2	0.281 (0.281)	0.006 (0.006)
Carangidae	9.0	6.4	<i>Caranx bucculentus</i>	96.3	256.911 (459.163)	6.990 (10.707)
			<i>Selaroides leptolepis</i>	56.1	491.471 (490.819)	15.318 (15.296)
			<i>Pantolabus radiatus</i>	43.9	3.407 (5.699)	0.138 (0.277)
			<i>Caranx kleinii</i>	37.8	37.477 (38.697)	1.076 (1.119)
			<i>Scomberoides tol</i>	19.5	0.783 (1.491)	0.040 (0.065)
			<i>Carangoides humerosus</i>	15.9	1.511 (1.710)	0.055 (0.095)
			<i>Carangoides hedlandensis</i>	13.4	0.846 (1.058)	0.097 (0.115)
			<i>Carangoides talamparoides</i>	8.5	0.756 (0.809)	0.014 (0.023)
			<i>Alectis indica</i>	7.3	1.868 (1.873)	0.085 (0.085)
			<i>Alepes apercna</i>	7.3	0.704 (0.925)	0.076 (0.093)
			<i>Carangoides malabaricus</i>	4.9	1.456 (1.535)	0.049 (0.062)
			<i>Selar boops</i>	4.9	2.515 (2.699)	0.054 (0.071)
			<i>Gnathanodon speciosus</i>	3.7	0.948 (1.016)	0.051 (0.062)
			<i>Atule mate</i>	2.4	0.300 (0.302)	0.005 (0.007)
			<i>Selar crumenophthalmus</i>	2.4	1.135 (1.470)	0.097 (0.122)
			<i>Decapterus russelli</i>	2.4	0.437 (0.454)	0.030 (0.030)
<i>Scomberoides commersonianus</i>	1.2	0.437 (0.437)	0.179 (0.179)			
<i>Ulua aurochs</i>	1.2	0.242 (0.242)	0.007 (0.007)			
Clupeidae	6.1	4.2	<i>Sardinella gibbosa</i>	81.7	84.518 (117.892)	2.162 (2.678)
			<i>Pellona ditchela</i>	76.8	258.807 (328.755)	4.356 (6.298)
			<i>Dussumieria elopsoides</i>	56.1	67.902 (83.760)	2.180 (2.688)
			<i>Herklotsichthys lippa</i>	52.4	17.741 (25.501)	0.660 (0.894)
			<i>Anodontostoma chacunda</i>	20.7	30.663 (35.092)	1.650 (1.943)
			<i>Sardinella albella</i>	15.9	41.310 (55.025)	0.962 (1.109)
			<i>Herklotsichthys koningsbergeri</i>	11.0	11.376 (13.188)	0.255 (0.304)
			<i>Ilisha lumula</i>	1.2	3.369 (3.369)	0.193 (0.193)

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Table 22 continued

Family	Family Number (%)	Family Weight (%)	Species Represented Family	per	Species Occurrence (%)	Mean Catch Rate n h^{-1} (\pm S.E)	Mean Catch Rate kg h^{-1} (\pm S.E)
Cynoglossidae*	<0.1	<0.1	<i>Paraplagusia sinerama</i>		6.1	0.728 (0.999)	0.052 (0.068)
			<i>Cynoglossus bilineatus</i>		3.7	0.295 (0.332)	0.058 (0.059)
			<i>Cynoglossidae - undifferentiated</i>		1.2	0.352 (0.352)	0.003 (0.003)
			<i>Paraplagusia bilineata</i>		1.2	0.041 (0.041)	0.002 (0.002)
Drepanidae	0.1	0.1	<i>Drepane punctata</i>		39.0	5.108 (9.093)	0.192 (0.272)
Echeneidae	<0.1	<0.1	<i>Echeneis naucrates</i>		3.7	0.437 (0.552)	0.322 (0.333)
Engraulidae	5.5	3.8	<i>Thryssa setirostris</i>		79.3	249.588 (439.669)	6.710 (9.633)
			<i>Stolephorus indicus</i>		32.9	10.564 (13.899)	0.100 (0.130)
			<i>Thryssa hamiltonii</i>		22.0	17.604 (23.540)	1.283 (1.589)
Ephippidae	<0.1	<0.1	<i>Zabidius novemaculeatus</i>		3.7	1.000 (1.081)	0.044 (0.051)
Exocoetidae	<0.1	<0.1	<i>Parexocoetus mento</i>		2.4	0.198 (0.205)	0.001 (0.001)
Fistulariidae	<0.1	<0.1	<i>Fistularia petimba</i>		12.2	1.029 (1.621)	0.022 (0.032)
Gerreidae	0.4	0.3	<i>Pentaprion longimanus</i>		41.5	20.119 (20.799)	0.386 (0.392)
			<i>Gerres filamentosus</i>		29.3	14.459 (18.571)	0.330 (0.517)
			<i>Gerres subfasciatus</i>		12.2	9.067 (9.650)	0.336 (0.363)
			<i>Gerres oyena</i>		1.2	0.046 (0.046)	0.002 (0.002)
Grammistidae	<0.1	<0.1	<i>Diploprion bifasciatum</i>		1.2	0.426 (0.426)	0.016 (0.016)
Haemulidae	19.9	21.9	<i>Pomadasys maculatus</i>		81.7	1730.246 (2340.16)	78.779 (100.79)
			<i>Pomadasys trifasciatus</i>		40.2	58.957 (87.677)	1.198 (1.895)
			<i>Pomadasys kaakan</i>		7.3	1.626 (1.894)	0.066 (0.080)
			<i>Diagramma labiosum</i>		1.2	0.426 (0.426)	0.039 (0.039)
			<i>Pomadasys argenteus</i>		1.2	0.163 (0.163)	0.004 (0.004)
Harpadontidae	<0.1	<0.1	<i>Harpadon translucens</i>		1.2	0.168 (0.168)	0.033 (0.033)
Hemiramphidae*	<0.1	<0.1	<i>Hemiramphidae - undifferentiated</i>		4.9	1.056 (1.084)	0.006 (0.008)
			<i>Hemiramphus robustus</i>		1.2	0.306 (0.306)	0.003 (0.003)
Labridae	<0.1	<0.1	<i>Choerodon monostigma</i>		2.4	0.481 (0.492)	0.064 (0.064)
			<i>Choerodon cephalotes</i>		2.4	0.611 (0.741)	0.063 (0.078)
Lactariidae	<0.1	0.1	<i>Lactarius lactarius</i>		17.1	2.871 (4.228)	0.121 (0.191)
Leiognathidae	9.6	3.5	<i>Leiognathus moretoniensis</i>		81.7	83.272 (90.581)	1.065 (1.130)
			<i>Leiognathus bindus</i>		72.0	414.017 (586.786)	4.624 (6.417)
			<i>Secutor insidiator</i>		46.3	49.522 (62.338)	0.470 (0.682)
			<i>Leiognathus decorus</i>		34.1	37.981 (44.069)	0.926 (1.052)
			<i>Leiognathus ruconius</i>		32.9	9.444 (13.695)	0.079 (0.117)
			<i>Gazza minuta</i>		32.9	11.486 (21.115)	0.274 (0.501)
			<i>Leiognathus leuciscus</i>		28.0	12.628 (17.237)	0.140 (0.265)
			<i>Leiognathus sp.</i>		26.8	2.881 (4.695)	0.091 (0.131)
			<i>Leiognathus equulus</i>		24.4	16.622 (19.362)	0.391 (0.488)
			<i>Leiognathus splendens</i>		23.2	103.579 (143.627)	3.678 (3.893)
			<i>Leiognathus blochii</i>		3.7	0.295 (0.357)	0.002 (0.002)
			<i>Leiognathus longispinis</i>		1.2	0.046 (0.046)	0.001 (0.001)
			Lethrinidae	<0.1	<0.1	<i>Lethrinus genivittatus</i>	
<i>Lethrinus lentjan</i>		1.2				0.142 (0.142)	0.011 (0.011)
Lutjanidae	<0.1	<0.1	<i>Lutjanus malabaricus</i>		4.9	1.438 (1.494)	0.039 (0.040)
			<i>Lutjanus lutjanus</i>		1.2	0.142 (0.142)	0.008 (0.008)
			<i>Lutjanus vitta</i>		1.2	0.961 (0.961)	0.026 (0.026)
Menidae	0.1	0.1	<i>Mene maculata</i>		13.4	17.569 (18.125)	0.595 (0.607)
Monacanthidae	2.2	0.4	<i>Paramonacanthus choirocephalus</i>		45.1	153.931 (253.026)	1.204 (1.713)
			<i>Monacanthus chinensis</i>		1.2	0.568 (0.568)	0.035 (0.035)
			<i>Chaetodermis penicilligera</i>		1.2	< 0.001	< 0.001
Mugilidae	<0.1	<0.1	<i>Valamugil perusii</i>		12.2	1.957 (2.256)	0.146 (0.174)

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Table 22 continued

Family	Family Number (%)	Family Weight (%)	Species Represented Family	per	Species Occurrence (%)	Mean Catch Rate n h^{-1} (\pm S.E)	Mean Catch Rate kg h^{-1} (\pm S.E)
Mullidae	0.8	0.6	<i>Upeneus sundaicus</i>		40.2	3.306 (4.125)	0.112 (0.150)
			<i>Upeneus sulphureus</i>		31.7	93.872 (113.240)	1.895 (2.352)
			<i>Upeneus asymmetricus</i>		26.8	6.529 (10.597)	0.254 (0.380)
			<i>Upeneus luzonius</i>		1.2	0.284 (0.284)	0.022 (0.022)
			<i>Upeneus sp. 1</i>		1.2	0.611 (0.611)	0.013 (0.013)
Nemipteridae	0.2	0.2	<i>Nemipterus peronii</i>		46.3	5.030 (8.110)	0.257 (0.367)
			<i>Nemipterus hexodon</i>		30.5	3.400 (4.205)	0.105 (0.151)
			<i>Scolopsis taenioptera</i>		13.4	2.446 (3.616)	0.116 (0.170)
			<i>Pentapodus paradiseus</i>		1.2	0.284 (0.284)	0.020 (0.020)
			<i>Pentapodus porosus</i>		1.2	0.481 (0.481)	0.028 (0.028)
Ostraciidae	<0.1	<0.1	<i>Rhynchostracion nasus</i>		4.9	0.240 (0.286)	0.022 (0.022)
Paralichthyidae	<0.1	<0.1	<i>Pseudorhombus arsius</i>		25.6	0.988 (2.148)	0.051 (0.106)
			<i>Pseudorhombus spinosus</i>		6.1	0.349 (0.527)	0.014 (0.022)
			<i>Pseudorhombus elevatus</i>		2.4	0.367 (0.370)	0.007 (0.008)
Pegasidae	0.0	<0.1	<i>Pegasus volitans</i>		13.4	0.933 (1.649)	0.004 (0.007)
Pinguipedidae	<0.1	<0.1	<i>Parapercis nebulosa</i>		6.1	0.593 (0.822)	0.021 (0.028)
Platycephalidae	0.1	0.1	<i>Inegocia japonica</i>		29.3	6.622 (9.023)	0.131 (0.183)
			<i>Elates ransonnetii</i>		15.9	3.855 (5.260)	0.033 (0.046)
			<i>Suggrundus macracanthus</i>		1.2	0.055 (0.055)	0.001 (0.001)
			<i>Platycephalus indicus</i>		1.2	0.728 (0.728)	0.129 (0.129)
Polynemidae	36.3	42.0	<i>Polydactylus multiradiatus</i>		100.0	2563.237 (2775.11)	97.835 (107.37)
Pomacanthidae	<0.1	<0.1	<i>Chaetodontoplus duboulayi</i>		2.4	0.481 (0.555)	0.077 (0.078)
			<i>Pristotis obtusirostris</i>		2.4	0.481 (0.555)	0.015 (0.017)
			<i>Priacanthus tayenus</i>		17.1	0.781 (1.480)	0.085 (0.104)
Priacanthidae	<0.1	<0.1	<i>Priacanthus tayenus</i>		17.1	0.781 (1.480)	0.085 (0.104)
Psettodidae	<0.1	<0.1	<i>Psettodes erumei</i>		11.0	0.446 (0.716)	0.065 (0.083)
Sciaenidae	5.3	5.1	<i>Johnius borneensis</i>		87.8	189.904 (357.034)	5.806 (11.736)
			<i>Johnius amblycephalus</i>		42.7	9.049 (15.005)	0.411 (0.696)
			<i>Austronibeia oedogenys</i>		6.1	0.856 (0.957)	0.150 (0.153)
			<i>Otolithes ruber</i>		4.9	3.424 (3.858)	0.521 (0.572)
			<i>Atrobucca brevis</i>		1.2	2.109 (2.109)	0.036 (0.036)
Scombridae	0.1	0.4	<i>Scomberomorus queenslandicus</i>		35.4	3.130 (4.111)	0.601 (0.867)
			<i>Rastrelliger kanagurta</i>		8.5	0.349 (0.648)	0.044 (0.048)
			<i>Scomberomorus commerson</i>		3.7	1.085 (1.150)	0.885 (1.010)
Serranidae	<0.1	<0.1	<i>Epinephelus sexfasciatus</i>		4.9	0.163 (0.200)	0.005 (0.007)
			<i>Centrogenys vaigiensis</i>		1.2	0.481 (0.481)	0.008 (0.008)
Siganidae	<0.1	<0.1	<i>Siganus canaliculatus</i>		1.2	0.852 (0.852)	0.031 (0.031)
Sillaginidae	0.3	0.3	<i>Sillago lutea</i>		37.8	19.267 (20.949)	0.665 (0.703)
			<i>Sillago burrus</i>		14.6	7.729 (8.385)	0.291 (0.352)
			<i>Sillago ingenuua</i>		14.6	17.103 (21.002)	0.603 (0.722)
Soleidae	<0.1	<0.1	<i>Aseraggodes melanostictus</i>		6.1	0.948 (0.994)	0.015 (0.016)
			<i>Brachirus muelleri</i>		2.4	0.349 (0.409)	0.012 (0.014)
			<i>Zebrias quagga</i>		1.2	0.240 (0.240)	0.005 (0.005)
			<i>Sphyrna obtusata</i>		4.9	2.871 (3.136)	0.188 (0.216)
Sphyrnaeidae	<0.1	0.1	<i>Sphyrna obtusata</i>		4.9	2.871 (3.136)	0.188 (0.216)
			<i>Sphyrna putnamae</i>		4.9	1.085 (1.219)	0.135 (0.145)
Syngnathidae*	<0.1	<0.1	<i>Syngnathidae - undifferentiated</i>		1.2	< 0.001	< 0.001
			<i>Trachyrhamphus longirostris</i>		1.2	< 0.001	< 0.001
Synodontidae	<0.1	<0.1	<i>Trachinocephalus myops</i>		1.2	0.352 (0.352)	0.012 (0.012)
Terapontidae	2.1	0.9	<i>Terapon theraps</i>		64.6	97.665 (146.415)	0.854 (1.601)
			<i>Terapon jarbua</i>		20.7	14.733 (15.374)	0.200 (0.248)
			<i>Pelates quadrilineatus</i>		17.1	3.424 (5.044)	0.177 (0.265)

Objective 4: Bycatch descriptions of banana prawn fisheries

Table 22 continued

Family	Family Number (%)	Family Weight (%)	Species Represented Family	per	Species Occurrence (%)	Mean Catch Rate n h^{-1} (\pm S.E)	Mean Catch Rate $kg h^{-1}$ (\pm S.E)
			<i>Terapon puta</i>		8.5	4.280 (4.402)	0.157 (0.159)
			<i>Lagocephalus sceleratus</i>		24.4	1.185 (1.679)	0.029 (0.048)
			<i>Lagocephalus lunaris</i>		18.3	20.067 (28.106)	1.198 (1.663)
			<i>Torquigener whiteleyi</i>		9.8	0.405 (0.678)	0.012 (0.017)
			<i>Chelonodon patoca</i>		3.7	0.565 (0.714)	0.039 (0.041)
			<i>Lagocephalus inermis</i>		2.4	2.515 (2.747)	0.245 (0.255)
			<i>Torquigener pallimaculatus</i>		2.4	0.163 (0.196)	0.007 (0.009)
Tetrarogidae	<0.1	<0.1	<i>Liocranium praepositum</i>		1.2	0.142 (0.142)	0.003 (0.003)
Triacanthidae	0.1	<0.1	<i>Trixipichthys weberi</i>		45.1	3.096 (5.805)	0.072 (0.121)
			<i>Tripodichthys angustifrons</i>		4.9	0.295 (0.344)	0.014 (0.015)
Trichiuridae	0.1	0.2	<i>Trichiurus lepturus</i>		29.3	5.743 (11.589)	0.287 (0.500)
ELASMOBRANCHS (n = 78 samples, except Pristidae n = 93)							
Carcharhinidae	0.5	5.7	<i>Rhizoprionodon acutus</i>		73.1	29.661 (36.974)	5.742 (7.549)
			<i>Carcharhinus tilstoni</i>		42.3	2.102 (2.710)	4.336 (6.529)
			<i>Carcharhinus sorrah</i>		1.3	1.194 (1.194)	5.074 (5.074)
			<i>Carcharhinus dussumieri</i>		5.1	1.051 (1.054)	3.285 (3.294)
Dasyatidae	<0.1	1.6	<i>Himantura toshi</i>		42.3	1.194 (1.667)	5.970 (6.937)
			<i>Dasyatis leylandi</i>		3.8	0.405 (0.415)	0.078 (0.080)
			<i>Dasyatis annotata</i>		3.8	0.337 (0.399)	0.058 (0.082)
			<i>Dasyatis kuhlii</i>		1.3	0.021 (0.021)	0.020 (0.020)
Gymnuridae	<0.1	0.1	<i>Gymnura australis</i>		12.8	0.601 (0.617)	0.206 (0.316)
Hemigaleidae	<0.1	<0.1	<i>Hemipristis elongata</i>		1.3	0.097 (0.097)	0.057 (0.057)
			<i>Hemigaleus australiensis</i>		1.3	0.073 (0.073)	0.038 (0.038)
Hemiscylliidae	<0.1	<0.1	<i>Chiloscyllium punctatum</i>		1.3	0.059 (0.059)	0.007 (0.007)
Myliobatidae	<0.1	0.1	<i>Aetomylaeus nichofii</i>		5.1	0.118 (0.141)	0.257 (0.261)
			<i>Aetobatus narinari</i>		1.3	0.096 (0.096)	0.529 (0.529)
Pristidae	<0.1	3.2	<i>Anoxypristis cuspidata</i>		11.8	0.161 (0.290)	11.470 (16.755)
Rhynchobatidae	<0.1	<0.1	<i>Rhynchobatus australiae</i>		2.6	0.059 (0.073)	0.022 (0.027)
Sphyrnidae	<0.1	<0.1	<i>Sphyrna lewini</i>		1.3	0.193 (0.193)	0.056 (0.056)
			<i>Eusphyra blochii</i>		3.8	0.085 (0.092)	0.075 (0.084)
REPTILES (Sea snakes, n = 93 samples)							
Hydrophiidae	<0.1	0.1	<i>Lapemis hardwickii</i>		17.2	0.258 (0.347)	0.093 (0.135)
			<i>Hydrophis elegans</i>		4.3	0.054 (0.094)	0.047 (0.082)
			<i>Hydrophis ornatus</i>		3.2	0.038 (0.052)	0.019 (0.023)
			<i>Enhydrina schistosa</i>		1.1	0.022 (0.022)	0.013 (0.013)
			<i>Hydrophis caeruleus</i>		1.1	0.059 (0.059)	0.016 (0.016)
			<i>Disteira major</i>		1.1	0.092 (0.092)	0.071 (0.071)
			<i>Disteira kingii</i>		1.1	0.059 (0.059)	0.034 (0.034)
INVERTEBRATES (n = 82 samples)							
Astropectinidae*	<0.1	<0.1	<i>Astropectinidae - undifferentiated</i>		2.4	0.129 (0.140)	0.008 (0.008)
Dorippidae	<0.1	<0.1	<i>Paradorippe australiensis</i>		1.2	0.067 (0.067)	< 0.001
Goniasteridae*	<0.1	<0.1	<i>Stellaster sp.</i>		3.7	0.064 (0.096)	0.003 (0.003)
Leucosiidae	<0.1	<0.1	<i>Leucosia whitei</i>		1.2	0.046 (0.046)	< 0.001

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Table 22 continued

Family	Family Number (%)	Family Weight (%)	Species Represented Family	per	Species Occurrence (%)	Mean Catch Rate n h^{-1} (\pm S.E)	Mean Catch Rate kg h^{-1} (\pm S.E)
Loliginidae*	<0.1	<0.1	<i>Loliginidae - undifferentiated</i>		34.1	1.430 (2.898)	0.083 (0.152)
Luidiidae*	<0.1	<0.1	<i>Luidiidae - undifferentiated</i>		1.2	0.129 (0.129)	0.024 (0.024)
Majidae	<0.1	<0.1	<i>Phalangipus longipes</i>		1.2	0.134 (0.134)	< 0.001
Matutidae	<0.1	<0.1	<i>Ashtoret granulosa</i>		11.0	8.202 (8.250)	0.210 (0.212)
			<i>Matuta victor</i>		1.2	4.101 (4.101)	0.034 (0.034)
			<i>Izanami inermis</i>		2.4	0.265 (0.357)	0.008 (0.009)
Parthenopidae*	<0.1	<0.1	<i>Parthenopidae - undifferentiated</i>		1.2	0.163 (0.163)	0.001 (0.001)
Pectinidae	<0.1	<0.1	<i>Amusium pleuronectes</i>		17.1	3.022 (4.079)	0.041 (0.056)
Penaeidae*	<0.1	<0.1	<i>Trachypenaeus spp.</i>		2.4	0.728 (0.731)	0.001 (0.001)
			<i>Trachypenaeus anchoralis</i>		6.1	0.405 (0.565)	0.001 (0.001)
			<i>Trachypenaeus curvirostris</i>		1.2	0.074 (0.074)	< 0.001
			<i>Metapenaeopsis spp.</i>		1.2	0.074 (0.074)	< 0.001
			<i>Parapenaeopsis cornuta</i>		1.2	0.056 (0.056)	< 0.001
Pennatulidae*	<0.1	<0.1	<i>Pennatulidae - undifferentiated</i>		1.2	0.435 (0.435)	0.004 (0.004)
Pilumnidae	1.0	<0.1	<i>Galene bispinosa</i>		1.2	0.074 (0.074)	< 0.001
Portunidae	0.1	0.1	<i>Portunus rubromarginatus</i>		12.2	2.012 (2.239)	0.035 (0.041)
			<i>Charybdis callianassa</i>		24.4	1.769 (2.744)	0.014 (0.026)
			<i>Portunus pelagicus</i>		31.7	1.277 (2.512)	0.205 (0.357)
			<i>Portunus sanguinolentus</i>		4.9	1.133 (1.173)	0.068 (0.069)
			<i>Charybdis jaubertensis</i>		2.4	0.728 (0.740)	0.012 (0.012)
			<i>Charybdis feriata</i>		2.4	0.378 (0.388)	0.032 (0.034)
			<i>Portunus gracilimanus</i>		7.3	0.352 (0.451)	0.004 (0.005)
			<i>Podophthalmus vigil</i>		1.2	0.074 (0.074)	0.001 (0.001)
			<i>Thenus indicus</i>		31.7	1.955 (3.180)	0.135 (0.232)
Scyllaridae	<0.1	0.1	<i>Petrarctus demani</i>		1.2	0.349 (0.349)	0.001 (0.001)
			<i>Sepia smithi</i>		3.7	0.349 (0.363)	0.059 (0.059)
Sepiidae	<0.1	<0.1	<i>Sepia elliptica</i>		11.0	0.208 (0.329)	0.006 (0.008)
			<i>Spatangidae - undifferentiated</i>		1.2	0.137 (0.137)	0.001 (0.001)
Spatangidae*	1.0	<0.1	<i>Sphenopus marsupialis</i>		2.4	0.728 (0.764)	0.010 (0.011)
Squillidae	<0.1	<0.1	<i>Oratosquilla gravieri</i>		9.8	0.728 (1.056)	0.006 (0.010)
			<i>Erugosquilla woodmasoni</i>		2.4	0.352 (0.375)	0.009 (0.010)
			<i>Miyakea nepa</i>		1.2	0.046 (0.046)	0.002 (0.002)
			<i>Harpisquilla stephensoni</i>		1.2	0.025 (0.025)	0.002 (0.002)
Phylum Bryozoa	<0.1	<0.1	<i>Bryozoa - undifferentiated</i>		3.7		0.185 (0.185)
Phylum Porifera	<0.1	<0.1	<i>Porifera - undifferentiated</i>		1.2		0.005 (0.005)
Class Hydrozoa	<0.1	<0.1	<i>Gorgonacea-sea fan</i>		1.2	0.041 (0.041)	0.003 (0.003)
Class							
Ophiuroidea	<0.1	<0.1	<i>Ophiuroidea sp. 1</i>		2.4	0.503 (0.590)	0.001 (0.001)
Class Scyphozoa	<0.1	<0.1	<i>Scyphozoa spp - undifferentiated</i>		3.7	0.391 (0.419)	0.010 (0.011)
Order Hydroida	<0.1	<0.1	<i>Order Hydroida - undifferentiated</i>		12.2	0.041 (0.041)	0.006 (0.010)

* Families where not all specimens could be identified to species level

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Of the threatened, endangered and protected species, no marine turtles (Cheloniidae) were captured. A total of 39 sea snakes (Hydrophiidae) were captured (Table 22); 25 of those were *Lapemis hardwickii*. Of the Syngnathidae, two species of pipefish were caught; one specimen of *Trachyrhamphus longirostris* and one specimen of an unidentified species (Table 22). Sixteen of one sawfish species *Anoxypristis cuspidata* (Pristidae) were caught (length range 251 mm to 2800 mm).

One teleost species, *Polydactylus multiradiatus* (Polynemidae), contributed 42.8% to the total bycatch weight and occurred in 100% of trawls at a mean catch rate of 97.8 kg h⁻¹ (Table 22). Eleven species (10 teleosts and 1 elasmobranch) occurred in greater than two thirds of trawls (Table 22), and 33 species (29 teleost, 3 elasmobranch and 1 invertebrate species) occurred in $\geq 33\%$ of trawls. A total of 185 species occurred in $< 33\%$ of trawls.

The SIMPER analysis indicated which species contributed the most consistently to the banana prawn fishery bycatch weight. These results revealed that just over three quarters of the similarity among banana prawn fishery bycatch was accounted for by three species: *P. multiradiatus* (65.7% of the similarity), *Caranx bucculentus* (7.8% of the similarity) and *Rhizoprionodon acutus* (3.9% of the similarity). Together, these three species accounted for just over half (51.4%) of the total bycatch weight. While there were 218 bycatch taxa overall, 90% of the similarity among banana prawn fishery bycatch was accounted for by only nine species. Together those nine species accounted for 78.7% of the total bycatch weight—they were (in descending order of similarity): *P. multiradiatus*, *C. bucculentus*, *R. acutus*, *Johnius borneensis*, *Sardinella gibbosa*, *Pomadasys maculatus*, *Thryssa setirostris*, *Carcharhinus tilstoni* and *Leiognathus bindus*.

The species contributions by abundance were comparable, with three quarters of the banana fishery bycatch accounted for by *P. multiradiatus*, *C. bucculentus* and *L. bindus*. As before, 90% of the similarity was accounted for by almost the same nine species, the exception being two small numerous teleosts (*Terapon theraps* and *Leiognathus moretoniensis*) in place of two heavier and less numerous elasmobranchs (*R. acutus* and *C. tilstoni*).

Fishing behaviour in the banana prawn fishery

During the banana prawn fishery, the fishers' primary objective was to locate and target—trawl large schools of *Penaeus merguensis* ('prawn school' trawls). This type of fishing behaviour occurred 75% of the time, with 'fish school' trawling 19.2% and 'dispersed school' trawling 5.8%.

The Kruskal–Wallis tests detected overall significant differences between factors (fishing behaviour) in the banana prawn fishery. At least one of the types of fishing behaviour was significantly different for: the number of species per trawl ($p = 0.03$); the percentage bycatch of the total catch per trawl ($p < 0.001$); and the bycatch catch rate per trawl ($p = 0.02$) (Figure 45). There was no significant difference between the types of fishing behaviour for the total bycatch weight per trawl.

The Wilcoxon tests determined which of the bycatch catch characteristics were significantly different between the three fishing behaviours. For the number of bycatch species per trawl (Figure 45a), a significant difference ($p = 0.03$) was detected between 'prawn school' (mean 30.8 species \pm 1.8 S.E) and 'dispersed prawn' trawls; with more bycatch species present in 'dispersed prawn' trawls (mean 46.0 species \pm 6.0 S.E). For the percentage of bycatch (by weight) of the total catch per trawl (Figure 45b), the Wilcoxon tests determined there were

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significant differences among all fishing behaviour types ($p < 0.01$ for all comparisons). The greatest percentage of bycatch occurred for 'fish school' trawls (mean $83.3\% \pm 2.9$ S.E), followed by 'dispersed prawn' trawls (mean $59.4\% \pm 8.9$ S.E) and 'prawn school' trawls (mean $32\% \pm 1.9$ S.E). For the bycatch catch rates per trawl (Figure 45d) the Wilcoxon tests determined there was a significant difference ($p = 0.006$) between 'prawn school' (mean 970 kg $h^{-1} \pm 197.2$ S.E) and 'fish school' trawls; with 'fish school' trawls having the greater catch rate (mean $2,137.8$ kg $h^{-1} \pm 683.7$ S.E).

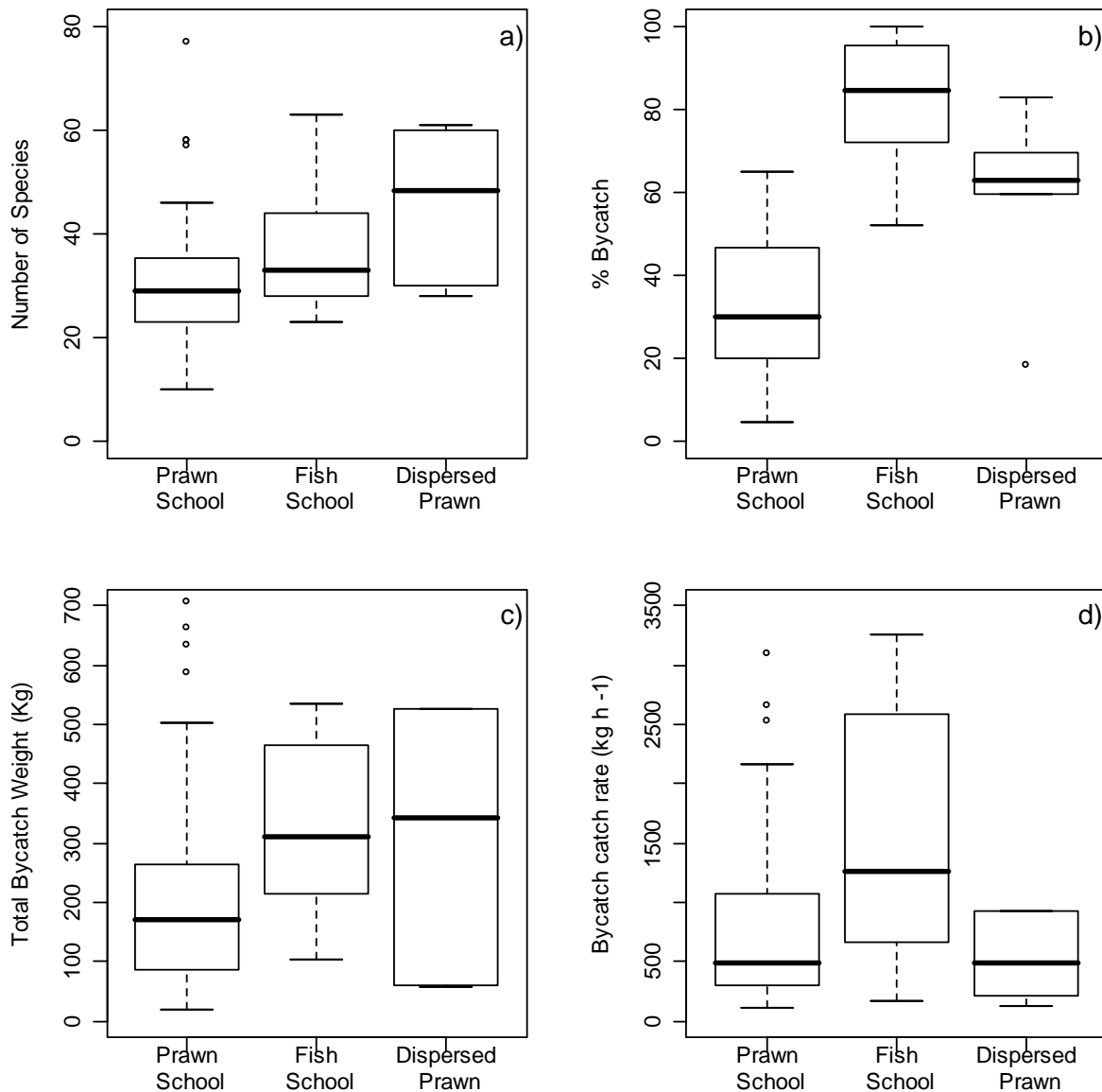


Figure 45. Box plots showing the statistical distribution (median, quartiles, upper and lower 5th percentiles) of banana prawn trawl bycatch for each type of fishing behaviour (x axis) regarding: (a) the number of bycatch species per trawl, (b) the percentage bycatch (by weight) of the total catch per trawl, (c) the total bycatch weight per trawl (kg) and (d) the bycatch catch rate per trawl (kg h^{-1}).

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Analysis of banana and tiger prawn fisheries

Both prawn trawl fisheries had characteristics that were particular to the species being targeted (Table 23). The banana prawn fishery had trawls with a higher mean catch rate of bycatch but less species per trawl than the tiger prawn fishery (Figure 46). The banana prawn fishery also had a lower estimated total bycatch ($1,502.1 \text{ t yr}^{-1} \pm 288.4 \text{ S.E}$) compared to the tiger prawn fishery estimated total bycatch of $20,073.4 \text{ kg yr}^{-1} \pm 568.1 \text{ S.E}$ (mean values for the period 2001–2005).

Teleosts dominated the bycatch of both penaeid fisheries (Figure 47). While the banana prawn trawl bycatch contained fewer teleost species, it had a greater proportion of fish and elasmobranchs in the bycatch. The tiger prawn fishery had a greater proportion of invertebrates and more invertebrate species than the banana prawn fishery (Figure 47). Less than 2% sea snakes (mean n per trawl) were caught by either fishery (Figure 47).

The nMDS ordinations revealed differences in the bycatch assemblage structure of the two fisheries, both in terms of weight and numerical abundance (Figure 48). ANOSIM confirmed that these differences were statistically significant for both weight ($R = 0.690$; $p < 0.0001$) and abundance ($R = 0.679$; $p < 0.0001$). SIMPER indicated that these differences were primarily due to a few bycatch species that made large contributions to the weight (Table 24) and abundance (Table 25) of each fishery; with the rest of the species making small contributions ($< 2\%$ dissimilarity). The majority of species captured only contributed small percentages to the total weight or abundance for either fishery, particularly for the banana prawn fishery (Figure 49). For example, for the species weight, five species accounted for $\sim 50\%$ of the dissimilarity (particularly *Polydactylus multiradiatus*) between the banana and tiger prawn fishery bycatch samples (Table 24). Likewise, in terms of abundance (Table 25), four species contributed $\sim 50\%$ to the dissimilarity between samples from the two fisheries (particularly *P. multiradiatus*). The species contributing more than 50% of the dissimilarity between samples were all caught in higher weight or abundance in the banana prawn trawl fishery.

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Table 23. Characteristics of the banana and tiger prawn trawl fisheries in the Gulf of Carpentaria. Data are from the tiger prawn fishery study (Stobutzki et al., 2001 *a*) and the present study.

Fishery Characteristics	Banana Trawl Fishery	Tiger Trawl Fishery
Fishing Period (GOC)	24 Hour trawling	"Night" trawling
Mean Depth	17.1m (± 0.5 S.E)	31.6 m (± 0.8 S.E)
Percentage Day Trawl	80.8%	0%
Percentage Dusk & Dawn Collectively	10.6%	12%
Percentage Night Trawl	8.7%	88%
Mean Trawl Duration; 'Bottom Time'	18.9 Minutes (± 1.1 S.E)	3-4 Hours*

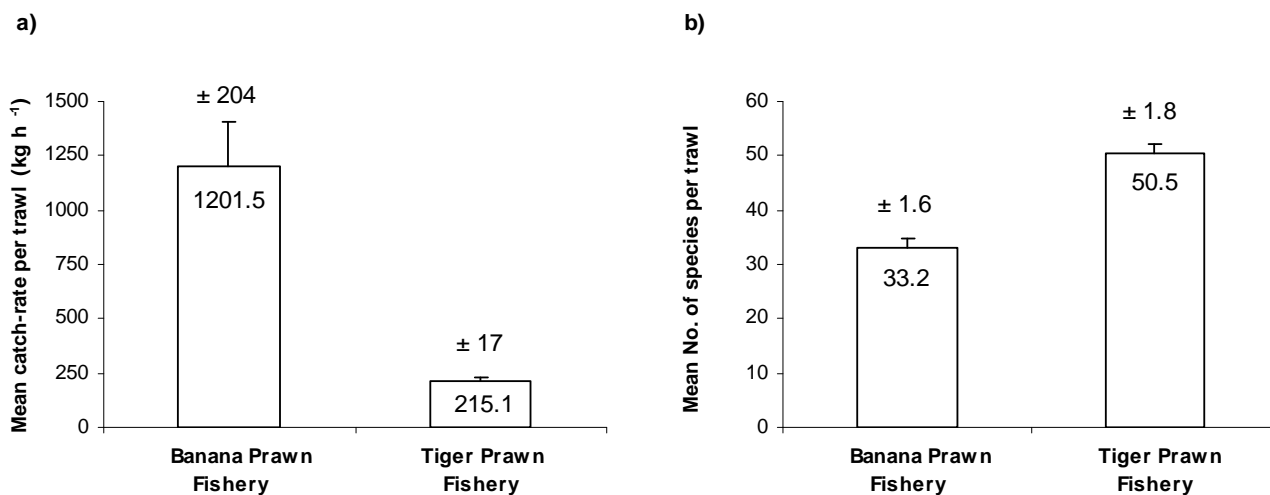


Figure 46. The (a) mean bycatch catch rates (\pm S.E), and (b) mean number of bycatch species (\pm S.E) caught by the two penaeid fisheries

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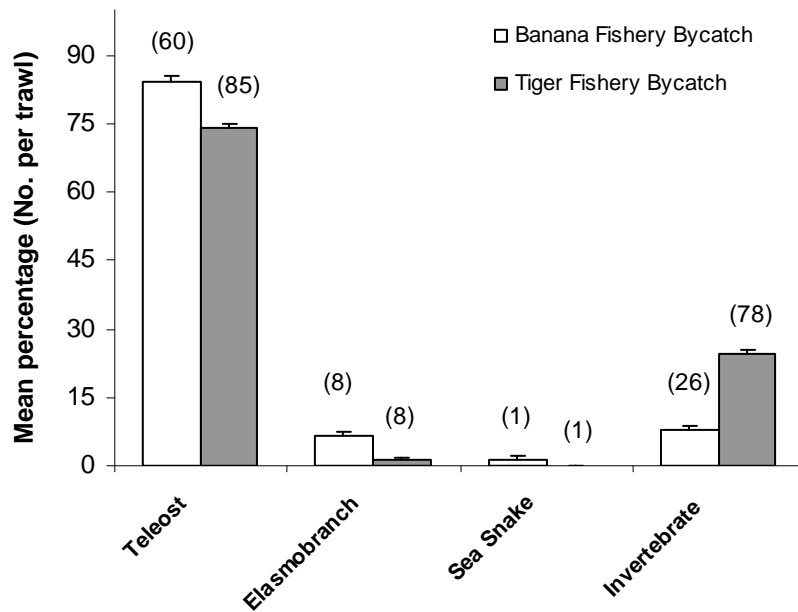


Figure 47. Mean percentage contributions (\pm S.E) of the main groups of bycatch in the banana and tiger prawn trawl fisheries. The number of families in each group is shown in brackets.

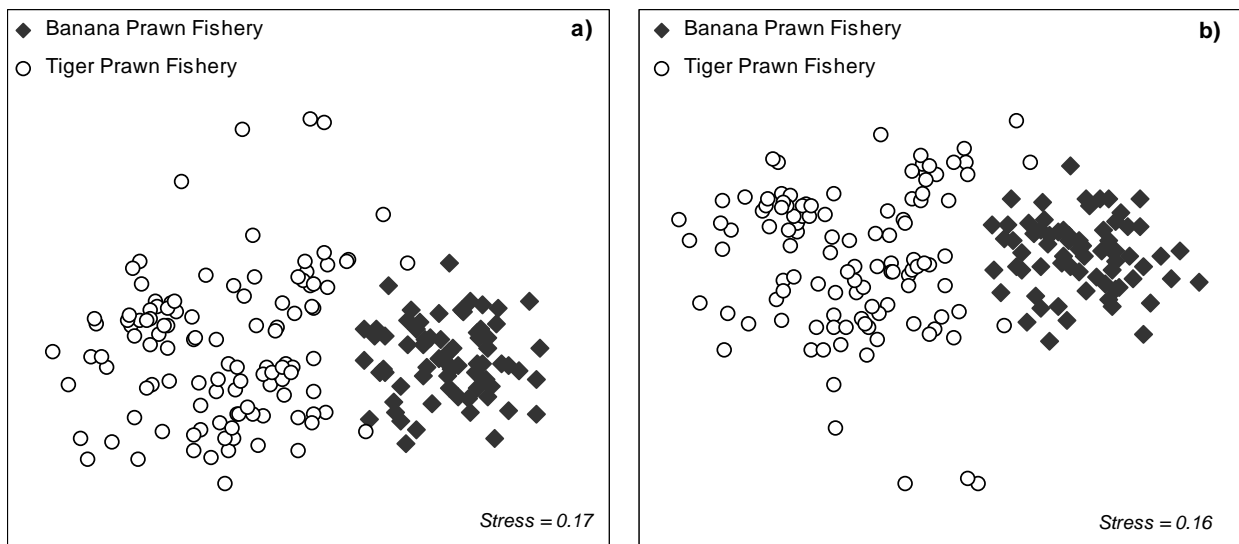


Figure 48. Non-metric multidimensional scaling (nMDS) ordination plots comparing bycatch assemblage structures in trawl catches from the two penaeid fisheries based on: (a) weight and (b) numerical abundance.

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Table 24. SIMPER results for contributions to the total mean dissimilarity between the banana and tiger prawn trawl fisheries based on species weight. Species likely to be the more consistent discriminators between the fisheries are indicated by an asterisk (higher ratio of mean contribution to standard deviation). The fishery contributing the greater relative weight is indicated as A: Banana prawn fishery. The table is limited to species contributing > 2% to the dissimilarity.

Species	Taxonomic Group	Greatest Contribution	Contribution %	Cumulative %
<i>Polydactylus multiradiatus</i> *	Teleost	A	32.2	32.2
<i>Pomadasys maculatus</i> *	Teleost	A	8.6	40.8
<i>Caranx bucculentus</i> *	Teleost	A	4.3	45.1
<i>Rhizoprionodon acutus</i> *	Elasmobranch	A	3.4	48.5
<i>Johnius borneensis</i> *	Teleost	A	2.9	51.4
<i>Anoxypristis cuspidata</i>	Elasmobranch	A	2.5	53.9
<i>Thryssa setirostris</i>	Teleost	A	2.5	56.3
<i>Leiognathus bindus</i>	Teleost	A	2.3	58.6
<i>Carcharhinus tilstoni</i> *	Elasmobranch	A	2.1	60.7

Table 25. SIMPER results for contributions to the total mean dissimilarity between the banana and tiger prawn trawl fisheries based on species numerical abundance. Species likely to be the more consistent discriminators between the fisheries are indicated by an asterisk (higher ratio of mean contribution to standard deviation). The fishery contributing the greater relative abundance is indicated as A: Banana prawn fishery or S: Tiger prawn fishery. The table is limited to species contributing > 2% to the dissimilarity.

Species	Taxonomic Group	Greatest Contribution	Contribution %	Cumulative %
<i>Polydactylus multiradiatus</i> *	Teleost	A	28.7	28.7
<i>Pomadasys maculatus</i> *	Teleost	A	8.9	37.7
<i>Caranx bucculentus</i> *	Teleost	A	6.2	43.8
<i>Leiognathus bindus</i> *	Teleost	A	5.7	49.5
<i>Thryssa setirostris</i>	Teleost	A	3.4	52.9
<i>Johnius borneensis</i> *	Teleost	A	3.2	56.2
<i>Terapon theraps</i> *	Teleost	A	3.0	59.1
<i>Leiognathus splendens</i>	Teleost	S	2.3	61.5
Loveniidae - undifferentiated	Invertebrate	S	2.1	63.5

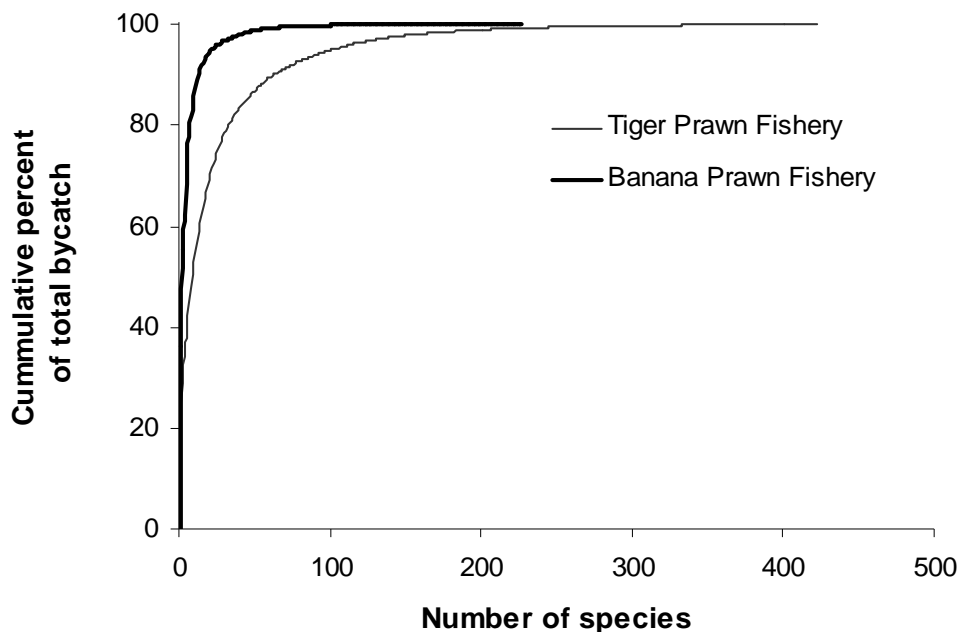


Figure 49. The cumulative percentage of the total bycatch abundance accounted for by species, for each fishery (the same trend was evident for bycatch weight).

Discussion

Bycatch of the banana prawn fishery

This study provides the first description of the bycatch from banana prawn trawling in the Northern Prawn Fishery (NPF). The bycatch of this fishery made up nearly half of the total catch per trawl, with three teleost species dominating the bycatch. No invertebrates were abundant and overall there was a predominance of many rare species contributing only small percentages to the overall weight or numbers. These species composition patterns have been recorded in other similar tropical penaeid trawl fisheries (e.g. Evans and Wahju, 1996; Ye et al., 2000). These trends are also in accordance with a general pattern common to tropical prawn (shrimp) fisheries where small teleosts make up the majority of bycatch, a small number of species dominate the bycatch and there are common teleost families abundant for many of the world's prawn fisheries (Andrew and Pepperell, 1992).

What differs for the *Penaeus merguensis* fishery in Australia is that the mean bycatch to prawn ratio (0.8:1) is low relative to other trawl fisheries. For prawn trawling in general the assumed ratios, of 5:1 for temperate–subtropical waters and 10:1 in tropical waters (Slavin, 1982), have often been quoted or used as reasonable estimates (see review by Andrew and Pepperell, 1992). The ratios of 8:1 to 21:1 were reported for the NPF tiger prawn trawl season (Pender et al., 1992), while in Gulf of Mexico shrimp trawl fishery the discarded bycatch to landed catch ratio was 4.56 and in the South Atlantic Bight was 2.95 (Harrington et al., 2005).

The bycatch to prawn ratio for the banana prawn trawl fishery explains the estimated total bycatch—which could be considered low relative to other prawn trawling operations worldwide. This also accounts for the perception among fishers that there is little bycatch in this fishery compared to other penaeid trawl operations (in addition to being able to fish aggregations of *P. merguensis*). However, because of the mean percentage of bycatch per

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trawl, and effort applied in this fishery, the total bycatch is not inconsequential for this area when also considering the impact from the other penaeid trawl component of the NPF. Hence, for fisheries management to properly address bycatch and sustainability objectives, it is important to account for more than the obvious or larger fisheries. The sum of all fishing impacts and mortality sources must be considered.

For the threatened, endangered and protected (TEP) species component of the banana prawn fishery, the most captures were for some sea snakes and sawfish ($\sim 0.6 \text{ kg h}^{-1}$). No marine turtles were captured. This was largely attributed to the use of Turtle Exclusion Devices (TEDs) which have been compulsory in the NPF since 2000 and have reduced catches of turtles by 99% (Brewer et al., 2006 *b*). Catches of syngnathids were rare ($< 0.1\%$ by number); as similarly reported by Stobutzki et al. (2001 *a*) for the tiger prawn fishery. However, because of their listed status, it is recommended that TEP species be included in monitoring programs to assess their longer term sustainability—which is still unknown for whole bycatch groups, such as elasmobranchs, and increasingly important in fisheries management (Griffiths et al., 2006).

Variation in banana prawn fishing behaviour

In the banana prawn fishery bycatch, the number of species, percentage of bycatch caught per trawl and the rate at which bycatch is caught can vary depending on the fishing behaviour. When banana prawns are abundant, fishers use their echo-sounders to target aggregations, which is why the percentage of bycatch in ‘prawn school’ trawls was significantly lower than the other types of fishing behaviour. The abundance of *P. merguensis* in aggregations is also why this fishing behaviour dominated over others.

There was little difference between the amounts of bycatch caught in the ‘fish school’ and ‘dispersed prawn’ trawls. But there were significant differences for the bycatch catch rate and percentage of bycatch. This is because ‘fish school’ trawls usually catch large quantities of schooling bycatch species in a short period of time, as fishers either misinterpret their echo-sounder signal or deployed trawls near fish schools because of a known or perceived association between some fish and prawn schools. The prawn catch in this instance can range from 0% upward. Furthermore, in the instance of a ‘dispersed prawn’ trawl, there is a deliberate effort by fishers to capture dispersed groups of *P. merguensis* which can result in a greater proportion of the target species.

Throughout the banana prawn season fishers decide where to trawl after weighing up whether there is any benefit in staying in a particular area (e.g. yield from trawls, operating expenses or searching time). Die and Ellis (1999) studied the 1991–92 aggregations of *P. merguensis* in the Gulf of Carpentaria banana prawn fishery and reported that the number of aggregations decreased by 83% in the first three weeks of the season; the average biomass of an aggregation decreasing even more rapidly than the number of aggregations. As the large schools of banana prawn become difficult to locate, are in smaller and dispersed aggregations or are declining toward the end of the season, the likelihood of ‘fish school’ and ‘dispersed prawn’ trawls (and potential bycatch) can arise—though there can also be a dramatic increase in search time with no or minimal trawling. These factors affect the degree of impact on bycatch throughout the season and consequently, the strategies required for monitoring bycatch in a variable fishing environment. While this study was conducted during the peek period where the majority of effort was applied, there is still a need for fisheries researchers and management to account for bycatch impacts from all periods of a fishery.

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Contrasting two related fisheries

While both penaeid–trawl fisheries in the NPF are managed separately for the target species, the bycatch components are not. This is because it was previously assumed that the banana prawn fishery caught little bycatch due to the targeting of schooling prawns with short–duration trawls (Stobutzki et al., 2001 *a*). Hence, concern about impacts on bycatch in this region has been focused on the tiger prawn fishery (Brewer et al., 2006 *b*) with its longer trawl durations and longer season (~12 weeks), greater total fishing effort (Somers 1985) and total bycatch than the shorter banana prawn fishery (~4–6 weeks). There is also considerable difficulty in replicating the banana prawn fishery operations for scientific survey. The present study overcomes these challenges and shows that the banana prawn fishery bycatch is not insignificant, that the fisheries and bycatch assemblage structures differ and therefore warrant a new management approach.

Fishers in the NPF tailor their operations according to the different behaviour of the target species (Robins and Sachse, 1994); including the different day or night emergence behaviour of the penaeids fished (Wassenberg and Hill, 1994). The differences detected between the bycatch assemblage structure of the banana prawn (mainly day time) and tiger prawn (night time) fisheries were likely to be influenced by diel differences in the vertical migratory behaviour of many species in the water column. In the Gulf of Carpentaria, Blaber et al. (1994) noted that the catch rates of bycatch were higher in day trawls than in night trawls; they attributed this to the vertical feeding migration of some demersal species, which influenced their susceptibility to trawling (Blaber et al., 1990). Diel differences in behaviour of the same and similar teleost species have also been recorded in the NW Red Sea trawl fishery (Yousif, 2003). The nocturnal activity and daytime burrowing of some invertebrates will also influence their catchability. Some crustaceans (that occur in NPF bycatch) have been shown to exhibit this behaviour; such as stomatopods (Froglia and Giannini, 1989; Dell and Sumpton, 1999), and both the commercially important Scyllaridae (*Thenus* spp.) (Jones, 1988) and portunid crabs (*Portunus pelagicus*) (Sumpton and Smith, 1990).

Overall the banana prawn fishery invertebrate catch comprised a much smaller contribution to the bycatch than in the tiger prawn trawl fishery. In addition to the emergence behaviour of some crustacean species, the susceptibility of benthic bycatch invertebrates to trawling may also be influenced by differences in the fishing gear. Compared to the tiger prawn nets, the banana prawn nets make lighter contact with the seabed (pers comm. Tony LaMacchia, NPF fisher and net maker). Die and Ellis (1999) noted that banana prawn nets fish ‘light’ because there is a larger vertical opening in the net, with little drag on the seabed, which is achieved by using floats on the headrope with far more meshes in the sides of the net and long flywires (Figure 44). Hence, the combined effects of lighter contact with the seabed, and shorter duration trawls over a smaller area of seabed, are likely to explain some of the differences in bycatch assemblage structure.

Variations in depth and sediment type, as a result of the different operations of the two fisheries, are also likely to influence the bycatch assemblages. In relation to the banana prawn fishery, the tiger prawn fishery operates further offshore and generally in deeper waters. Somers (1987) demonstrated the importance of depth and sediment type for the distribution of commercial penaeids. The distribution of some bycatch species is affected in the same way. For example, in the southeastern Gulf of Carpentaria, species distribution patterns of teleosts (many of which occur in NPF bycatch) were influenced by both depth and sediment type (Rainer and Munro, 1982). Similarly, Ramm et al. (1990) also found the NPF bycatch teleosts in the western Gulf of Carpentaria formed distinct groups at < 30 m and > 30 m. Both Rainer and Munro (1982) and Ramm et al. (1990) additionally found that some species were

Objective 4: Bycatch descriptions of banana prawn fisheries

distributed regardless of depth, or were widely distributed, which most likely accounts for species that occur in the bycatch of both fisheries and throughout the NPF.

Conclusions

This study provides the first description of the unique bycatch assemblage from trawling operations for *P. merguensis* in the Gulf of Carpentaria. It has also shown that two prawn trawling operations in one managed area (with similar trawl gear and the same vessels and skippers) have different bycatch assemblage structures. For fisheries to comply with increasingly stringent legislation, both new risk assessment methods and management rely on accurate information on the impacts of all mortality sources. This study highlights that no component of a fishery can be ignored in order to properly account for and manage bycatch; particularly fisheries with multiple target species, including those with large spatial and temporal distributions. Consequently, strategies for dealing with bycatch need to be more holistic in light of ecosystem based fisheries management and in the establishment of long-term bycatch monitoring programs. The present study provides information on bycatch from an unaccounted trawl fishery, crucial for properly quantifying trawl bycatch impacts in a large fishery management area within the Indo-Pacific.

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5.5.5 Summary and conclusions

S. Griffiths, M. Tonks and Q. Dell

Description of the bycatch from the Joseph Bonaparte Gulf banana prawn fishery

- The Joseph Bonaparte Gulf (JBG) is a unique and important region within the NPF. Despite comprising only 5% of the NPF area it contributes about 65% of the red-legged banana prawn (*Penaeus indicus*) catch (~360 t annually). However, the bycatch from this region is poorly known.
- This project provided the first comprehensive description of the JBG prawn trawl bycatch. We estimated that ~5000 t of bycatch is taken annually from the JBG. The bycatch comprised 195 taxa and 85 families, mainly teleosts (91% biomass), of which only six families contribute 81.6% of the biomass.
- The JBG assemblage structure contrasts with other NPF regions in that the portunid crab (*Charybdis callianassa*) and a few teleosts (*Harpadon translucens*, *Rhinoprenes pentanemus* and *Trichiurus lepturus*) dominated the catch. Furthermore, eight species identified from the JBG do not occur in other NPF regions.
- Bycatch assemblage varied temporally with higher total bycatch biomass being caught during spring (Aug–Nov) than in autumn (April–May). Some individual species showed seasonal and diel differences in catch rates and size composition.
- The new data will contribute to the long-term management and improve the sustainability of NPF bycatch species through ecological risk assessment (e.g. SAFE) and the establishment of a monitoring program.

Bycatch guide of prawn trawl species from the Joseph Bonaparte Gulf

- An identification guide was produced to provide industry, scientists, observers and managers with a useful reference for identifying the bycatch species of the JBG.
- Colour photographs, key taxonomic characters, size and distributional information is provided for 77 common species, representing elasmobranches, teleosts, crustaceans, cephalopods and marine reptiles.

Description of the bycatch of the Gulf of Carpentaria (GoC) banana prawn fishery

- The ‘tiger prawn’ trawl fishery in the GoC has been studied in detail, but the ‘banana prawn’ fishery in the GoC is undescribed. The cumulative impact from both fisheries on bycatch is also poorly understood.
- This project provided the first comprehensive description of the GoC banana prawn fishery bycatch. We estimated that ~1500 t of bycatch is taken annually. The bycatch comprised 226 taxa, although only 3 species (*Polydactylus multiradiatus*, *Caranx bucculentus* and *Rhizoprionodon acutus*) accounted for 75% of the total biomass.
- The bycatch of the banana and tiger prawn fisheries were both characterised by many species that made small contributions to the total bycatch, although the banana fishery

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interacted with fewer species. The banana fishery had a higher mean bycatch catch-rate from shorter duration trawls than the tiger fishery.

- The bycatch assemblage structure of the two fisheries significantly differed. The banana fishery bycatch had a higher proportion of teleosts, fewer numbers and species of invertebrates.
- This study showed the two NPF fisheries have different bycatch impacts. The new data will allow cumulative bycatch impacts to be better quantified and improve bycatch sustainability assessments.

6. GENERAL BYCATCH MONITORING PROGRAM RECOMMENDATIONS

6.1 Bycatch Monitoring Program

- 1. The NPF adopt a long term Bycatch Monitoring Program (BMP) in order to meet Ecosystem Based Fishery Management (EBFM) objectives for the NPF by (i) assessing the sustainability of bycatch species and communities impacted by the fishery into the long term and (ii) describing total discards from the fishery; as specified in the NPF BAP (2003), NPF industry code of practice for responsible fishing (2004) and Australia's EPBC Act 1999.**
- 2. The BMP be an ongoing collaborative program between AFMA, CSIRO and Industry.**

This collaborative model relies on using the key skills of each party, with AFMA providing the observer programs (Crew Member Observers (CMOs) and scientific observers) and data entry; CSIRO undertaking the fishery-independent survey, data analyses, risk assessments and annual bycatch sustainability reports; and industry providing the access to samples, other data before, during, and after the fishing seasons, and support for both CMOs and scientific observers at sea (Section 5.3).

- 3. CSIRO and AFMA via the BMP provide annual status reports of the sustainability for (i) Threatened, Endangered and Protected species (sea turtles, sea snakes, syngnathids), (ii) 'at-risk' species, (iii) bycatch community composition; and provide descriptions of the total bycatch discarded. This is to be delivered to NORMAC and the NPF RAG by August each year, reporting on the previous calendar year (Section 5.3).**
- 4. The long-term BMP adopt a slow start approach to help the fishery deal with the additional financial burden associated with the program.**

This is reflected mainly in the number of scientific observers in the fishery; starting with less than one full time equivalent in 2005 and 2006, and building to a 5% coverage in 2010. The recruitment of CMOs (and hence training costs) should increase from about 10 to 15 by 2008. Observer coverage to be re-assessed annually to ensure cost-effective coverage for key species (TEP and 'at-risk' species). This is also reflected in initially restricting the fishery-independent survey to the same regions currently visited by the mid-year prawn monitoring survey.

6.2 Monitoring

- 5. Monitoring of bycatch species will include a combination of sampling methods in order to provide the most cost-effective approach to assess the sustainability of all major bycatch groups (Section 5.2), in particular:**
 - a. Logbooks to provide most of the data necessary for sea turtles and total sea snake numbers; and later, for sawfish (see point 6 below).**

- b. Crew-Member Observers to provide most of the data necessary for syngnathids, sea snakes, sawfish, 'at-risk' species and total bycatch; and data for validation of logbook information.
- c. Scientific observers to provide additional data and validation of CMO data for syngnathids, sea snakes, sawfish, other 'at-risk' species, total bycatch; as well as collecting broad-ranging environmental data.
- d. Fishery-independent surveys to collect annual data during the mid-year prawn survey, on all groups and environmental data, and to collect the only suitable data for demersal faunal assemblage and size structure, including coverage of the key bioregions. This should include both trawled and control regions to interpret whether any changes in communities are due to fishing or from other sources (e.g. climate change, invasion of exotic species) (see recommendation 8).

6. Logbooks will be updated to provide more robust estimates of NPF fishing impact on individual sawfish species and estimates of the total bycatch.

Logbooks have proven to be reliable and accurate for sea turtle monitoring and should, with encouragement and some culture change, provide the same valuable information for sawfish and total bycatch estimates (Section 5.2). Sawfish are of considerable concern due to their vulnerable life history and global depletion. They are easily recognised and identified, but are relatively rarely caught. Logbook data collection for individual sawfish species will provide adequate data to assess changes in numbers caught over time and hence their population health. Total bycatch estimates are easily collected and an accepted indicator of the fisheries impact (Hall and Mainprize, 2005).

7. CMO and scientific observer programs should include annual training and appropriate support from AFMA, Industry and CSIRO to ensure their longevity (Section 5.3).

Risk of momentum loss in such programs is real (e.g. loss of industry participation in the previous sea turtle monitoring program), and the program support (dedicated staffing and funding) and protocols is critical to their long term success (Watson and Novelty, 2004).

8. Fishery-independent surveys should be added to the existing winter prawn monitoring survey to collect data on species composition of the small bycatch communities from the two bioregions in the Gulf of Carpentaria (represented by Nth Groote and Nth Mornington) , and control sites to be included to identify differences between fishery impacts and other factors (e.g. climate change) .

Changes to demersal faunal assemblage and size structure are critical indicators of ecosystem health (Fulton et al., 2004; Hall and Mainprize 2005) under the NPF EBFM objectives. Assessment of species composition is most cost-effectively done by focusing on the main bioregions (as per IMCRA process, also see Blaber et al., 1994; Somers and Long, 1994 and Stobutzki et al., 2001) in the Gulf of Carpentaria.

Adding to the current prawn monitoring surveys will provide the control and independence necessary for accurate, cost-effective assessments in two of the main bioregions, as well as creating a combined prawn and bycatch long term data set critical for assessing the health of broader marine communities (Section 5.2).

Control sites are required to explain any future changes to demersal faunal assemblages and size structure. Control sites are the only way to demonstrate if changes are due to fishing impacts or other factors such as climate change or invasion of exotic species.

9. The JBG be a third important bioregion included in the NPF fishery-independent survey program (Section 5.2.6).

The JBG has very different fishery activities and impacts a unique demersal assemblage compared to other regions in the NPF. It is therefore critical to monitor fishery impacts in this region. Monitoring of the JBG could be combined with a prawn monitoring survey to increase cost-effectiveness.

10. The recovery of bycatch populations, especially large, potentially vulnerable species (e.g. elasmobranchs, sea turtles), be demonstrated by periodic assessments (e.g. using fish or prawn trawls) with nets that have no TEDs installed.

Since the introduction of TEDs, catch rates of many large, ecologically important species have been difficult to assess due to small numbers caught in prawn trawls with TEDs. Periodic assessments without TEDs installed would provide data on relative catch rates for these key species and could be compared to similar surveys made in the late 1980's, 1990's and 2001.

6.3 Risk Assessment

11. A quantitative ecological risk assessment be used to assess the fisheries-induced risks to selected bycatch species, following the collection of new data (Section 5.4).

The risk assessments should be repeated where new species-specific data may improve the assessment for species already identified as 'at-risk' by the method. This will ensure accurate and ongoing demonstration of sustainability for all bycatch species.

12. A quantitative ecological risk assessment used for all bycatch species following any major change to fishing gear or effort distribution patterns.

13. Reference points used in the risk assessment be re-evaluated to ensure their legitimacy and acceptance.

Reference points used in the risk assessment to assess potential risk (μ_{msm} or μ_{crash}) have been chosen within the project and reviewed by the NPF RAG. It is critical that the reference points be reviewed to ensure their level of precaution is appropriate and acceptable for a much broader range of fisheries (Section 5.2.6).

14. Cumulative ecological risk assessments be developed to incorporate influences from all northern fisheries and other human activities.

In cases where the NPF is not the only significant human influence on mortality, risk assessments should be broadened to include all influences; for example, the impacts of other northern Australian and Indonesian fisheries on sawfish, shark and ray

populations. AFMA and CSIRO will engage the national ERA forums for support and momentum for this action.

6.4 Alternative Management Strategies

15. Alternative management strategies be developed to promote sustainability for rare species that cannot be robustly assessed using the recommended monitoring strategy.

Alternative management strategies appear to be the only way to cost-effectively demonstrate sustainability for species too rare to detect changes with monitoring (Section 5.2.4). We recommend that a committee of industry, scientific members and other stakeholders, such as the bycatch sub-committee, examine the range and usefulness of alternate management strategies for this purpose.

6.5 Management Actions

16. The BMP recommend a series of reference points, triggers and management actions to provide appropriate responses to trends in bycatch data (Section 5.2.6).

Each year catch data will be collected on a range of targeted bycatch species. Management actions in response to changes in relative catches are critical to guide fishery management (Hall and Mainprize 2005). The BMP will participate in national ERA forums for guidance and consistency across fisheries on these issues.

17. The BMP be reviewed periodically and adapted to maximise the cost effectiveness of the program and the needs of the fishery management (Section 5.2.6).

Processes such as periodic risk assessments and sustainability assessments may flag changes in the needs of the monitoring program. It should be recognised that the ongoing program be able to incorporate changes such as:

- **removal or addition of species potentially ‘at risk’ in the monitoring program;**
- **use of upgraded versions of the risk assessment technique.**

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7. BENEFITS AND ADOPTION

The main beneficiary of this work is the Northern Prawn Fishery, which can implement a cost-effective, world's best practice bycatch monitoring program. This will allow the fishery to assess whether the catches of most of the species it directly impacts are remaining sustainable. It provides a critical step towards the fishery fulfilling its obligations under the DEH strategic assessment and EPBC Act 1999. The adoption of the monitoring program as recommended, is directly dependent on consistent and adequate funding by NORMAC and AFMA.

Similar fisheries, such as the Torres Strait, Qld east coast and Western Australian prawn trawl fisheries will also benefit from the innovative risk assessment methods and comparison of data collection methods, in particular. These outcomes can also provide guidance for other trawl fisheries in how to assess and demonstrate sustainability for species impacted for which little is known.

Other fisheries can also benefit from the new quantitative risk assessment method. It can be adapted to provide a quantitative assessment of risk for species impacted for which little is known, especially fishery bycatch. It has particular potential for assessing the risk to vulnerable species impacted by more than one fishery. This can provide Australian fishery management with a powerful tool for assessing the fully inclusive risk to key species from fisheries impacts.

The general community will also benefit as fisheries adopt techniques that flow from this project, as they will allow fisheries to provide a more precise assessment of their impact on a wider range of species impacted. The longer term corollary of this should be an increased awareness of fishery impacts and increased species and community sustainability in marine systems.

7.1 Beneficiary responses

7.1.1 Edward Hegerl (Conservation Representative, NORMAC)

I believe that the research team has carried out an excellent and innovative piece of work that will have important long-term benefits to the Northern Prawn Fishery. Combined with the broad bycatch knowledge base accumulated over previous bycatch projects, the fishery is now well-positioned to monitor the impact of trawling on bycatch communities, and to ensure their sustainability.

Monitoring of selected bycatch species by both scientific observers and crew member observers has been undertaken since 2003. The data collected will form the basis for better understanding the impact of trawling on these populations. In my opinion, the current level of monitoring by scientific observers is too low to allow validation of data collected through other methods (eg crew member observers or logbooks). Increased validation should result in greater public acceptance and confidence in management decisions where bycatch species are concerned.

There is a strong need for AFMA to digest the recommendations of this report, in particular, the setting of biological reference points and agreed management responses to emerging situations if limits are reached. As the Conservation Member on NORMAC, as well as a Steering Committee member for this project, I am well aware of and support the need to reduce fishery costs as much as is reasonably possible, but not to the detriment of bycatch sustainability. Many practical options for monitoring are canvassed in this report, and they should be seriously considered by management.

Bycatch monitoring and attempts to reduce the total volume of bycatch and ensure ecological sustainability will be ongoing in this fishery. This project has defined the important ground rules that need to be part of a successful toolkit for fishery managers to achieve sustainability. Over a protracted period, the Northern Prawn Fishery has demonstrated its willingness to address these tough issues, and is to be further congratulated for the example it sets to other trawl fisheries with bycatch issues.

Edward Hegerl

7.1.2 Mike O'Brien (Industry Representative, NORMAC)



RAPTIS

A RAPTIS & SONS PTY LTD

ABN 47 065 021 463

17 August 2007

CSIRO
233 Middle Street
CLEVELAND QLD 4163

Attention: Mr David Brewer
E-mail: David.Brewer@csiro.au

Dear David

Re: Beneficiary Response for the NPF Bycatch Monitoring Project Report

Please find below my Beneficiary Response for the NPF Bycatch Monitoring Project Report:

Designing a cost-effective long-term monitoring program suitable for the bycatch of the Northern Prawn Fishery was always going to be a "tall order". Ever increasing checks and balances on the sustainability of bycatch impacted by the fishery have been introduced since the 1990s. As a member of the Steering Committee, I have been closely involved in the direction of this project, and have gained a much greater understanding of the nuts and bolts needed to construct a useful and cost-effective monitoring program. This project provides a series of recommendations that will form the basis for this fishery to move forward in monitoring its impact on bycatch.

I feel that the introduction of quantitative risk assessments has better positioned the fishery to deal with the legislation and answer criticisms from NGOs. However there is still some way to go in settling on an effective program where all parties agree on the long-term objectives. Although the recommendations are wide-ranging, stakeholders need to agree on triggers and actions to be undertaken if a species is considered to be impacted unsustainably.

In general, the project has been well managed and its results form a critical stepping stone for the successful future management of the bycatch of this fishery.

Should you require further information, please do not hesitate to contact me on 0427 600205.

Yours faithfully

Michael O'Brien
Northern Division Manager

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8. COMMUNICATION OUTPUTS

8.1 Abstract

Effective communication and liaison was a critical part of this project. Strong links with stakeholders ensured cooperation between all parties to enable industry involvement in experimental design and data collection, better understanding by researchers of the industry and their needs and the input of management and conservation perspectives. Success has been demonstrated by the strong support and participation of the industry in the scientific observer, Crew Member Observer (CMO) and Requested Industry Collection (RIC) programs, the endorsement of the recommendations by the Steering Committee and the high level of formal (nine scientific papers and one industry bycatch guide) and informal communication products (brochures, flyers, newsletters and progress reports) published and distributed throughout the project.

8.2 Introduction

The outcomes of this project were highly dependent on targeted and successful communication and consultation with management, industry and other stakeholders of the Northern Prawn Fishery (NPF). It was considered critical to the success of this project that all stakeholders be fully informed of the objectives, activities, results and benefits of the project. Strong links with stakeholders ensured cooperation by all to enable industry involvement in experimental design and data collection, better understanding by researchers of the industry and their needs and the input of management and conservation perspectives. It has been recognised that effective communication and liaison within this project was a critical component.

8.3 Objectives

1. To ensure a high level of participation in the CMO and RIC programs to collect NPF bycatch data.
2. To build on the growing culture of environmental accountability in the NPF by using a program of printed information, industry workshops and personal communication within the NPF industry.
3. To provide the NPF and associated stakeholders with up-to-date information on planned and completed activities throughout the life of the project.
4. To ensure that the program to monitor bycatch in the NPF continues as a normal part of the industry's activities into the long term
5. To make managers of other fisheries, and other associated industries aware of the program that has been developed and to examine it for their fishery.

8.4 Target Audiences

The main target audience engaged was the NPF fishers, especially skippers and crew that participated, or may be future participants in the bycatch monitoring program.

The other critical target audiences included other stakeholders of the NPF such as the Department of Environment and Heritage (DEH), non-government conservation groups (NGOs), other Australian fisheries, (particularly prawn trawl fisheries), and the general

public.

8.5 Key messages delivered

1. A bycatch monitoring program for the NPF is a necessary step for the industry to satisfy its own management objectives as well as the requirements of National legislation and the international marketplace.
2. The most cost-effective way for industry to fulfil its bycatch monitoring obligations is to participate in this program to trial the most likely methods.
3. Participation in the NPF scientific observer, CMO and RIC programs will be interesting, include training and workshops that will be fun and rewarding to be involved in.
4. All stakeholders should have confidence in the process. The development of a bycatch monitoring program will include the active participation and decision making of a range of industry stakeholders including CSIRO – CMAR (research providers), AFMA, NORMAC (management), the fishing industry, DEH, and other stakeholders. This will ensure that the most cost effective and acceptable methods are trialled, and the most suitable methods are used into the long term.
5. The development process can be used as a prototype for other fisheries.

8.6 External communication and liaison

Communication activities were varied, including:

- Reporting to the steering committee every 12 months
- CMO training to clarify and prioritise the objectives of the monitoring program, discuss methods and feasibility issues, instigate a trial program, and evaluate the results of the trial.
- Newsletters distributed at port visits to the NPF fleet as required.
- Personal communication between project staff and industry stakeholders about all aspects of the project, via phone conversations, fax messages, workshops and project meetings.
- Articles in industry and other magazines such as “Professional Fisherman”, “Fishing Future”, “Fishing Boat World”, “Waves” and R&D NEWS.
- Press releases, media interviews etc.
- Scientific paper and industry guides

A description of the external communication and liaison activities used in the project are described below.

8.6.1 Planned activities

Project staff maintained personal interaction, had frequent phone contact and distributed written progress reports about the project to a range of stakeholders, especially NPF operators (skippers and crew, trawler owners, net makers and other involved companies), other researchers and government managers. Planned activities were as follows:

Personal contact and port visits

Personal contact with sea-going personnel occurred during fishery closure periods, usually just before the start of the fishing seasons for the convenience of trawler skippers and crews. Opportunities to contact shore-based people, either in person or by phone were taken at

Communication Outputs

various times throughout the year. Contact ranged from casual informal one-on-one discussions to more formal meetings and workshops with both small and large groups.

Project staff made regular visits to NPF ports (Table 26) to take advantage of the gathering of NPF operators just before the start, and at the end of each fishing season. A total of 12 port visits were made during the project. Ports visited included Brisbane, Cairns, Karumba, and Darwin. These ranged from formal meetings at advertised venues and in company premises, workshops and practical demonstrations, to informal discussions at wharf-side and on-board operator's vessels.

Distribution of written material

Written material included project newsletters, circulars, notices and articles in popular literature and formal scientific papers.

Throughout the project research and project newsletters/progress reports (listed in Table 27) were produced and distributed NPF operators and stakeholders (examples have been included in Appendix C). The *Bycatch newsletters* included project objectives and methods, introduced staff and collaborator lists, detailed the proposed and completed activities, and presented project results and progress reports. A number of brochures were produced and distributed to NPF operators and other stakeholders.

Communication Outputs

Table 26. Schedule of port visits and other communication and liaison events with NPF industry.

Date	Port(s)	Events and Primary Objectives	Staff
July 2003	Gold Coast	CMO Training	
Pre-Tiger Aug/Sept 2003	Darwin, Cairns	Distribute RIC and CMO kits, inform and encourage participation by fishers in Bycatch Monitoring Program, general liaison, verbal progress reports	Kenyon, Taylor, Dell, Tonks, Gregor, Bain
Post-Tiger Dec 2003	Darwin	Recruit and liaise with CMO's and RIC vessels for feedback and distribute project t-shirts + hats to thank operators	Taylor, Bain
Pre-Banana April 2004	Darwin, Cairns	Recruit fishers into the RIC Program, general liaison, verbal progress reports, handout feedback forms, distribute project t-shirts + hats to thank operators, liase with CMO's and re-stock sampling kits	Dell, Taylor, Tonks, Whitelaw, Gregor
August 2004	Gold Coast	CMO Training	
Pre-Tiger Aug 2004	Darwin, Cairns	Recruit fishers into the RIC Program, general liaison, verbal progress reports, handout feedback forms, distribute project t-shirts + hats to thank operators, AFMA briefing	Dell, Tonks, Whitelaw, Gregor
Post-Tiger Dec 2004	Darwin, Cairns	Recruit and liaise with CMO, RIC and BRD vessels	Dell, Whitelaw, Piasente
Pre-Banana April 2005	Darwin, Cairns	Recruit and liaise with CMO's, general liaison, verbal progress reports, distribute project sampling kits, AFMA briefing	Dell, Whitelaw, Piasente, Burke, Tonks
Pre-Tiger July 2005	Darwin, Cairns	Recruit and liaise with CMO, RIC and BRD vessels, re-stock sampling kits, distribute project t-shirts + hats to thank operators, AFMA briefing	Dell, Whitelaw
Post-Tiger Nov 2005	Darwin, Cairns	Recruit and liaise with CMO's and RIC vessels and distribute fish guide book as thankyou to CMO's.	Whitelaw, Burke
July 2006	Gold Coast	CMO Training	
Pre-Tiger July 2006	Darwin, Cairns, Karumba	Recruit and liaise with CMO, RIC and BRD vessels, distribute progress reports and literature, AFMA briefing	Dell, Whitelaw, Burke, Tonks

Communication Outputs

Table 27. List of external communications

Format	Title	Distributed
Newsletters	1. “New by-catch monitoring project for the NPF”	2002
	2. “NPF Research: Tiger Prawn species distribution”	
	3. New NPF projects focus on prawn and by-product monitoring and prawn movements”	
	4. “Northern Prawn Fishery TED and BRD Project. Newsletter No 5 – March	
	5. “BRDs in the NPF”	
	6. Project Newsletter – “New by-catch monitoring project for the NPF”	2003
	7. Research Newsletter – “New by-NPF projects focus on prawn and by-product monitoring and prawn movements”	
	8. Project Newsletter – NPF Bycatch & other news - December 03	
	9. TED and BRD Project Newsletter	2004
	10. Bycatch monitoring in the NPF” coloured progress report	
	11. “Bycatch Monitoring: Feedback Newsletter”. <i>CSIRO industry brochure</i> , December 2004.	Late 2004/early 2005
	12. Bycatch monitoring research progress report. <i>CSIRO industry brochure</i> , March 2005.	
	13. NPF bycatch monitoring project progress report, March 2005	
Brochures/ Posters	1. Bycatch Monitoring Project– General brochure	2003
	2. Moving towards sustainable impacts in Australia’s NPF” poster	2004
	3. Crew awareness program” flyer and booklet	
Media Releases	Media releases	
	CMO press release	2004
Magazine Articles	1. Magazine article describing the CMO training workshop (FISHERIES NEWS)	2003
	2. Optimising TED performance in tropical prawn trawl fisheries, parts 1 & 2. <i>Professional Fisherman</i> June and July 2004. Eayrs and Day	Late 2004/early 2005
	3. “Partnership is the key to sustainable management in the NPF”. <i>Fishing Future</i> , August 2004.	
	4. Reducing bycatch in the NPF. <i>Waves</i> , Winter 2004.	
	5. “Northern prawn fishers return snakes alive”. <i>Professional Fisherman</i> , September 2004.	
	6. “Managing bycatch in the Northern Prawn Fishery”. <i>Professional Fisherman</i> , October 2004.	
	7. “Volunteer crew recruits help with scientific data collection”. <i>Fishing Future</i> , December 2004.	
	8. “Dramatic bycatch reductions in Australia’s Northern Prawn Fishery”. <i>Fishing Boat World</i> , January 2005.	
Pro-formas, letters, other	1. Bycatch monitoring stakeholders acceptance level pro-forma	
	2. Bycatch monitoring (RIC and CMO’s) feedback survey forms	
	3. Bycatch (CMO’s) exit survey form	
	4. Certificates presented to vessels, fleet managers, owners and others who helped the first years data collection (e.g. mothership skippers), in appreciation for their efforts	2005

Communication Outputs

Format	Title	Distributed
	5. Awards given to selected Crew-member Observers during the 2004 training – large framed prints of marine fish. 6. Letters of appreciation sent to fishers, fleet managers and owners – June 05	

8.6.2 Promotional material for the project

Caps and T-shirts: Caps, T-shirts, cups and bags with the project logo (See Figure 50) were designed and distributed to NPF operators and associates in the early stages of the project.

Certificates of Appreciation: Certificates of appreciation were sent to the owners and vessels who participated in the Observer Program during 2003 to 2006 of the project.



Working towards sustainability

Figure 50. Project logo

8.7 Indications of success

The success of the project was measured by the:

1. Participation of NPF crews and skippers in the trial of the crew member observer program.
2. Acceptance by all stakeholders of the recommendations for monitoring NPF bycatch into the long term; via agreement by the full project steering committee as well as endorsement by NORMAC and DEH.
3. Formal written publications.

8.7.1 Participation

Participation in project activities: There was a high level of interest, participation and co-operation from trawler owners, skippers, crew and other industry stakeholders (for example mothership, freezer and transport companies) during the at-sea components of the project. Approximately 10 CMO's per year, participated in the Observer Program and between 20 and 30 boats were involved in the RIC component each year. This ensured there was sufficient industry coverage to successfully assess the bycatch monitoring methods trialled in this project (Section 5.2.1). Mothership operators were kept well informed of project activities and provided a high level of co-operation during the observer programs.

The attendance by operators at meetings and participation at workshops and practical demonstrations: Pre-season meetings, usually held by company operators and AFMA provided forums for effective information exchange with large numbers of skippers and crew. CMO training workshops were very effective forums for industry involvement.

The willingness of individuals to discuss the project on boats and wharf-side: The timing of port visits maximised the ability of project staff to interact with a significant proportion of NPF operators at several times during each year (up to 90% of skippers and owners Darwin and Cairns).

Good industry awareness of relevant literature available and potential sources of information: The awareness of members of the fishing industry was apparent during discussions and when distributing material. This was also reflected in their willingness to cooperate with CSIRO and other organisations in subsequent fishery-dependent projects (for example, RIC and CMO programs): The ongoing presence of CSIRO and AFMA project staff at NPF ports, at industry forums and on vessels has played a part in the successful level of co-operation gained from NPF operators during this prawn sampling project.

It has been clearly demonstrated that the objectives of the project's liaison and communication activities were met. The collaborative relationships that have been established between CSIRO, AFMA and NPF operators, provides an excellent platform for the large amount of fishery-dependent data collection activities planned for future years.

Project staff will continue to provide information on the results of this project through industry workshops and port visits initiated in subsequent projects and through publication of results in fishing magazine articles and scientific papers.

8.7.2 Acceptance of the recommendations for monitoring NPF bycatch

The recommendations listed in this report were endorsed by the Steering Committee at the final Steering Committee meeting on the 11th of October 2006. NORMAC and DEH are currently considering the recommendations and a series of cost scenarios for future long term monitoring programs.

8.7.3 Formal written publications

This project report includes nine scientific papers which will be published in internationally refereed journals (two are already published) and one industry bycatch guide. This high level of formal output is exceptional and in conjunction with all the other less formal publications (Table 27) demonstrates the commitment this project has to delivering and distributing the science developed to as many stakeholders as possible.

9. FURTHER DEVELOPMENT

The outcomes of this project have continued in the form of an ongoing bycatch monitoring program (BMP) in the NPF. This has provided four years of continuous data collection in the fishery on Threatened, Endangered and Protected (TEP) species in particular. It has also helped the fishery to develop a new culture and awareness associated with bycatch and their data collection.

The ongoing BMP, assuming an appropriate level of funding, will now begin collecting time series data on all species of concern (TEP and 'at-risk' species), allowing the fishery to be able to assess the sustainability for all species impacted directly in trawls. The ongoing BMP will continue to develop some of the methods that are critical to the assessment of this work, in particular, the details of the analyses and responses required to assess sustainability for the wide variety of poorly understood species included in the program. This will include strong engagement in Ecological Risk Assessment and other forums to ensure a national approach to resolving this issue.

The recent ministerial directive to halve discarding in fisheries by 2008 and current BRD initiatives in the industry will also interact with this program. These and other initiatives will take advantage of the observer presence in the industry to collect data vital to their programs

The risk assessment developed by this project has great potential to assess risk in other fisheries and combined risk of more than one fishery of impact on individual species. This is likely to be an important tool in future fisheries management in Australia and elsewhere. The risk assessment technique can also be developed to provide relatively fast assessment of the impacts of a range of fishery management initiatives (e.g. spatial or effort management) on the sustainability of all bycatch caught.

10. PLANNED OUTCOMES

1. **Continued, cost effective, long-term monitoring of bycatch in the NPF managed by NORMAC and AFMA that fulfils the NPF's reporting requirements and has the confidence of all NPF stakeholders.** A series of recommendations describing a comprehensive, but cost effective program have been accepted by the projects steering committee and are presented in this report. Implementation of these recommendations will ensure that the NPF will satisfy national reporting requirements for bycatch and deliver a model equivalent to current worlds best practice.
2. **Transfer of the guidelines to other Australian and international fisheries to assist the development of long-term monitoring programs that address their legislative reporting requirements.** This will contribute to the fulfilment of ecosystem-based fishery management objectives being implemented in an increasing number of Australian and international fisheries.
3. **The continued development of an industry culture that recognises the need to monitor bycatch and provide accurate and reliable information, as is the current situation for target species.** Collections directly from both industry and scientific platforms are critical to monitoring bycatch, in order to provide adequate data and independent validation to ensure that all stakeholders are confident that the program will satisfy its goals. Through working with industry during this and related projects we have enhanced their capacity and emphasised the importance of providing high quality data to ensure the program's continued success.
4. **A favourable strategic assessment of the NPF against DEH's guidelines.** The second principle of the DEH guidelines for strategic assessments of fisheries, assesses a fishery's management of bycatch and protected species. This project will provide the NPF with a long-term monitoring program for bycatch, enabling reporting against DEH bycatch objectives.
5. **A favourable assessment against Ministerial guidelines to reduce total discarding.** The December 05 ministerial directive requires fisheries to 'reduce discards by half by 2008'. The NPF bycatch monitoring program will provide data collection strategies to allow the fishery to assess future discarding rates, through the crew member and scientific observer activities.
6. **Demonstration of the NPF's intent to develop an ecosystem-based fishery management plan.** The effective management and sustainability of bycatch assemblages would make a significant contribution to such a plan. This will contribute to demonstrating the NPF's intent to develop recognised environmental standards for incorporation into fishery certification processes that lead to world's best practice.
7. **Improved acceptance of NPF practices and reduced concern by stakeholders such as conservation groups, recreational fishing groups and the general community.** Ongoing monitoring of bycatch would assist the NPF to address

Planned Outcomes

concerns from government and NGOs regarding the sustainability of species impacted by the fishery. In particular, monitoring will provide ongoing annual assessments on the sustainability of Threatened, Endangered, Protected, vulnerable, and 'at-risk' species interacting with the fishery.

11. CONCLUSIONS

A combination of sampling methods can provide a cost-effective bycatch data-collection program for the NPF. The recommended approach uses the high sampling power of fishery-dependent methods to collect monitoring data for the main, and mostly rare, species of concern, namely Threatened, Endangered and Protected (TEP) sea turtles, sea snakes, syngnathids, sawfish; and ‘at-risk’ elasmobranchs and fish. This is combined with a high level of broader stakeholder acceptance, by including annual training for Crew Member Observers and validation of data for all species groups using a combination of methods including scientific observers and fishery-independent surveys. The sustainability of species that are too rare to allow detection of declines can only be managed using alternative management strategies. The assessment of bycatch species composition can only be based on data collected by scientific observers or fishery-independent surveys as the other methods collected an unacceptably high proportion of inaccurate data. The cost of the program will vary from an estimated \$270,000 to \$885,000 depending on the level of species coverage, the inclusion of bycatch assemblage structure assessment and extent of regional coverage.

A new, quantitative, ecological risk assessment approach – Sustainability Assessment for Fishing Effects (SAFE) – was used to identify at-risk species and focus the Bycatch Monitoring Program (BMP) on species of concern only. SAFE identified six species for inclusion in the NPF BMP: four elasmobranch species (*Orectolobus ornatus*, *Squatina sp. A*, *Taeniura meyeri* and *Urogymnus asperrimus*) and two teleost species (*Dendrochirus brachypterus* and *Scorpaenopsis venosa*). However, in order to assess the true sustainability of species impacted by several fisheries in northern Australia (e.g. sawfish, sharks and rays) a cumulative impact ecological risk assessment is required. This is possible by modifying the existing SAFE model.

The BMP will collect medium and long-term data sets on TEP and ‘at-risk’ species to determine whether (i) there is ongoing risk (= remain in the monitoring program); (ii) the risk is not real or removed (= cease monitoring); or (iii) populations are unsustainable under the current impact. In the latter case, the fishery should formulate and instigate a specific threat abatement plan to remove this risk, such as using a specific bycatch reduction device to reduce its catch and improve survival.

The project also established that an estimated 4,868 t of bycatch, including 195 different taxa, is taken from the Joseph Bonaparte Gulf annually. The composition of these catches are distinctly different from that of other tropical regions, and includes eight species that have never been recorded from other bycatch studies in northern Australia. The bycatch of the Gulf of Carpentaria banana prawn trawl fishery was also described for the first time. These catches were also highly diverse (226 taxa) and uniquely different from the other NPF sub-fisheries.

AFMA, CSIRO and the NPF industry will work in collaboration to provide the data and analyses required to assess the sustainability of bycatch in the NPF. Adequate and ongoing commitment to funding from NORMAC and AFMA are required to ensure ongoing momentum and the delivery of related outcomes by the NPF BMP.

12. APPENDICES

12.1 Appendix A: Intellectual Property

All components of this research are in the public domain.

12.2 Appendix B: Staff

CSIRO

David Brewer – Project Coordination, methods for monitoring (Obj 1) and transfer to AFMA (Obj 2)

Sandy Keys – Project scientific support officer

Shane Griffiths – Risk assessment (Obj 3) and bycatch of Joseph Bonaparte Gulf (Obj 4)

Don Heales – Methods for monitoring (Obj 1), coordination of field and training activities

Shijie Zhou – Quantitative risk assessment model development (Obj 3)

Brian Taylor (retired) – Communication with Industry and preparation of newsletters

Margaret Miller – Database, data summaries

Quinton Dell – Scientific observer, sample processing, monitoring data management, GoC banana fishery bycatch description (Obj 4), anecdotal information database

Mark Tonks – Scientific observer, sample processing, monitoring data management, JBG fishery bycatch description (Obj 4), JBG bycatch guide

AFMA

Wade Whitelaw – Project co-investigator, AFMA-CSIRO co-ordination, handover to AFMA (Obj 2)

Adrienne Burke – CMO and scientific observer program, coordination with CSIRO

Erik Raudzens – CMO program, AFMA data

Current external steering committee members

Mike O'Brien (RAPTIS) – Industry Representative


Eddie Hegerl – Conservation Representative

Tony Gofton – Industry Representative

Paul Williamson – Industry Representative

12.3 Appendix C: Examples of communications outputs

Project introduction flyer



New by-catch monitoring project for the NPF

A new FRDC, CSIRO and AFMA funded project will design, trial and implement a by-catch monitoring program in the NPF. The project, which began in August 2002, will:

- Design, trial and implement an integrated long-term by-catch monitoring program that addresses total by-catch, protected species and high-risk species.
- Transfer the ownership of and responsibility for ongoing monitoring to NORMAC and AFMA.
- Validate the risk assessment of high-risk by-catch species.
- Provide the first description of by-catch from the Joseph Bonaparte Gulf.

Research activity

Crew-member observers, scientific observers and fishery independent research will be used to collect the most accurate, least biased and cost-effective catch information.

The project will assess the best way for the NPF to measure its ongoing impact on the by-catch and demonstrate its intention to ensure ecological sustainability of these species. To maximise industry involvement and commitment, and at the same time, fulfil the NPF's own reporting requirements, the system must have the confidence of all involved stakeholders.

We need your help!

The initial work selecting the best method will rely on by-catch samples collected by commercial trawlers, and, for occasional comparison, a few research surveys. Commercial operators prepared to collect samples next year should think about this. Expressions of interest will be welcomed and will be called for later. You will then receive detailed instruction about collecting procedures. The collection of samples should have very little interference with normal commercial operations.

Joseph Bonaparte Gulf

Little has been recorded about the by-catch in JBG, which is a unique and important part of the NPF. The project will provide for the first time some preliminary knowledge of the by-catch from this region.

What's in it for fishers?

More and more fisheries are seeking certification by organisations such as the Marine Stewardship Council, which provide internationally accepted standards and certification of sustainable practice

that can be used to commercial advantage in many international marketplaces. The NPF is seeking this form of certification and a by-catch monitoring program will also play an important role in achieving this.

Industry involvement and the willingness and ability to monitor impacts on the environment demonstrate a culture of environmental accountability. This promotes acceptance of NPF practices by stakeholders such as conservation groups, recreational fishing groups, the general community and national and international seafood traders.

Legislative requirements

NORMAC has recently reviewed and revised its by-catch action plan, which aims to ensure that the NPF can comply with guidelines for Environment Australia's (EA) strategic assessment. Under the Commonwealth's new Environmental Protection and Biodiversity Diversity Conservation legislation (EPBC Act) fisheries must have management practices in place that address the long-term sustainability of all species impacted.





This project will greatly improve the NPF's credentials against these assessments (export permit depends on this).

Some take home messages

- A by-catch monitoring program for the NPF is a necessary step for the industry to satisfy its own management objectives, as well as satisfying the requirements of national legislation and the international marketplace.
- The most cost-effective way for the industry to fulfill its by-catch monitoring requirements is to participate in this program to trial likely methods.
- All involved stakeholders must have confidence in the process. The development of a by-catch monitoring program will include the active participation and decision making expertise of a range of stakeholders including the fishing industry, CSIRO, AFMA and NORMAC, EA, and all other stakeholders. This will ensure that the most cost effective and acceptable methods are trialled, and the most suitable method(s) is selected for use.
- Participation in the NPF crew-member observer program (yet to be detailed) will be interesting and include training and workshops that will be fun and rewarding.

It is our intent to keep operators fully informed as this project progresses and we welcome any constructive comments including those about communications.

Contact:
 AFMA: Northern Prawn Section Phone 02 6272 5039
 CSIRO Marine Research: Iona Stobuzki Phone 07 3826 7295

August 2002

CMO Flyer

NPF BYCATCH MONITORING PROGRAM



Working towards sustainability



2003-04 Crew Member Observers

WHAT IS THE PROGRAM?

Crew Member Observers (CMOs) are one part of the Bycatch Monitoring Program that is testing the best way for the Northern Prawn Fishery to collect long-term bycatch data. By collecting information about the volume and type of bycatch, you can help promote the fishery and better protect its long term future.

WHAT DOES BEING A CMO INVOLVE?

- Attending a two-day training workshop
- Estimating bycatch volumes & occasionally weighing bycatch during the season
- Identifying and measuring turtles, sharks, rays & sawfish
- Taking photos of seasnakes
- Collecting small samples of bycatch
- Filling out field sheets

WHAT DO I GET OUT OF IT?

- An all expenses paid two-day workshop (on the Gold Coast last year)
- Work alongside NPF scientists and management
- Training in species identification and bycatch issues
- Satisfaction that you're helping to provide your fishery with more accurate information that will ensure better decision-making.

WHEN DOES ALL THIS HAPPEN?

Your workshop will be held mid-season 2004. The Bycatch Monitoring Program started during the second season last year and CMO participation continues through both seasons this year.

HOW DO I GET INVOLVED?

Ring Reuben Gregor at AFMA on (02) 6272 5632 or 0410 630 142. Reuben will also be visiting boats in April 2004 to recruit volunteers.

AFMA Project Introduction Flyer

NPF BYCATCH MONITORING PROGRAM



WHAT IS THE PROGRAM?

This program is about getting operators involved in collecting information about bycatch across the fishery (in the same way as the Turtle Monitoring Project). By collecting information about bycatch volume and type you can help promote the fishery and better protect its long term future.

WHAT DO I HAVE TO DO?

- Attend a two-day training workshop
- Estimate bycatch volumes and occasionally weigh bycatch during the season
- Identify and measure turtles, sharks, rays and sawfish
- Take photos of sea snakes
- Collect small samples of bycatch at certain times
- Fill out a logbook.

WHAT DO I GET OUT OF IT?

- An all-expenses paid two-day workshop on the Gold Coast
- Work alongside NPF scientists and management
- Training in species identification and bycatch issues
- Satisfaction that you're helping to provide your fishery with more accurate information that will ensure better decision-making.

WHEN DOES IT START?

Your workshop will be held mid-season 2003. The actual monitoring program will begin at the start of Tiger season (1 September 2003).

HOW DO I GET INVOLVED?

Call Alistair Bain or Reuben Gregor at AFMA on (02) 6272 5632.

Alistair and Reuben will also be visiting boats in March 2003 to recruit volunteers.

Project Brochure










David Brewer Shane Griffiths Don Heales Quinton Dell Mark Tonks Brian Taylor Alastair Bain Reuben Gregor

monitoring bycatch

in the Northern Prawn Fishery

Questions and suggestions

CSIRO Marine Research, Cleveland
 Brian Taylor (07) 3826 7226 ■ project mobile 0427 722 782
 Project team: Neil Loneragan, David Brewer, Shane Griffiths, Don Heales, Quinton Dell, and Mark Tonks. Margaret Miller, Peter Toscas and Chaiti Bunidge provide statistical and computing expertise.

AFMA, Canberra
 NPF manager (02) 6272 5099 ■ Alastair Bain (02) 6272 4834 ■ Reuben Gregor (02) 6272 5632

PHOTO CREDITS: Fisheries: Ben Bird, Australian Maritime College, Quinton Dell, CSIRO, Stirling Peverell, ODPI Fisheries, CSIRO project staff: John Saltn, CSIRO.
 AFMA project staff: Courtesy of AFMA.

- JULY 2003 -







A method for monitoring bycatch, to support sustainable management of the Northern Prawn Fishery (NPF), is being developed in a research partnership between scientists, government agencies and industry.

The three-year bycatch-monitoring project, which began in March 2003, pools the knowledge and resources of CSIRO, the Australian Fisheries Management Authority (AFMA), the Fisheries Research and Development Corporation (FRDC) and operators in the NPF banana and tiger-prawn fisheries.

It will determine the best way for fishers and AFMA to monitor bycatch – including protected and high-risk species – and will help the industry to demonstrate compliance with national and international environmental standards.



Showing good management

Under the Environmental Protection and Biodiversity Conservation Act (1999), all Commonwealth fisheries, including the NPF, must apply management practices that promote the long-term sustainability of species affected by the fishery. This is essential to the granting of export permits by Environment Australia (EA).

The NPF Bycatch Action Plan developed by the Northern Prawn Management Advisory Committee (NPMAC) outlines appropriate bycatch management practices for the NPF. Turtle exclusion devices (TEDs) and bycatch reduction devices (BRDs) have been introduced under the plan.

An effective system for measuring the fishery's impact on bycatch – accepted by all stakeholders

– is another key component of the plan, and will help to target and assess bycatch management strategies.

The bycatch-monitoring project will also move the industry toward achieving certification by the Marine Stewardship Council, which carries a commercial advantage on local and international markets.

What information is being collected?

For data collection purposes, bycatch in the NPF can be divided into two groups:

- large and obvious animals such as turtles, sharks, rays, sawfish, sponges and fish; and
- trash-chute material such as smaller sharks, rays, fish and invertebrates (crabs, non-commercial prawns and other crustacea, sea-eggs, sponges and corals).

Scientists need more information on bycatch in both groups to help determine which species are at risk from trawling. Bycatch-management strategies can then be developed.

Five methods of sample and data collection are being tested. They are:

- sampling and data collection on commercial boats by scientific observers;
- sampling and data collection on commercial boats by trained, volunteer crew members recruited by AFMA;
- additional samples collected by commercial operators;
- log-book records (for some of the bycatch species groups); and
- surveys by fishery-independent researchers on charters or research boats.

How you can help

Sample collection
 During the project, some trawler operators will be asked to collect samples of the trash chute bycatch, some will be asked to take scientific observers on board, and some may have trained observers on board as part of their regular crew.

Fishers' knowledge
 Many industry operators will be asked to share their observations and knowledge about bycatch: questions such as what species were caught, when, where and under what conditions, and what has changed over time?

By providing insights on individual species, this anecdotal information will highlight any significant historical changes in bycatch species composition. The information will be collected through informal discussions, questionnaires, or workshops.

Joseph Bonaparte Gulf
 Part of the bycatch-monitoring project will focus on the little-studied region of Joseph Bonaparte Gulf, a unique and important area of the NPF. Some industry members have already collected bycatch samples. Further assistance will be needed in future seasons and your help will be greatly appreciated.



Progress Report March 2006

NPF Bycatch monitoring project

MARCH 2006 *progress report*

The Bycatch Monitoring Project, funded by the Fisheries Research and Development Corporation (FRDC), aims to establish a reliable long-term monitoring program for the Northern Prawn Fishery (NPF).

The project is assessing several methods of collecting data on various bycatch species groups impacted by trawling. The methods are:

- industry-based data collection;
- trained crew member observer;
- directed industry collection;
- logbook return information;
- scientific observers aboard commercially operated vessels; and
- scientific surveys aboard chartered or government vessels.

Research progress

The project is in its final year with the final trial of bycatch-sampling methods taking place this banana prawn trawl season.

After this banana season there will be continued processing of bycatch samples and comparison of the sampling methods. Recommendations will then be made in an FRDC report for consideration by the bycatch project steering committee (research, industry, management and conservation representatives).

What happens after the research project?

The crew member observer program (including training) will run in a scaled-down mode during the 2005 tiger prawn season. This will cover the period at the end of the banana season through to the 2005 fishing seasons.

The recommended bycatch monitoring program may incorporate several data-collection methods, as different methods may prove appropriate for different species groups. There are many factors to consider, such as cost, industry participation, and data accuracy.

The bycatch monitoring program that follows this project could begin with a smaller number of scientific observers and trained crew member observers to minimise industry cost in the short term.

Crew member observers

A training program for the 2004 tiger season and 2005 banana season crew member observers was conducted last year. A press release headed, 'Prawn fisher return makes alive', was issued to coincide with the course to publicise the training and at-sea work of fishers.

Fifteen observers (including six skippers) from 10 trawlers attended the training course. Several observers from the previous year were again involved, providing constructive and helpful feedback.

Sampling was subsequently carried out during the 2004 tiger prawn season and we look forward to processing samples taken during the 2005 banana season.

Directed industry collectors

Fishers in this program are provided with kits to undertake small bycatch sampling operations similar to the crew member observers. Unlike the crew member observers, however, this method involves no training program and looks at a scaled-down sampling operation at set periods during the season.

Vessel skippers and crew were recruited before the 2004 tiger season, with 25 vessels accepting kits and participating in sampling operations.

Port visits will be conducted again before this banana season to recruit directed industry collection vessels for the final assessment of this data-collection method.

Log book returns

The collection of log book data, and other information supplied by NPF fishermen, is ongoing.

Scientific observers

Scientific observers sample small bycatch and gather data about many of the species caught incidentally.

For this project, two scientific observers were at sea during the 2005 tiger season and the

2004 banana season, and will again be at sea aboard industry vessels during the 2005 banana season.

The observer effort has concentrated on bycatch associated with the Joseph Bonaparte Gulf and the banana prawn season of the Gulf of Carpentaria. This is the first comprehensive effort made to collect information of this kind.

Our appreciation

The interest and cooperation of NPF members in the trained crew member observer and directed industry collections has been encouraging.

Some 50 individual vessels have actively supported this research project, and up to 25 NPF vessels have also participated in prawn species distribution research.

Our sincere thanks go to the owners, skipper and crew of these boats for your help and involvement. We look forward to your continued support.

Other useful reading material

Your attention is drawn to the Steve Goyt and Gary Day articles (both AMO), a two-part series about TEDs and BODs in *Professional Fisherman*, June and July 2004.

Also there is an article about TEDs and bycatch monitoring by Bryony Bennett of CSIRO in the October 2004 issue of *Professional Fisherman*.

PROJECT CONTACTS

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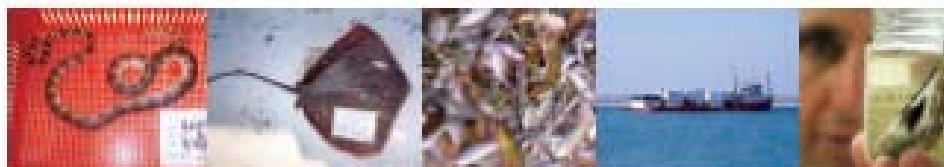


Working towards sustainability



Progress Report April 2004

Bycatch Monitoring Progress Report APRIL 2004



Industry commitment to the Bycatch Monitoring Project has been encouraging, with about half the fleet becoming involved via the crew member observer program and directed industry collection.

Crew member observers (CMO)

In July 2003, crew from 15 vessels participated in a CMO workshop run by CSIRO and AFMA staff on the Gold Coast.

Participants learned to identify and record sawfish and turtle interactions, collect small bycatch samples, identify sharks and rays, photograph sea snakes and estimate total bycatch.

They put their new skills to work during the 2003 Tiger season, collecting 360 small bycatch samples and recording 46 sawfish and five turtle interactions.

The method used to collect sea snake information proved successful and resulted in 1269 sea snakes being labelled and photographed on the back deck with disposable cameras. *Hydrophis elegans* and *Estater*

major were the most common species. (See the end of this report for pictures and information.)

A detailed review of CMO efforts during the 2003 Tiger season will be provided at the next CMO training workshop in July 2004.

If you are not involved in the CMO program and are interested in participating, please contact Reuben Gregor at AFMA on (02) 62725682 for more information.

Directed Industry Collection (DIC)

During pre-tiger season port visits in 2003, CSIRO and AFMA staff recruited crew from 36 vessels to collect small bycatch samples.

NFF Bycatch Sample Kits were provided and vessels were contacted during the season to collect samples just before the new moon.

This method proved successful, with about 170 samples or 570 cartons of well-recorded bycatch collected throughout this season.

From the bycatch samples processed so far, we are able to provide pictures and information for some of the more abundant bycatch species caught in the Gulf of Carpentaria and Joseph Bonaparte Gulf.

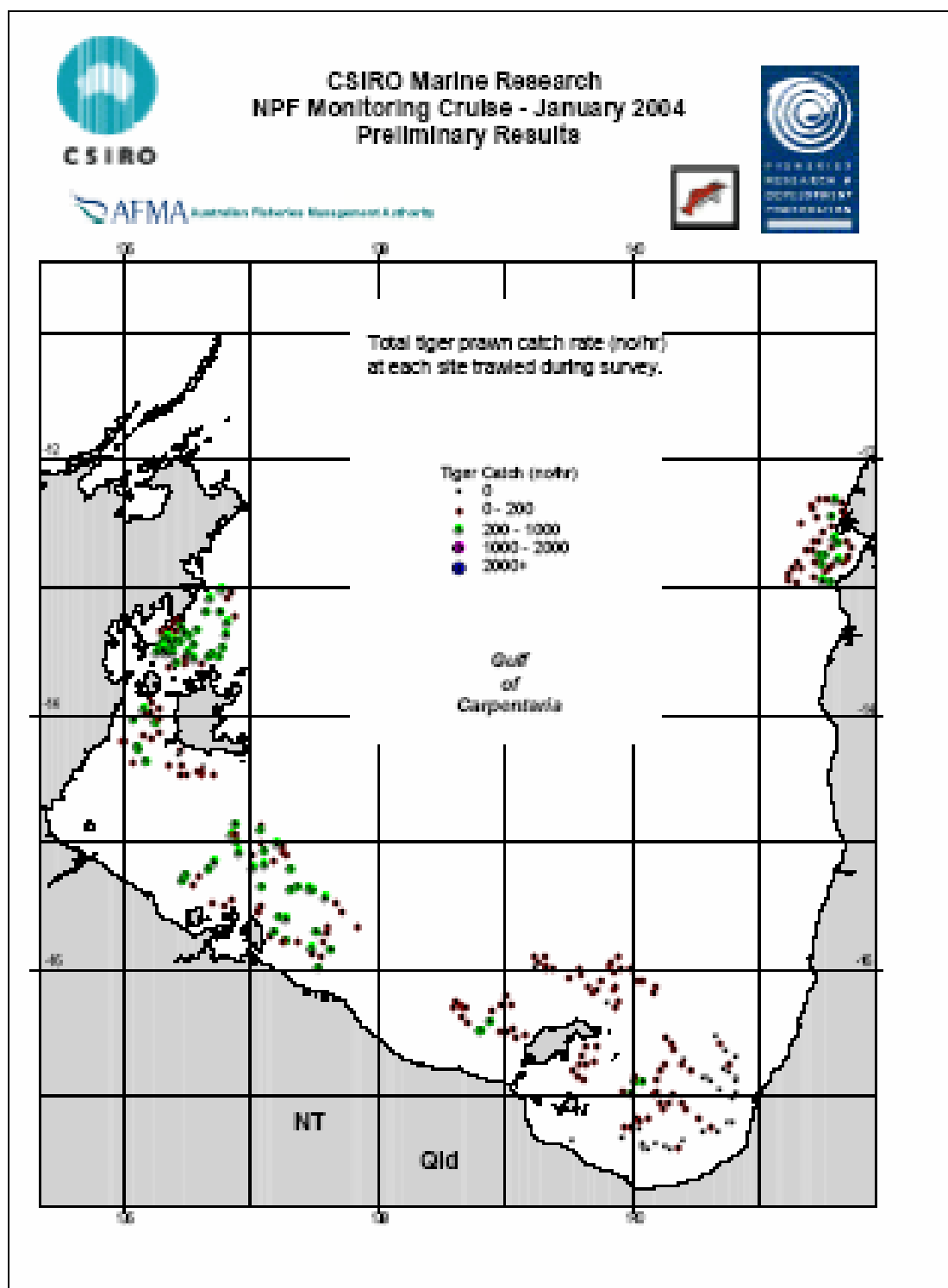
On behalf of all staff associated with this project, we'd like to thank you for your involvement and are looking forward to your continued support.



Working towards sustainability



Example of Preliminary Results from 2004 Monitoring Survey



Project Sustainable Impacts Poster

Moving towards sustainable bycatch populations in Australia's Northern Prawn Fishery

+ David Brewer + Don Heales + Shane Grimtro + Michael Hayward + Peter Jones + Nicole Stobutzki + David Milton + Stephen Blaber



The goal - sustainable fishing
Recent legislation and market drivers have forced or influenced changes in fishing practices in many countries. These often require ecosystem-based approaches to ensure that all species, communities and habitats are not impacted in a way that compromises their long term viability. Australia's extensive Northern Prawn Fishery (NPF) takes about 6000 tonnes of prawns (shrimp) per year. However, catches were dominated by a wide range of unwanted species (bycatch), including small fish, sharks, sturgeons, sea snakes, sea turtles, and benthic invertebrates. The past decade has seen a series of research projects that have focused on reducing impacts on non-target species and designing processes to enable the industry to demonstrate that its practices do not impact the sustainability of any species or marine community. The progress and impacts of this research are described here.

Effort reduction
Since the peak in the number of NPF vessels in 1986, there have been a series of fishing effort reductions aimed at maintaining stocks of the target species of prawns. These initiatives have resulted in a reduced environmental footprint, including large reductions in the impact on the hundreds of vertebrate and invertebrate species caught as bycatch. Changes to fishing patterns during recent years have also resulted in a significant reduction in the area of sea bed trawled by this fleet from 1240 sq nautical miles in 1984 to 804 sq nautical miles in 2000. The number of trawls undertaken by this fleet has also dramatically reduced during this period.

The impact of TEDs and BRDs
The introduction in 2006 of mandatory use of both TEDs (Turtle Excluder Devices) and BRDs (Bycatch Reduction Devices) has seen reductions in the NPF's impact on a range of species groups. Catches of turtles have been reduced by 60% from about 5300 per year to less than 50 per year. Reductions in catches of sharks (21%) and rays (26%) will help reduce the risk to these vulnerable groups, although the impact on individual species ranges from 0% to 100% reduction. Sea fish are not excluded by these devices and remain a concern for the industry. The TEDs and BRDs being used have had little impact on small fish and invertebrates (9% reduction) and sea snakes (no change), resulting in the banning of the main 500 used in the fleet (the Stigge). Losses in prawn catches of between 2% and 6% are expected to reduce in the years ahead as fishers improve their ability to tune the performance of these devices to exclude more bycatch while maintaining or improving prawn catches. Prawn losses are partly compensated for by significant reductions (~40%) in the catches of prawns that are 'lost' or damaged by large unwanted species.

Determining risk for hundreds of species
Demonstrating sustainability for the non-target species is a huge challenge, not only because there are hundreds of different groups, but because of the limited amount of historical, biological and ecological data for most species. Part of the approach involves determining which species are most at risk of being unsustainable under current trawling effort and focusing a monitoring program on these. To this end scientists at the CSIRO Marine Research have developed a modelling approach to assess the level of risk to each bycatch species from trawling by vessels in the NPF. This approach uses a range of parameters relating to the vulnerability (catchability) and recovery of each species with an emphasis on their broad scale geographic distribution inside and outside the fishing zone.

Long term monitoring to demonstrate sustainability
A current project is trialling a range of both fishery-dependent and fishery-independent methods to find the most cost-effective and acceptable solutions for monitoring the wide range of bycatch species impacted by the NPF. Power analyses indicate that a minimum of five years of annual data collection is required to detect any real temporal trends. Consequently, by 2010 the NPF should be able to demonstrate a measure of sustainability for a wide range of bycatch species impacted.

Challenges for the future
Despite the significant progress to address its impact on bycatch, it still remains for this fishery to improve its exclusion rates for sea snakes, sea fish, small fish and where possible, sharks and rays. Continuous improvement in the design and use of TEDs and BRDs will be critical to reducing catches of these species, although new devices or technologies may need to be developed. Understanding the impacts of prawn trawling on benthic communities and ecosystem processes is also underway in the Northern Fisheries and Ecosystems Research Group of CSIRO Marine Research.

1963: Fishery discovery.


1981: 286 vessels in the fleet.


1981-2000: Staged effort reduction from 286 to 100 operating vessels.

2000: Introduction of TEDs and BRDs on all vessels.


2001: Ban on shark finning.


2002-2005: Trial of methods for long-term bycatch monitoring program and Development of risk assessment to determine bycatch species under threat leading to Start of long-term bycatch monitoring program.


2005-2010: Adjustments to fishery practices to reduce impacts on species under threat.

2010
Sustainability of bycatch species demonstrated through ongoing monitoring program.



CSIRO MARINE RESEARCH
Marine Fisheries and Ecosystems Research Group of CSIRO Marine Research
CSIRO Research Infrastructure
Science
Australian Fisheries Management Authority
NPF Industry
Australian Research and Development Corporation

Fishing Boat World Article Jan 2005

Dramatic bycatch reductions in Australia's Northern Prawn Fishery

By BRYONY BENNETT



By introducing TEDs and BRDs, the Northern Prawn fishery has taken a major step towards minimising its impact on bycatch species

Bycatch is being addressed from all angles to support the long-term future of Australia's Northern Prawn Fishery.

The Commonwealth Scientific and Industrial Research Organisation (CSIRO)'s Brisbane marine laboratory is bursting at the seams with bycatch samples collected from trawlers in the Northern Prawn Fishery (NPF).

The latest batch was sent south by 21 volunteer crew members, trained in the art of bycatch sampling and data collection.

Their efforts are part of a bycatch monitoring study that pools the knowledge and resources of CSIRO, the Australian Fisheries Management Authority, the Fisheries Research and Development Corporation and NPF operators.

Results of the three-year study, which began in March last year, will help scientists and fishery managers develop a routine bycatch-monitoring program that will serve the fishery in the long-term.

Bycatch management practices

The NPF is one of Australia's most valuable Commonwealth fisheries, with the catch in recent years averaging 8,000 tonnes of tiger, endeavour and banana prawns valued at about \$100 million.

In the process of trawling for prawns, many non-target species are caught and discarded. These include small fish, sharks, stingrays, sea snakes, sea turtles and benthic invertebrates.

Practices aimed at managing the fishery's impact on bycatch include establishing a bycatch monitoring program for the fishery and compulsory use of turtle exclusion devices (TEDs) and bycatch reduction devices (BRDs). Both practices are the subject of ongoing, industry-supported research.

"Bycatch monitoring is vital to ensuring the sustainability of all species affected by the fishery, not just commercial prawn species," David Brewer of CSIRO says.

"Accurate information is needed to assess the effects of fishing on bycatch, and to respond with management practices that meet national and international standards."

Mr Brewer says crew members participating in the monitoring study have been keen to help the fishery assume responsibility for ecological issues, and ultimately contribute to the industry's survival.

More than half the vessels in the fleet have been involved in some way in the project's first year," he says. "They have gathered information on more than 1,400 sea snakes, and hundreds of other species.

"We have two full-time scientists processing the samples at our laboratory so

we can compare the efficiency, reliability and accuracy of each monitoring technique."

Manager of Perth-based company Newfishing Australia and Northern Prawn Management Advisory Committee member, David Carter, says the industry is keen to play an active role in bycatch monitoring.

"Involving fishing crews in monitoring is a great way to further their understanding of the ecosystems that support the prawns, and the need to maintain them the best possible condition," Mr Carter says.

"The commitment of the NPF to reducing its impact on bycatch was a key factor in the fishery's successful environmental audit by the Federal Government earlier this year."

Testing TEDs and BRDs

In another collaborative project, CSIRO, the Australian Maritime College and NPF operators, with funding from the Fisheries Research and Development Corporation, have assessed the performance of TEDs and BRDs.

TEDs are hard grids placed in trawl nets to exclude turtles and other large animals – many species of sharks, rays, larger fish and sponges – from the trawl. BRDs are escape devices designed to enable smaller animals to escape from the net. From 1997 to 2001, NPF operators were given at sea assistance with rigging, operating and maintaining these devices.

The study combined the skills of NPF operators, fisheries technologists, scientists, economists, statisticians and conservationists.

Five scientific observers gathered information for the assessment during two-week stays on each of 23 NPF vessels. They compared catches between nets with different gear configurations to assess the impact of TEDs and BRDs on a wide range of species groups.

The study concluded that, by introducing TEDs and BRDs, the NPF had taken a major step towards minimising its impact on unwanted species. While the impact varied widely according to species, the capture of turtles and many highest risk sharks and rays had been dramatically reduced.

The exclusion of larger animals made catches easier to handle and sort, reduced danger to the crew, and increased the value of the catch by reducing the crushing of commercial product. While catches of commercially important prawns had been reduced by three to six per cent, damage to prawns by heavy animals was reduced by 40 per cent.

The greatest bycatch reduction was recorded for turtles, whose capture had been reduced from about 3,500 in 1999 to less than 100 in 2000, following the introduction of TEDs and BRDs. All species of turtle are excluded from trawls.



The greatest bycatch reduction was recorded for turtles



Catches of sharks and rays, (elasmobranchs), had been reduced by about 20 per cent and 40 per cent, respectively

Catches of sharks and rays, (elasmobranchs), had been reduced by about 20 per cent and 40 per cent, respectively, but individual species differed from no change (usually smaller species) to greater than 95 per cent exclusion (nine larger species).

The elasmobranch group of greatest concern were sawfish, as these animals are easily tangled in netting. About 80 per cent of the most commonly caught sawfish species was found to be excluded by the TEDs, while the capture of the more rarely caught shallower water species was unchanged.

More than 90 per cent of large sponges were excluded, but it is not known to what extent they survive the trawling process. Their removal improved processing and sorting efficiency.

The smaller bycatch group – hundreds of species of small fish and invertebrates – received little benefit from TEDs and BRDs, with catches reduced by a maximum of eight per cent. This appeared largely due to the type and rigging of BRDs and the most popular of these – the Bigeye BRD – has since been removed from use in the NPF (from mid 2004) due to poor performance.

Neither device excluded sea snakes, but this result could be improved if fishers shifted focus from prawn retention alone to a combination of prawn retention while maximising bycatch reduction.

Economics

Overall, the use of a combined TED and BRD would result in an annual estimated loss of A\$1.2-2.4 million to the fleet (based on a three-six per cent loss of prawn catch and depending on prawn prices). This will vary with prawn price, but in 2001 amounted to an annual cost of A\$12,500-25,000 a year for vessel of average size, but could be reduced with more experienced use of TEDs and BRDs.

Future challenges

Despite making significant progress in lessening its impact on bycatch, the NPF has still to improve its exclusion rates for sea snakes, sawfish, small fish and, where possible, sharks and rays.

"Continuous improvement in the design and use of TEDs and BRDs will be critical, and new devices or technologies may be needed," Mr Brewer says.

"This may require a greater understanding of prawn and fish behaviour during the capture process so that BRDs can be designed that allow fish to swim to freedom while retaining the prawns."

"Bryony Bennett is a communication officer at CSIRO Marine Research in Hobart, Australia."



*Working towards
sustainability*