Estimating the impacts of management changes on bycatch reduction and sustainability of high-risk bycatch species in the Queensland East Coast Otter Trawl Fishery

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Abbreviations

BRD	Bycatch reduction device
DAF	Queensland Department of Agriculture and Fisheries
DOEE	Australian Department of the Environment and Energy (formerly the Department of Environment and Heritage
QECOTF	Queensland East Coast Otter Trawl Fishery
ERA	Ecological risk assessment
GBRMP	Great Barrier Reef Marine Park
GBRMPA	Great Barrier Reef Marine Park Authority
SAFE	Sustainability Assessment for Effects of Fishing
TED	Turtle excluder device

Executive Summary

Researchers from the Queensland Government's Department of Agriculture and Fisheries (DAF) have assessed aspects of the environmental impacts of Australia's largest trawl fishery, the Queensland East Coast Otter Trawl Fishery (QECOTF). A suite of management measures implemented in 2001 have resulted in a reduction in discards from a peak of 87,175 tonnes in 1997 to 25,271 in 2014, or 71%. This reduction was measured using quantitative methods based on catch, fishing effort and swept area. The risk posed to elasmobranchs from trawling operations south of the Great Barrier Marine Park was also assessed using a quantitative method known as Sustainability Assessment for Fishing Effects (SAFE). Of the 47 species assessed, one, the Piked Spurdog, was found to be at high risk. A further six species were found to be at medium risk, with the remainder at low risk. These findings satisfy environmental constraints placed on the QECOTF by the Federal Government, allowing fishers to access fishing grounds within the World Heritage-listed Great Barrier Reef Marine Park. Fishers are also able to continue accessing lucrative overseas markets, increasing the profitability of the fishery.

Background

Tropical prawn trawling accounts for 27.3% of the world's fisheries discards. This is due to the relatively small size of the target species, requiring the use of small mesh which results in an inherently non-selective method of capture. Previous research in Queensland has shown that prawn trawl bycatch is comprised of hundreds of species and includes species of high conservation interest such as sea turtles and sea snakes. This, combined with the impacts of trawl gear on seabed habitats, has resulted in considerable concern regarding the impacts of prawn trawl fisheries on the ecosystems in which they operate.

Significant resources have been expended in trialling technologies that mitigate the incidental capture of species of conservation concern by commercial prawn trawlers. Turtle excluder devices (TEDs) and bycatch reduction devices (BRDs) have been mandatory in north Australian prawn trawl fisheries since the early 2000s and their efficacy has been the subject of several research projects. This research has shown that well-designed TEDs are efficient at removing large animals such as turtles and large rays and sharks from trawls while BRDs allow some fish and sea snakes to escape.

Reducing discards has been a priority in the QECOTF. The QECOTF is a multi-sector trawl fishery primarily targeting prawns, scallops, Moreton Bay bugs, squid and stout whiting using demersal trawl gear. The fishery is divided into eight separate sectors based on the primary target species: tiger/endeavour prawn (TEP), banana prawn, red-spot king prawn (RSK), scallop, Moreton Bay, shallow water (<91m) eastern king prawn (EKP), deep water EKP fishery and stout whiting. In 2015, total landed catch from the QECOTF was 6,382 tonnes valued at $80.9 million. In 2000, the Queensland government implemented significant management changes via the *Fisheries (East Coast Trawl) Management Plan 1999* (hereafter referred to as the "Trawl Plan"), which has reduced nominal effort in the fishery by about 50%.

TEDs and BRDs were also introduced as part of the Trawl Plan. By 2002, all vessels operating in the QECOTF were required to have both a TED and at least one of seven prescribed BRDs installed in their nets. At this time, significant resources were expended in determining the effects of TEDs and BRDs through a dedicated research project conducted by DAF. This project showed that the introduction of TEDs and BRDs resulted in significant decreases in the amount of discards produced in the QECOTF. For example, the use of an efficient TED and a square mesh codend BRD resulted in a 78% reduction of discards produced in the scallop sector in central Queensland or approximately 10,588 tonnes, annually.

This research also demonstrated that TEDs and BRDs are ineffective at reducing the number of predominantly small elasmobranchs (i.e. sharks and rays) caught incidentally by prawn and scallop trawls. Elasmobranchs are an important component of trawl discards and have received increasing attention since the early 2000s: slow growth, low reproductive rate and longevity combine to make these animals susceptible to overfishing. Many of the elasmobranchs caught by trawls operating in the QECOTF are small enough to pass through the bars of a TED, given the required bar space is 12cm. As part of the Trawl Plan, fishers are required to return all elasmobranchs to the sea as quickly as possible and, as such, very little information is available on the sustainability of these animals.

A common method for assessing data-poor species is via ecological risk assessments (ERAs). ERAs use a matrix of fishery impacts (catch, effort, discard rate, etc.) and resilience attributes (growth rate, discard survival, exclusion by TEDs, etc.) to determine the ecological risk posed to discard species by trawling. Over the last decade, more quantitative (data-based) methods have been developed to assess risk. One quantitative method developed by the Commonwealth Scientific and Industrial Research Organisation (CSIRO), known as the SAFE, uses the amount of trawling within a species' distribution as a proxy for fishing mortality, along with metrics for post-trawl survival and escapement from TEDs. Fishing mortality is then compared to the level of sustainable fishing mortality quantified using various life-history characteristics such as growth, age-at-maturity and longevity.

There is a paucity of post-trawl survival (PTS) estimates of elasmobranchs in the scientific literature. This is particularly the case for estimates derived from prawn trawling, with only two studies in the primary literature. The lack of survival studies is a result of the logistical and budgetary issues associated with conducting experiments aboard commercial vessels at sea. To overcome this, several methods have been used to assess PTS in a variety of trawl fisheries including: land-based holding tanks, tagging and laboratory studies. Despite problems associated with keeping animals in tanks for a period of time, the most common method of quantifying PTS is by keeping captured elasmobranchs in on-board tanks for a period of two to three days and assessing survival after this time.

Aims/objectives

The aims of this study are to: 1) Quantify the survival of elasmobranchs (i.e. sharks and rays) that are caught incidentally in Queensland prawn trawl nets and discarded; 2) Quantify reductions in bycatch over the last 20-30 years in the Queensland East Coast Otter Trawl Fishery and describe how these have come about e.g. Fleet reduction, gear technology: and 3) Assess the risk that trawling poses to the sustainability of high risk bycatch species, including elasmobranchs, from the Queensland East Coast Otter Trawl Fishery.

Methodology

The post-trawl survival of two common elasmobranchs, the Common Stingaree and the Eastern Shovelnose ray, was quantified using animals caught during commercial trawling operations. An observer boarded a commercial trawl vessel fishing in the waters off Southport, in southern Queensland, and collected animals as they were found on the sorting tray at the end of each trawl. The animals were stored in a holding tank supplied with seawater via the vessel's deck hose. At the end of each night, the vessel returned to port where the live animals were transferred to one of two large tanks located on the back deck of the Fisheries Research Vessel FRV *Tom Marshall*, where they were monitored for 72 hours.

The reduction in discards achieved through management changes was quantified using discard catch rate data from previous research and from Fisheries Queensland's Fishery Observer Program. From these data, measures of discards as a function of catch, effort and swept area were derived via extrapolation using logbook data. These methods were able to test the effects of various factors such as trawl sector, location, depth, moon phase and, most importantly, TEDs and/or BRDs. The results from these analyses provide the first quantitative estimates of discards produced by all sectors of the QECOTF for the period 1988 – 2014.

The ecological risk posed to 47 elasmobranch species in southern Queensland by trawling was assessed using a quantitative method developed by CSIRO. Using various sources, the distribution of each species was mapped, as was the trawl effort within each species' distribution using high-resolution location data supplied by the Vessel Monitoring System. With these data, it was possible to calculate the length of trawls undertaken by commercial vessels in each of the species' distributions. These lengths were then multiplied by the width of an average trawl to give the total area fished within each distribution. Using other metrics such as gear efficiency, post-trawl survival and escapement rates, fishing mortality was quantified for each species which was then compared to the sustainable levels of fishing mortality for the respective species based on life history characteristics. This comparison informed the level of risk posed to each species by trawling.

Results/key findings

The PTS of the Common Stingaree and the Eastern Shovelnose Ray represent the first short-term PTS estimates from prawn trawls. The mean PTS for female and male Common Stingarees was 33.5% and 17.3%, respectively, and 86.8% for Eastern Shovelnose Rays (both sexes combined). For both species, size and the amount of time out of water were found to affect survival. Female Common Stingarees survived better than males, while the longer trawl times resulted in lower survival of Eastern Shovelnose Rays.

Long-term trends in the production of discards from the QECOTF declined markedly between 1988 and 2014. Total estimated mean discards peaked at 87,175 tonnes in 1997 and declined thereafter to ~25,000 in 2014. The decline has coincided with a marked decline in nominal fishing effort over this period. Factoring in the mandatory introduction of TEDs and BRDs in the early 2000s in the methods used to derive discards also contributed to the reduction, particularly in the scallop sector. Discard catch rates were highest in the shallow sectors such as the banana prawn and shallow water EKP sectors, while the deep water EKP sector produces the lowest discards.

Of the 47 elasmobranch species assessed, the Piked Spurdog was found to be at high risk from trawling in southern Queensland. A further six species (Brown Stingray, Crested Hornshark, Eastern Spotted Gummy Shark, Collar Carpetshark, Sandyback Stingaree and Patchwork Stingaree) were assessed as being at medium risk. In most cases, these species are found in deeper water where fishing effort is high. The two most common species, the Common Stingaree and the Eastern Shovelnose Ray, were found to be at low risk.

Implications for relevant stakeholders

The PTS estimates increase knowledge for the scientific community. The lack of survival estimates in the scientific literature requires attention and the results from the current project will provide additional knowledge regarding this important metric. The survival estimates provide trawl fishers and fishery managers with evidence that elasmobranch survival is variable but not necessarily poor as is generally assumed in qualitative risk assessments.

The long-term decline in discards is likely to be viewed favourably by the public and relevant State and Federal agencies. The estimates also provide a more reliable basis upon which to derive Queensland's and Australia's contribution to global discards – this is noteworthy as the QECOTF is the largest prawn trawl fishery nationally. The reductions in discards will directly contribute to the Great Barrier Reef Outlook Report process and the Wildlife Trade Operation process, allowing for continued access to overseas markets. The reduction in discards is a significant achievement by the prawn trawl industry in Queensland and has significant positive implications for trawl fishers and fishery managers.

Similarly, the ERA conducted as part of the current project satisfies one of the recommendations specified by the Department of the Environment and Energy (DOEE) in granting Wildlife Trade Operation accreditation to the QECOTF. This accreditation allows fishers to access overseas markets, increasing the profitability of the fishery. Seven species, however, were assessed as being at medium or high risk. As such, steps are required to mitigate this risk, particularly for the high risk species, *Squalus megalops*. Given that effort levels in the deep water EKP sector have remained relatively constant, uncertainty around fishing mortality estimates can be reduced by improving escapement from commercial fishing nets and survival estimates, as well as improving life history parameters. An important implication of this work, therefore, is that important metrics used to assess risk are lacking and steps should be taken by Fisheries Queensland to correct this.

Recommendations

1) Data quality is a source of uncertainty in assessing the risk posed to elasmobranchs in the current project. Where possible, steps should be taken to improve life history data and distribution information for all species assessed, particularly for those assessed as being at medium or high risk.
2) The ERA should be extended to the entire east coast of Queensland and, if possible, south into northern New South Wales. (Up until this work, most risk assessments have been focused within the GBRMP).
3) For those species with no published escapement or survival estimates, future ERAs should include estimates for similar species or mean estimates derived from a meta-analyses of published rates.
4) The effect of reducing bar spacing for TEDs on elasmobranchs and other species should be quantified. A decrease in bar spacing would result in fewer elasmobranchs being caught by trawlers.
5) Validation of species distribution information is required.
6) Further ERAs should be undertaken on elasmobranchs and other species (e.g. Syngnathids, Moreton Bay bugs) at a minimum of every five years. Subsequent ERAs should incorporate any relevant updated input data such as escapement or survival estimates. A work group should be convened to decide a list of priority species to be assessed annually made up of shark and ray researchers, conservationists, GBRMPA staff, fishery managers, fishers and DAF researchers.
7) Discard rates and estimates of total annual discards from the QECOTF could be improved if additional discard measures were obtained by re-introducing an observer program.

Keywords

Discards, elasmobranchs, post-trawl survival, *Aptychotrema rostrata*, *Trygonoptera testacea*, turtle excluder device, bycatch reduction device, TEDs, BRDs, ecological risk assessment, ERA

Introduction

It has been estimated that tropical prawn trawl fisheries generate approximately 27% of global discards (Kelleher 2005). This is due to the relatively small size of the target species, requiring the use of small mesh sizes which results in an inherently non-selective method of capture (Griffiths *et al.* 2006). Previous research in Australia has shown that prawn trawl discards are comprised of hundreds of species and includes species of high conservation interest such as sea turtles and sea snakes (Robins *et al.* 2000; Milton 2001; Stobutzki *et al.* 2001b; Courtney *et al.* 2007b; Tonks *et al.* 2008). This, combined with the impacts of trawl gear on seabed habitats, has resulted in considerable concern regarding the impacts of prawn trawl fisheries on the ecosystems in which they operate.

Significant resources have been expended in trialling technologies that reduce discards in Australia's prawn trawl fisheries (e.g. Robins *et al.* 2000). Turtle excluder devices (TEDs) and bycatch reduction devices (BRDs) have been mandatory in northern Australia since 2000 and their efficacy has been the subject of several research projects (e.g. Brewer *et al.* 2004; Courtney *et al.* 2007b; Courtney *et al.* 2010) and numerous scientific studies (see review by Broadhurst 2000). This research has shown that well-designed TEDs are efficient at removing large animals such as turtles and large rays and sharks from trawls while BRDs allow some fish and sea snakes to escape. However, it is very difficult to exclude a high proportion of bycatch species, particularly the small, slow-swimming animals often caught as bycatch when targeting prawns.

The mitigation of discards in Queensland's East Coast Otter Trawl Fishery (QECOTF) has been a priority for two decades. The QECOTF is a multi-sector trawl fishery targeting Penaeid prawns (*Penaeus esculentus*, *P. semisulcatus*, *Fenneropenaeus merguiensis*, *P. monodon*, *Melicertus plebejus*, *Melicertus longistylus*, *Melicertus latisulcatus*, *Metapenaeus endeavouri*, and *Metapenaeus ensis*), sea scallops (*Ylistrum balloti*), scyllarid lobsters (*Thenus parindicus*, *Thenus australiensis*, *Ibacus brucei* and *Ibacus chacei*) and stout whiting (*Sillago robusta*). The fishery can be divided into eight separate sectors based on the primary target species: tiger/endeavour prawn (TEP), banana prawn, red-spot king prawn (RSK), scallop, Moreton Bay, shallow water (<91m) eastern king prawn (EKP), deep water EKP fishery and stout whiting. In 2015, total landed catch from the QECOTF was 6,382 t valued at $80.9million[1]. In 2000, the Queensland government implemented the *Fisheries (East Coast Trawl) Management Plan 1999*, which included the unitisation of effort, resulting in a significant decrease in the number of vessels accessing the fishery: in 2001, 572 vessels landed catch compared to 289 in 2015. This corresponded to a decrease in nominal fishing effort from 67,800 boat-days in 2001 to 33,141 boat-days in 2015, a reduction of ~51%. Since 2000, access to the stout whiting fishery has been restricted to two vessels which, in 2015, landed 786 tonnes from a 1,150 t TAC, worth $1.6 million[2].

To date, the only estimate of discard weight from the QECOTF was derived by Robins and Courtney (1998). These authors estimated that this fishery produced at least 25,000 t of discards. However, this estimate was based on very few datasets and excluded estimates from the EKP, banana prawn and scallop fisheries. Subsequent to this study, the mandatory use of TEDs and BRDs represented a significant change implemented as part of the Trawl Plan. The devices were progressively introduced into all sectors of the fishery between May 1999 and July 2002 and their effect on discards is yet to be quantified across the QECOTF.

Courtney *et al.* (2007b) tested the effects of TEDs and BRDs on the catch rates of target species and discards in the TEP, scallop and EKP fisheries. These authors also reported preliminary estimates of the weight of discards from these fisheries: estimated discards from the scallop, shallow water EKP and deep water EKP fisheries were 13,750 t (Courtney *et al.* 2008), 10,949 t (Courtney *et al.* 2006) and 1,400 t (Courtney *et al.* 2007b) per annum, respectively. These coarse estimates increase the discard estimates reported by Robins and Courtney (1998) to >50,000 t per annum. There is, therefore, a need to quantify discards and assess changes in discards resulting from the management changes implemented as part of the *Fisheries (East Coast Trawl) Management Plan 1999.*

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[1] https://www.daf.qld.gov.au/fisheries/monitoring-our-fisheries/data-reports/sustainability-reporting/queensland-fisheries-summary/east-coast-otter-trawl-fishery
[2] https://www.daf.qld.gov.au/fisheries/monitoring-our-fisheries/data-reports/sustainability-reporting/queensland-fisheries-summary/fin-fish-stout-whiting-trawl-fishery

The introduction of TEDs has likely resulted in a reduction in the number of elasmobranchs landed by prawn trawl gear (Brewer *et al.* 2006). This component of prawn trawl discards has received increasing interest since the early 1990s (Molina and Cooke 2012). Elasmobranchs are characterised by late maturity, few offspring, long life spans and slow growth (Dulvy *et al.* 2008), making them vulnerable to overexploitation (Ellis *et al.* 2008). While the introduction of TEDs has gone some way to decreasing the ecological risk posed to large elasmobranchs by prawn trawling (Kendall 1990; Fennessy 1994; Brewer *et al.* 1998; Gorman and Dixon 2015), numerous studies have shown that the catch rates of smaller species and small individuals of large species remain unaffected (e.g. Brewer *et al.* 2006; Griffiths *et al.* 2006; Courtney *et al.* 2008; Raborn *et al.* 2012).

The introduction of TEDs in the EKP fishery in southern Queensland in 2001 has had little effect on the catch rates of small elasmobranchs (Kyne *et al.* 2002). Research conducted in the early 2000s revealed that the discards in the shallow water fishery include relatively high numbers of batoids (Kyne *et al.* 2002), most of which are small enough to pass through TEDs and into the codend due to the regulated bar spacing in this fishery of 12cm. Kyne *et al.* (2002) reported that the two most common elasmobranchs found in the discarded portion of the EKP catch were the Common Stingaree (Urolophidae: *Trygonoptera testacea*) and the Eastern Shovelnose Ray (Trygonorrhinidae: *Aptychotrema rostrata*).

Trygonoptera testacea and *A. rostrata* are small (<1.2m TL) batoids endemic to Australia's east coast (Last *et al.* 2016). Both species are known to occur to depths of 90-100m, feeding on benthic crustaceans (Kyne and Bennett 2002; Marshall *et al.* 2008). Despite their occurrence in catches, relatively little is known about the life history of either species. As a result, a recent risk assessment (Pears *et al.* 2012) conducted within the World Heritage-listed Great Barrier Reef Marine Park (GBRMP) categorised prawn trawling as posing a high ecological risk to both species. Given that TEDs are ineffective at excluding these two species, Pears *et al.* (2012) stated that a lack of post-trawl survival estimates for these and other species represent the greatest source of uncertainty in assessing the impact of prawn trawling on elasmobranchs within, and adjacent to, the GBRMP.

The post-trawl survival (PTS) of elasmobranchs is poorly understood (Braccini *et al.* 2012; Oliver *et al.* 2015; Dapp *et al.* 2016; Willems *et al.* 2016) despite its importance when assessing ecological risk (Stobutzki *et al.* 2002; Zhou *et al.* 2011). Ellis *et al.* (2017) recently reviewed 79 studies detailing the post-release survival of elasmobranchs and found the majority of studies in the primary literature were conducted in pelagic longline fisheries, while 21 were trawl-related (including beam trawl and scallop dredge) and prawn trawls were the subject of only two studies (Fennessy 1994; Stobutzki *et al.* 2002). The paucity of PTS studies in trawl fisheries is most likely due to the cost and logistical constraints of field-based experiments needed to quantify post-release survival (Musyl *et al.* 2011; Benoît *et al.* 2012; Benoît *et al.* 2013; Dapp *et al.* 2016). Most of the trawl-based field studies assessing the PTS of elasmobranchs have been conducted in northern hemisphere fish trawls (e.g. Revill *et al.* 2005; Rodríguez-Cabello *et al.* 2005; Mandelman and Farrington 2007a; Mandelman *et al.* 2013) and have shown that survival is highly variable between species (Ellis *et al.* 2017). For example, Laptikhovsky (2004) found that PTS of skates (Rajidae) ranged between 0% for *Bathyraja griseocauda* and *B. macloviana* to 71.4% for *B. albomaculata* in the Falkland Islands squid trawl fishery.

Retention of elasmobranchs has been prohibited in the EKP fishery since 2000, with fishers required to return all animals to the sea as soon as practicable. For the most part, the fate of these discards is unknown, complicating the assessment of ecological risk posed to elasmobranchs by trawling. Ecological risk assessments (ERAs) enable fishery managers to quantify the effects of trawling on non-target species to ensure that fishing impacts are sustainable in the long term. Generally, ERAs can identify at-risk species and measures can be developed to reduce impacts.

Qualitative ERAs generally use a matrix of resilience and fishery impacts to measure risk (e.g. Astles *et al.* 2009; Pears *et al.* 2012): resilience is the ability to withstand fishery impacts based on life history characteristics and PTS, while fishery impacts relate to catch levels, discard rates and gear selectivity. This method leads to bias towards an overestimation of the level of risk (Pears *et al.* 2012; Zhou *et al.* 2016). To overcome this deficiency, CSIRO developed quantitative methods to determine the risk posed by prawn trawling (Zhou and Griffiths 2008; Zhou *et al.* 2009; Zhou *et al.* 2011; Zhou *et al.* 2014) for data-poor fisheries. This method estimates the fishing impact, based on the area trawled within a species' distribution, and compares this to sustainability reference points derived from life history characteristics. This method, known as SAFE (Sustainability Assessment for Fishing Effects), has been shown to more accurately reflect the status of assessed stocks when compared to qualitative assessments (Zhou *et al.* 2016).

Pears *et al.* (2012) used the qualitative methods described by Astles *et al.* (2009) to assess the ecological risk posed to elasmobranchs by the QECOTF. This study was conducted in the area within the Great Barrier Reef Marine Park (GBRMP) and found that prawn and scallop trawling posed a high ecological risk to 11 species. This was based on high levels of interaction with the fishery, poor or very poor post-trawl survival and low escapement from TEDs. These attributes, combined with life history characteristics that make the species susceptible to overfishing, led to the high risk categorisation. Where data were unavailable, expertise from a panel of stakeholders informed subjective assessment of metrics such as post-trawl survival and escapement from TEDs. Given the precautionary nature of this risk assessment, conservative estimates of unknown metrics were used to reduce the risk of assigning a lower ecological risk category than the actual value.

All of the species found to be at high risk by Pears *et al.* (2012) were small batoids (<~1m TL). Of these, eight species occur in the EKP fishery, south of the GBRMP, prompting development of the current project. Further, the Federal Department of the Environment and Energy (DOEE) recommended that Fisheries Queensland publish an ecological risk assessment for the area south of the GBRMP. This recommendation was part of an assessment of the QECOTF by DOEE to ensure the fishery is able to export product to lucrative overseas markets as a Wildlife Trade Operation. This accreditation ensures that fishing impacts are sustainable for all species.

In summary, Fisheries Queensland are required to demonstrate the sustainability of the QECOTF. DOEE has recommended that an ecological risk assessment be undertaken for elasmobranchs south of the GBRMP. A previous risk assessment conducted in the GBRMP highlighted that the lack of post-trawl survival information was a significant impediment to accurately determining the risk posed to elasmobranchs and, as such, there is a need to quantify this important metric. Specifically, the need to quantify the post-trawl survival of the two most common elasmobranchs, *Trygonoptera testacea* and *Aptychotrema rostrata*, was highlighted by Fisheries Queensland during project development. Further, the effects of management changes implemented as part of the *Fisheries (East Coast Trawl) Management Plan 1999* on discards, including the effects from the mandatory implementation of TEDs and BRDs in the early 2000s, remains a priority for stakeholders.

Objectives

1. Quantify the survival of elasmobranchs (i.e. sharks and rays) that are caught incidentally in Queensland prawn trawl nets and discarded

2. Quantify reductions in bycatch over the last 20-30 years in the Queensland East Coast Otter Trawl Fishery and describe how these have come about e.g. Fleet reduction, gear technology

3. Assess the risk that trawling poses to the sustainability of high risk bycatch species, including elasmobranchs, from the Queensland East Coast Otter Trawl Fishery

Method

Objective 1 (elasmobranch post-trawl survival)

The post-trawl survival (PTS) of *T. testacea* and *A. rostrata* was assessed during a dedicated experiment conducted off Southport (Figure 1) in southern Queensland (27.82° S; 153.5° E). Southport is a popular port, supporting at least 12 trawlers targeting eastern king prawns (EKP: *Melicertus plebejus*) year-round, although at least 200 vessels operate in the EKP fishery annually. This area was chosen for the survival experiment as past research (Courtney *et al.* 2007b) indicated that both species occur in this area.

The FV *C-Rainger*, a 15.6m steel prawn trawler, was engaged to undertake trawls on known prawn grounds. The *C-Rainger* employed triple gear, consisting of three 12.8m headline Florida Flyer nets, spread by louvre-style otter boards. The body of each net was constructed from ~50mm (2 inch), #36-ply polyethylene trawl mesh, while codends were constructed from ~45mm (1¾ inch), #60-ply polyethylene mesh. All nets were fitted with a top-shooting, single grid hard turtle excluder device (TED) with a bar space of 12cm and a bigeye bycatch reduction device (BRD) as required by legislation. The FRV *Tom Marshall* was chartered to undertake trawls to catch control animals. The *Tom Marshall* is a 14.5m aluminium catamaran, which used a single beam trawl towed from the stern. The beam net was a 6.5m Florida Flyer equipped with a top-shooter Wicks TED with a bar space of 12cm and a fisheye BRD. The body of the net and the codend were constructed from the same materials used on the *C-Rainger*.

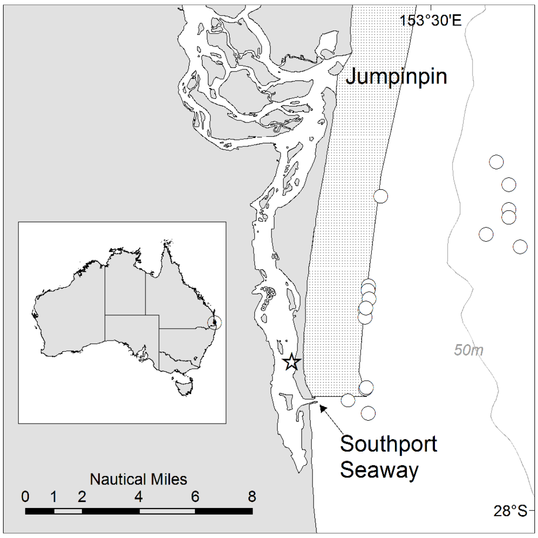


Figure 1: Location of the post-release survival experiments conducted during 2016. The X's represent the starting point of each trawl undertaken by the *C-Rainger* and the star represents the anchorage used by the *Tom Marshall* whilst the animals were housed in the vessel's on-board tanks. The shaded area represents grounds closed to trawling.

Experimental procedure

The PTS of *A. rostrata* and *T. testacea* was assessed during three separate sampling trips. The sampling trips were conducted over five days from 11 March 2016, five days from 28 October 2016 and four days from 15 December 2016, respectively. On the first night of sampling, an observer boarded the *C-Rainger* and collected samples under commercial trawling conditions. At the end of each trawl, sorting commenced: as elasmobranchs were found in the catch, they were tagged with a uniquely numbered streamer tag and removed to a ~150 L holding tank, located on the deck adjacent to the sorting tray, and supplied with flow-through seawater via the vessel's deck hose. Tags were placed at the distal edge of the pectoral fin level with the anterior gill slits. In order to assess the effect of air exposure, time-on-deck was recorded for each individual, quantified as the time, in minutes, between the codend being emptied onto the sorting tray and the time each individual was placed in the holding tank. A qualitative assessment of the condition of each animal, or Condition Index, was also recorded and followed Enever *et al.* (2009), whereby: 1 = Dead or nearly dead, no body movement, slight movement of spiracles; 2 = limp wing movement, some spiracle movement; and 3 = vigorous wing and/or body movement; rapid spiracle movement. The weight of the discards was quantified as a function of the number of baskets (i.e. approximately 30 kg) of discards. Trawl duration, defined as the time between the end of winch-away and the start of haul back, depth and location were also recorded.

This process was repeated for all trawls on the first night until the cessation of fishing before dawn. Once the *C-Rainger* returned to port, live animals were transferred randomly to one of two 1400 L insulated plastic holding tanks aboard the FRV *Tom Marshall*. This vessel was anchored in the Gold Coast Broadwater, close to the Southport Seaway to ensure adequate water quality throughout the holding period (Figure 1). Flow-through seawater was supplied to each container at approximately 36 L per minute. The tag number of any deceased animals was recorded before the deceased animals were stored in the vessel's on-board freezer for later examination. This process was repeated on the following night.

Also on the first night, the *Tom Marshall* undertook 20 individual trawls using the 5-metre beam. The objective of these trawls was to collect 'control' individuals of both species: short (~10-20 minutes) trawls were conducted in relatively shallow (~25m) water, minimising factors known to affect post-trawl survival such as total catch weight and trawl duration (Fennessy 1994; Enever *et al.* 2009). All individuals were tagged with a streamer tag before being placed in a 150 L insulated fish box supplied with flow-through seawater. Total air exposure was limited where possible. At the cessation of trawl operations, the *Tom Marshall* returned to port and anchored in the Gold Coast Broadwater close to the Southport Seaway and all live 'control' animals were transferred randomly to one of the two 1.4 t on-board holding tanks.

For all three sampling trips, the animals were monitored and, in accord with previous studies (Mandelman and Farrington 2007a; Enever *et al.* 2009; Mandelman *et al.* 2013), survival was assessed after three days (72 hours). During that time, the tanks were inspected every two hours and deceased animals removed and stored in the vessel's on-board freezer. For each animal, total length for *A. rostrata* or disc width for *T. testacea* (in millimetres) were recorded, along with the unique tag number. At the end of each sampling trip, all live animals were returned to the sea after the streamer tags were removed.

Statistical analyses

In accord with methods described by Campbell *et al.* (2014), PTS was quantified using generalised linear modelling (GLM) via a binomial distribution with a logit link function, where survival (a binary variable with 0 = dead and 1 = alive) was the response variable. Separate models were developed for each species. For each model, several categorical factors were added to assess their effect on PTS: sampling trip (1, 2 or 3); control (0 = no, 1 = yes); and gender (male, female). Covariates for total length or disc width, trawl duration, total discards and time-on-deck, transformed using either their natural logarithm or a square-root transformation, were also tested. Statistical analyses were performed using 'R' statistical software (R Core Team 2016). Appropriate models were determined via the "step" function.

In accord with Enever *et al.* (2009), a second model was developed to determine the correlation, if any, between Condition Index and PTS for each species. The only variable tested in this model was Condition Index as a categorical term with three levels ('1', '2' or '3').

Objective 2 (discard estimation)

Definitions

Our definition of discards is similar to that used by Kelleher (2005), which was adopted from the FAO Fisheries Report No. 547 (FAO 1997). We define discards as the weight of organic matter that is returned to the sea by the fishers. This differs slightly from Kelleher's as we include coral rubble and plants (i.e. algae and seagrass), which comprise a very minor component of the discards in Queensland (i.e. < 1%). Our definition also includes the weight of discards returned to the sea by researchers during experimental charters designed to undertake research on discards and bycatch reduction devices (BRDs)..

We refer to retained catch weight, which is composed of the 'principal species' (i.e. targeted species) and 'permitted species' (i.e. byproduct), as defined in the QECOTF Management Plan[1]. Principal species are penaeid prawns (*Penaeus* spp., *Melicertus* spp., *Metapenaeus* spp., *Fenneropenaeus* spp.), saucer scallops

[1] https://www.legislation.qld.gov.au/LEGISLTN/REPEALED/F/FisherECTMP99_02K_031219.pdf

(*Ylistrum balloti*), scyllarid lobsters (*Thenus* spp.) and stout whiting, *Sillago robusta*). Permitted species are portunid crabs (*Portunus* spp.), Balmain Bugs (*Ibacus* spp.), cuttlefish (*Sepia* spp.), barking crayfish (*Linuparus trigonus*), octopus (*Octopus* spp.), squid (*Loligo* spp.), mantis shrimp (*Squilla* spp., *Oratosquilla* spp.), pipefish (*Solegnathus* spp.) and threadfin breams (*Nemipterus* spp.).

Discards can include target and byproduct species that are not retained for various reasons, as determined by individual fishers (i.e. undersize or sub-optimal individuals, poor prices or low demand, small or inconsistent catches which are difficult to market). About 1300 taxa have been reported in the fishery's discards (Courtney *et al.* 2006; 2007b; 2008; 2014).

Estimating total discards

The Queensland East Coast Otter Trawl Fishery (QECOTF) can be stratified into eight sectors, based on location, target species, fishing gear and depth (Table 1). Stratifying the fishery increases the amount of variation in discard rates that can be explained, as each sector has its own discard assemblages and catch rates which are related to the physical, biological and environmental features of the sector (i.e. sediment, salinity, depth, temperature, chlorophyll, etc.), as well the type of trawl fishing gear used.

Three methods for quantifying the production of discards in each sector were compared: 1) retained catch method, where discard weight (kg) was estimated as a function of the retained catch (kg) reported in the logbook database, 2) effort method, where discard weight (kg) was estimated as a function of fishing effort (i.e. boat-days), and 3) swept area method, where discard weight (kg) was estimated as a function of the bottom area (ha) swept by the fleet's trawl gear.

Retained catch method

Estimating discards as a proportion of retained catch is the most common method used to quantify discards, and was used by Kelleher (2005) to derive global estimates. This is because catch is the only available metric in many fisheries. In the present study, the proportion of discards to total catch (i.e. retained catch + discards) was based on 3436 paired measures of discards and retained catch in QECOTF sectors between 1996 and 2010. The data were obtained by fishery observers and researchers aboard commercial trawlers that were undertaking their normal fishing activities, as well as research charters that were designed to evaluate BRDs. Further details of the data and sampling methods used to collect these data can be found in Robins *et al.* (2000), Stobutzki *et al.* (2000) and Courtney *et al*. (2006; 2007a; 2007b; 2008; 2014).

The weight of discards and retained catch were measured and recorded to the nearest kilogram. In addition, for each paired measure the following potential explanatory terms were recorded;

1) sector (categorical term, 8 levels from Table 1);
2) sampling trip (unique categorical blocking term for each sampling excursion, which ranged between 2 and 10 days);
3) trawl shot number (unique categorical blocking term for each completed deployment of the trawl gear which consisted of towing 1-5 nets simultaneously);
4) lunar phase (covariate based on lunar luminance);
5) lunar phase advanced 7 days (same as lunar phase except the phase is advanced 7 days to account for the waxing or waning phase);
6) sampling program (a binary term differentiating commercial fishing and research charters) and
7) presence of a BRD in the net/s (binary term where a BRD was defined as a turtle exclude device (TED) or any other device designed for excluding bycatch, or both).

A logistic regression was applied to the data, using the number of kilograms of discards (successes, *x*) and the number of kilograms of total catch (retained catch + discards) for each observation. The above terms were added to the following model to assess their impacts on the discard proportion:

$$\ln\left(\frac{p(x)}{1-p(x)}\right) = \beta_0 + \mathbf{X}\boldsymbol{\beta} + \varepsilon$$

where $p(x)$ is the probability that the dependent variable equals a case (discards), given some linear combination of the explanatory terms. $\mathbf{X}$ includes all the explanatory terms (i.e. sector, sampling trip, trawl shot number, lunar phase, sampling program and BRD), $\boldsymbol{\beta}$ is the parameter vector to be estimated and ε is the error term.

Table 1: Details and descriptions of the eight sectors that comprise the Queensland east coast otter trawl fishery (QECOTF).

Sector	Target species	Details
Tiger and endeavour prawn	Brown tiger prawn *Penaeus esculentus*, grooved tiger prawn *Penaeus semisulcatus*, blue tail endeavour prawn *Metapenaeus endeavouri* and red endeavour prawn *Metapenaeus ensis*	11.0-22.0°S. Generally in depths < 30m. Maximum net size[1] of 88m.
Red spot king prawn	Red spot king prawn *Melicertus longistylus* and minor catches of blue-legged king prawn *Melicertus latisulcatus*	11.0-22.0°S. Generally in depths 30-70m. Maximum net size[1] of 88m.
Shallow water eastern king prawn	Eastern king prawn *Melicertus plebejus*	22.0-28.3°S in depths < 91m (50 fathoms). Maximum net size[1] of 88m.
Deep water eastern king prawn	Eastern king prawn *Melicertus plebejus*	22.0-28.3°S in depths ≥ 91m (50 fathoms). Maximum net size[1] of 184m.
Banana prawn	Banana prawn *Fenneropenaeus merguiensis*	16.0-28.3°S in < 30m, mainly February and June. Maximum net size[1] of 88m.
Saucer scallop	Saucer scallop *Ylistrum balloti*	22.0-27.0°S in 20-60m. Maximum net size[1] of 109m. Minimum codend mesh size of 75mm.
Moreton Bay	Greentail prawn *Metapenaeus bennettae,* brown tiger prawn *P. esculentus* and juvenile eastern king prawn *M. plebejus*	Within Moreton Bay. Maximum vessel length of 14m. Maximum net size[1] of 32.5m.
Stout whiting	*Sillago robusta*	24.4-28.3°S in depths of 37-91m (20-50 fathoms). Five licenses. Managed using annual total allowable catch (TAC). Gear restricted to single otter trawl with 128 m sweeps or Danish seine with 2x2500 m haul ropes.

[1] Net size refers to the combined length of the head rope and foot rope for all nets used on the vessel.

The derived discard rates, expressed in kg.kg^{-1} of retained catch, were then multiplied by the annual reported catch for each sector from 1988-2014 using logbook data.

Effort method

Generalised Linear Modelling (GLM) was used to examine the potential effects of the above terms on discard rates,

$$U = \beta_0 + \mathbf{X}\boldsymbol{\beta} + \varepsilon$$

where U is discard weight boat-day^{-1}. The same data were used for this method except that the research charter data were omitted. This is because the charters produced fewer discards per boat-day than an average commercial trawling day due to the need to impose experimental control, which included increased travel/steaming to and from predetermined sites (hence less trawling time), undertaking trawls of shorter duration to obtain adequate replication, and time spent swapping experimental gears (i.e. BRDs and codends) between trawls.

Discard rates (kg boat-day^{-1}) were calculated by summing discards from all trawls from a vessel on a given boat-day. For all sectors, except the banana prawn sector, which operates predominantly during daylight hours, a fishing day was defined as 12:00pm to 12:00pm. For banana prawns, a fishing day was defined as 12:00am to 12:00am. Because the data were restricted to commercial vessels undertaking normal fishing activities and summed to whole boat-days, the number of observations was reduced to 381 boat-days. Summing data to whole boat-days meant variation between trawls could not be considered and so the 'trawl shot number' term was dropped from the model. For sampling trips on board vessels where a BRD was installed in half of the nets and not installed in the other half (e.g. for comparative reasons), the observations from each net were doubled to represent whole boat-days.

Total discards for each sector were calculated by multiplying the discard rate (kg boat-day^{-1}) by the number of boat-days in each month and year from 1988-2014 based on logbook data.

Swept area method

Discard weights were obtained from 4,012 trawls from individual nets. These are the same data as those used for the retained catch method, except the data frame was restructured to include information on individual net deployments, specifically net head rope length, bottom trawl duration and discard weight, where this information was recorded. As the swept area method does not require retained catch, some additional observations that were not included in the retained catch method data were added. The research charter data were also included in this model (unlike the effort model) because the charters used commercial fishing nets and undertook all sampling on commercial fishing grounds. Hence, there was no reason to assume any difference between the catch rate of discards (kg ha^{-1}) from commercial vessels undertaking normal fishing activities and the research charters. The swept area $SA_{i,n}$ of each net n, from trawl i, was calculated as:

$$SA_{i,n} = L_{i,n}S_{i,n}h_{i,n}f_{i,n}/10{,}000$$

where $L_{i,n}$ is the net head rope length in metres; $S_{i,n}$ is the speed of the trawl in metres h^{-1}; $h_{i,n}$ is the bottom trawl duration in hours (h). $f_{i,n}$ is a trawl net spread factor from Sterling (2000) of 0.650, 0.704, 0.794, 0.764, 0.704 for single, twin, triple, quad and five gear, respectively. Division by 10,000 converts square metres to hectares (ha).

The stout whiting fishery is a minor component of the QECOTF. Unlike the other sectors, it is managed by a total allowable catch (TAC) which is allocated among five licenses that are owned by two fishers. In the early 1990s, these license holders harvested whiting on trawlers using a single prawn trawl net with extended sweeps of 128 m, but in recent years Danish Seine has become the main method. The swept area for the fish trawl was estimated to be the sum of the combined sweeps and net head rope lengths multiplied by a spread factor of 0.289. The Danish Seine uses two sweeps each with a maximum regulated length of 2500 m and was estimated to sweep 110.5 ha per deployment.

The discard rate (kg ha^{-1}), $D_{i,n}$, was calculated as: $D_{i,n} = d_{i,n}/SA_{i,n}$, where $d_{i,n}$ and $SA_{i,n}$ are the discard weight (kg) and swept area (ha), respectively, for net n, from trawl i. This ratio was modelled using a GLM which is similar to Equation (2), resulting in a mean rate (kg ha^{-1}) for each sector.

The area swept by the fleet in each sector was calculated as the product of the mean area (ha) swept boat-day^{-1} and fishing effort (in boat-days). Mean area (ha) swept boat-day^{-1} was calculated using survey data on the number and size of nets towed by fishers and their trawl speeds (O'Neill and Leigh 2007) from 1988-2014, as well as the abovementioned spread factors. The mean daily trawl duration (i.e. length of time that nets were towed along the bottom, hr) was derived and based on on-board observations of commercial vessels when the discard measures were obtained and information on hours-trawled provided by fishers in their daily logbook records.

The mean daily swept area (ha boat-day^{-1}) for each month and year was then multiplied by the corresponding effort (boat-days) to produce the total swept area in each sector. Confidence intervals (95%) for the means were based on 100,000 bootstrap samplings of vessels. The mean area swept (ha boat-day^{-1}) was then multiplied by the mean discard rate (kg ha^{-1}) to derive the total discards for each month and year from 1988-2014. The standard error (SE) for the product of the two means was calculated for each sector s_1 thus:

$$\mathrm{SE}(s_1) = \sqrt{(\bar{x}_1 \text{ x } se_2)^2 + (\bar{x}_2 \text{ x } se_1)^2 + (se_1 \text{ x } se_2)^2}$$

where $\overline{x}_1$ is the mean discard rate (kg ha^{-1}) and $\overline{x}_2$ is the mean daily swept area (ha boat-day^{-1}), and se_1 and se_2 are their respective standard errors.

Annual discards for each sector were derived using the mean of the three estimates (retained catch, effort and swept area). Total annual discards for the whole QECOTF were obtained by summing sector means. Standard errors for the whole fishery (w) were estimated according to Snedecor and Cochran (1967) :

$$\mathrm{SE}(w) = \sqrt{\mathrm{SE}(s_1)^2 + \mathrm{SE}(s_2)^2 + \mathrm{SE}(s_3)^2 + \ldots + \mathrm{SE}(s_n)^2}$$

where s_1, s_2, $s_{3\ldots}$ s_n are individual sectors.

Objective 3 (risk assessment)

The QECOTF south of the Great Barrier Reef Marine Park (GBRMP) represents approximately 30,455 km^2. The fishery is divided into six separate sectors (Figure 6) based on the target species: banana prawn (*F. merguiensis*), tiger prawn (*P. esculentus*), saucer scallop (*Ylistrum balloti*), shallow water (<50fathoms/91 m) eastern king prawn (EKP, *Melicertus plebejus*) and deep water EKP fishery. The stout whiting (*Sillago robusta*) fishery occurs within the shallow water EKP fishery. Previous research (Dodt 2005; Courtney *et al.* 2007b), Queensland Fisheries' Fishery Observer Program (FOP) and Queensland Museum data indicate that at least 47 species of elasmobranch occur south of the GBRMP (see, Table 7 page 46 for a complete list of species assessed).

The ecological risk posed to these 47 elasmobranchs by prawn trawling in the area south of the GBRMP (hereafter referred to as 'the fishery') in 2015 will be quantified using the enhanced SAFE (eSAFE) method described by Zhou *et al.* (2014). The SAFE method was developed to assess the risk posed to species discarded from the catch in the Northern Prawn Fishery (NPF) and is now used in several Australian fisheries (Zhou and Griffiths 2008; Zhou *et al.* 2009; Zhou *et al.* 2011; Zhou *et al.* 2013 ; Zhou *et al.* 2014; Zhou *et al.* 2015) and is appropriate for data-poor fisheries.

The general approach of these quantitative ERAs is to estimate the spatial overlap between species distribution and the distribution of fishing effort, catchability, exclusion as a result of TEDs/BRDs and post-release survival. This is then compared to various sustainability reference points to assess risk for each species in any given year. The sustainability reference point of interest is F_{msm}, the instantaneous fishing mortality rate that corresponds to the maximum sustainable yield (Zhou *et al.* 2011; Zhou *et al.* 2015).

Fishing mortality

In accord with Zhou *et al.* (2015), the instantaneous fishing mortality, F, for species x for each gear type g is:

$$F_{x,g} = \frac{C_{x,g}}{\overline{N_{x,g}}} = \frac{Q_{x,g}(1 - E_{x,g})(1 - S_{x,g})A_{t,x,g}}{A_x}$$

where: C_x is number of individuals of species x dying as a result of interaction with trawl gear in 2015; $\overline{N_x}$ is the mean population size of species x in 2015; Q_x is the gear efficiency; E_x is the escapement rate resulting from the presence of a turtle excluder device (TED) if required; S_x is the discard survival rate; $A_{t,x}$ is the area trawled by each gear type within the distribution of species x south of the GBRMP; and A_x is the area over which species x is distributed south of the GBRMP. In the current study, the impacts of four gear types are assessed: prawn trawl, scallop trawl, stout whiting trawl and Danish seine.

Following Zhou *et al.* (2014), Q_x is estimated from data collected from past research (Dodt 2005; Courtney *et al.* 2007b) and from the Queensland Government's Fishery Observer Program. These data provided catches from the four gear types in each 30 minute by 30 minute CFISH grid (~cell). The distribution of each species was modelled at between-cell and within-cell levels using the Poisson function in a Bayesian framework. WinBUGS (Spiegelhalter *et al.* 1999) software was used to estimate Q_x for each gear type (see R code on page 44). Where parameters were estimated by this process, non-informative prior distributions were used.

Escapement (E) and discard survival rate (S) of each species were determined from previous research, particularly Stobutzki *et al.* (2002), Brewer *et al.* (2006), Courtney *et al.* (2007b) and Zhou and Griffiths (2008), where available. Survival rates derived from the current project were used for *A. rostrata* and *T. testacea*. $E = 0$ and $S = 0$ were used for those species where there is no published survival or escapement data.

The area trawled by each gear ($A_{t,x,g}$) was calculated using TrackMapper (Good *et al.* 2007), a program designed to provide high spatial resolution of trawl catch and effort data. This program uses Vessel Monitoring System (VMS) data to construct trawl tracks for vessels operating in the fishery. The area south of the Great Barrier Reef (GBR) was divided into separate fishing sectors based on TrackMapper data (Figure 6a): the trawl track data incorporates information regarding species composition and catch weight, as recorded in the logbook database. The length of each trawl was quantified in ArcMap (ESRI 2011) and summed with respect to sector to produce total trawl length in kilometres. The total length of trawls within each sector were then multiplied by the mean trawl width of trawls within each sector to produce the area trawled by each gear ($A_{t,x,g}$). Mean trawl width in each sector was calculated from logbook data gear sheets submitted by fishers.

Table 2: Mean trawl width in each fishery. These means were derived from logbook data and spread ratios reported by Sterling (2000).

Fishery	Mean trawl width (m)
Banana	20.5
Deepwater EKP	50.6
Scallop	34.3
Shallow water EKP	30.0
Tiger	10.3

The area fished by the stout whiting fish trawl vessel and the Danish seine vessel in 2015 was calculated using compulsory logbook information. The area trawled by the fish trawl vessel was calculated from logbook data as A = 85 x speed x duration/1,000,000, where board spread is 85m (Cliff Patrick, skipper FV *Caledon Pearl*, pers. comm. 8/10/2015), resulting in the swept area in square kilometres. Each Danish seine shot was calculated as being 1.1km^2 after discussions with the skipper of the FV *San Antone*, Michael Pinzone.

The area over which each species occurs throughout the fishery (A_x) was derived primarily from two sources: Last and Stevens (2009) and the Atlas of Living Australia (ALA; http://www.ala.org.au/). Using these resources, distribution maps have been constructed for 47 species of elasmobranch which occur south of the GBR (see Figure 6b and Figure 6c for examples). The area of each species distribution was quantified using ArcMap (ESRI 2011). The data in Table 7 on page 46 informed the distribution of each species.

The cumulative impact of all fishing gears in 2015 is given by:

$$F_x{}^C = \sum_g F_{x,g}$$

It should be noted that Moreton Bay has been deliberately excluded from these analyses. This is due to the fact that M2-licenced vessels are not required to have VMS and the methods used in the current study are not applicable.

Reference points

In accord with Zhou *et al.* (2011), we use three biological reference points to assess the risk posed to 48 elasmobranchs caught in the fishery. The reference points are as follows:

- F_{msm}: instantaneous fishing mortality rate corresponding to the number of fish in the population that can be killed by fishing in the long term or maximum sustainable mortality (MSM). This corresponds to the biomass that supports MSM (or B_{msm}). F_{msm} is similar to MSY for a target species in a stock assessment;

- F_{lim}: instantaneous fishing mortality rate that corresponds to the biomass B_{lim} which represents $0.5B_{msm}$; and

- F_{crash}: minimum unsustainable instantaneous fishing mortality rate that will lead to population extinction in the long term.

These reference points are a function of the life history characteristics of each species such that $F_{msm} = \omega M$, where $\omega = 0.41$ for elasmobranchs based on empirical data (Zhou *et al.* 2012) and M is the instantaneous rate of natural mortality. Further, $F_{lim} = 1.5\omega M$ and $F_{crash} = 2\omega M$. Where possible, 5 methods were used to estimate M (Table 3).

Table 3: Five methods used to estimate the instantaneous rate of natural mortality (M) for use in the derivation of biological reference points F_{msm}, F_{lim} and F_{crash}. L_∞ and k are the von Bertalanffy growth parameters, T is the average annual water temperature, t_{mat} is the average age-at-maturity, t_{max} is the maximum age.

Equation	Reference
$\ln(M) = -0.015 - 0.279\ln(L_\infty) + 0.6543\ln(k) + 0.4634\ln T$	Pauly (1980)
$\ln(M) = 1.44 - 0.982\ln(t_{max})$	Hoenig (1983)
$M = 10^{0.566-0.718\ln(L_\infty)} + 0.02T$	http://www.fishbase.org
$M = 1.65/t_{mat}$	Jensen (1996)
$M = 4.118k^{0.73}L_\infty{}^{-0.33}$	Then *et al.* (2014)

The relevant life history characteristics were obtained from the primary literature (Table 8 page 47): however, many species assessed had no published growth data. In these cases, any relevant data were sourced from the Fishbase life history tool on the respective species pages (http://www.fishbase.org).

Assessing risk posed by trawling

The instantaneous rate of fishing mortality is compared to the mean of the five F_{msm} values to categorise risk (see Table 4). Further, given the uncertainty around the instantaneous rate of natural mortality, the range of M estimates are used to quantify precautionary risk categories, such that:

- Precautionary medium risk: $F > \min(F_{msm})$ OR $F + 90\%\text{CI} \geq F_{msm}$

- Precautionary high risk: $F > \min(F_{lim})$ OR $F + 90\%\text{CI} \geq F_{lim}$

- Precautionary extreme high risk: $F > \min(F_{crash})$ OR $F + 90\%\text{CI} \geq F_{crash}$

Table 4: Biological reference points, proposed risk category and ecological consequence used for the current project as reported by Zhou *et al.* (2011).

Metric	$F < F_{msm}$	$F_{lim} > F > F_{msm}$	$F_{crash} > F > F_{lim}$	$F > F_{crash}$
Risk	Low	Medium	High	Extreme high
Ecological consequence	Overfishing not occurring. May keep population above 50% of virgin level	Overfishing is occurring but population is sustainable	May drive population to very low levels in longer term	Population is unsustainable in long term – possibility of extinction

Results

Objective 1 (elasmobranch post-trawl survival)

The *C-Rainger* completed 18 trawls during the three sampling trips (Table 9, page 49), with trawl duration ranging from 64 minutes to 217 minutes. All trawls were undertaken on trawl grounds that receive considerable fishing effort in depths between 27 and 54 metres. Apart from *T. testacea* and *A. rostrata*, very few elasmobranchs were caught: six maskrays (*Neotrygon* spp.) and three coffin rays (*Hypnos monopterygius*) were caught during sampling. The maskrays and the coffin rays were discarded alive on capture.

The *Tom Marshall* completed 36 control trawls, ranging in duration from 10 minutes to 20 minutes (Table 9, page 49). All trawls were conducted adjacent to the Southport Seaway (Figure 1) in depths between 19 and 47 metres (mean = 27.2m, S.E. = 1.0m). The only other elasmobranch species caught by *Tom Marshall* was the Kapala stingaree (*Urolophus kapalensis*): two individuals, 113mm and 210mm DW, were caught during the second and third sampling trips, respectively, the larger of which aborted two pups (78mm and 79mm DW) during the sorting process. The two larger animals were tagged and retained, while the pups died soon after capture.

Trygonoptera testacea

A total of 187 *T. testacea* were assessed for post-trawl survival (PTS) during the three experiments (Table 9, page 48), ranging in size between 75mm and 245mm disc width (Figure 9a). Of the 103 *T. testacea* caught on the *C-Rainger*, 68 (66%) died, as did 56 of the 84 (67%) caught on the *Tom Marshall*. Females were more prevalent than males (Figure 9b), with 109 and 73 caught, respectively; and females were significantly larger than males (t = 2.8776, P < 0.01). For *T. testacea*, time-on-deck ranged between 1 minute and 28 minutes (μ =14.5 minutes; σ = 7.1). *T. testacea* were more abundant on inshore grounds: the *C-Rainger* caught none on deeper (~50m) trawl grounds during the first experiment (Figure 1). Of the 187 *T. testacea* caught, 11 were given a Condition Index of '1', while 71 were given a score of '3' (Table 10, page 49). Generally, individuals were in good condition when placed in the holding tanks: however, the area around the tag site became infected in about a quarter of the animals retained during the second experiment.

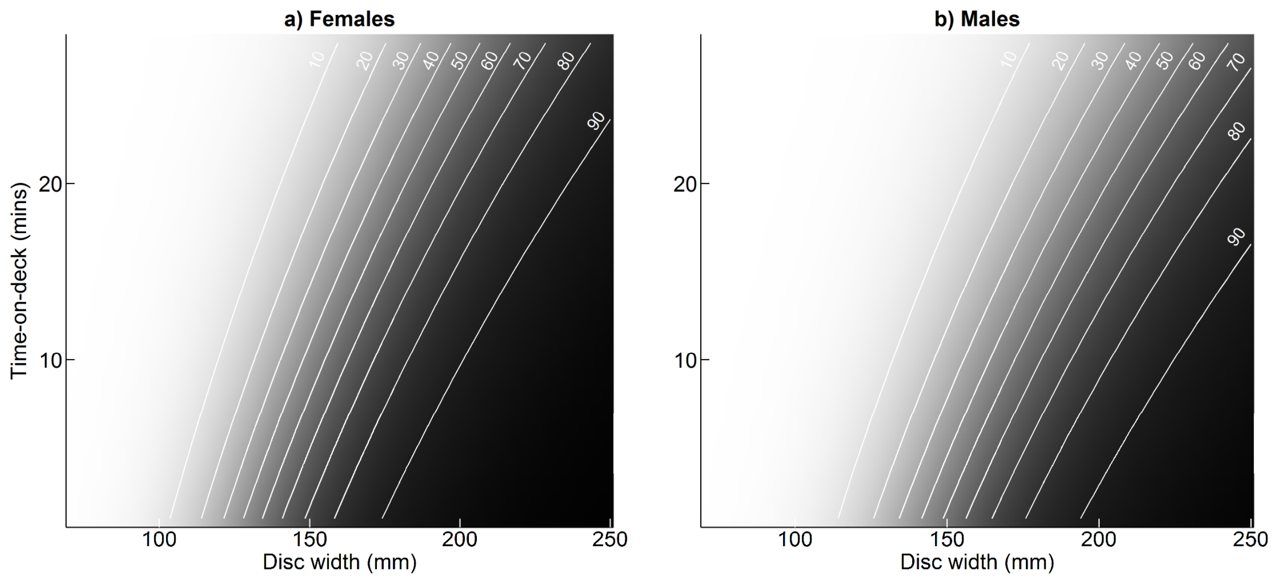


Figure 2: Post-trawl survival of Common Stingarees (*Trygonoptera testacea*) as a function of disc width (mm) and time-on-deck (minutes) for a) females and b) males caught during two sampling trips conducted in southern Queensland, Australia.

Given no *T. testacea* were caught on the *C-Rainger* during the first sampling trip, it was excluded from the analysis. There was no significant difference (P = 0.346) in PTS between the second and third trips so data from the two trips were pooled. Survival of the controls during the two trips did not differ significantly (P = 0.603) from that of those caught on the *C-Rainger* and trawl duration had no effect on PTS (P = 0.154). The natural logarithm of disc width was the best predictor of survival ($P < 0.001$), with survival increasing (Figure 2) with size (β = 8.598, S.E. = 1.454). Both gender (P = 0.05) and time-on-deck (P = 0.05) had marginally significant effects on survival. The survival of females was higher than males (β = 0.895, S.E. = 0.461), while increasing time-on-deck was found to result in lower survival (β = –0.109, S.E. = 0.057). Given the confounding effect of females having a larger mean disc width than males (Figure 2), a term representing the interaction of gender and size was added to the model and the two first-order terms were dropped. The final reduced model included only the interaction term and the time-on-deck covariate, both of which had a significant effect ($P < 0.01$) on PTS (Figure 2). Mean overall PTS for female *T. testacea* was 33.5% (S.E. = 6.0%) and 17.3% (S.E. = 5.5%) for males. Condition Index did not affect PTS despite a trend in higher survival for increasing levels of condition (Table 10).

Aptychotrema rostrata

Of the 155 *A. rostrata* assessed for PTS during the three experiments, 118 (~76%) were caught aboard the *C-Rainger* (Table 9 page 48). Of these, 24 (~20.3%) died while only one of the 37 (2.7%) controls caught on the *Tom Marshall* died. *Aptychotrema rostrata* ranged in size between 166mm and 555mm TL (Figure 10a). Females and males were equally represented in catches (n = 78 and n = 77, respectively) and their size (Figure 10b) did not differ significantly (t = –0.321, $P > 0.1$). Time-on-deck ranged between 0 minutes and 63 minutes (mean = 18.4; S.D. = 12.0). Of the 155 *A. rostrata* caught, 12 were given a Condition Index of '1', 101 a score of '2' and 42 a score of '3' (Table 10).

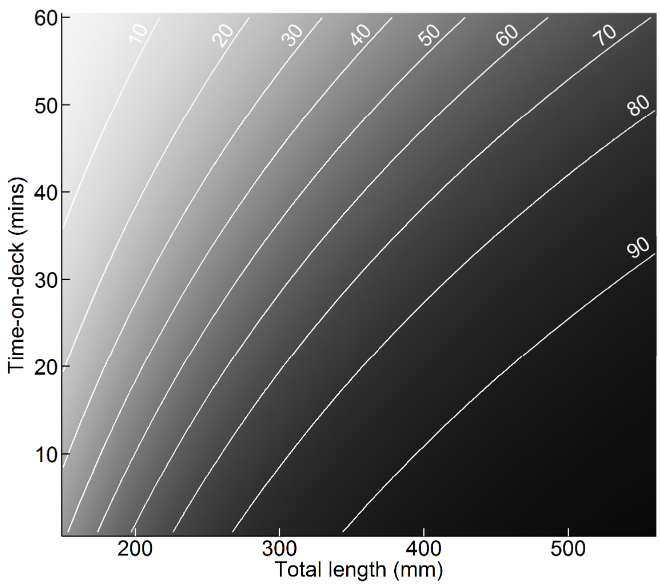


Figure 3: Post-release survival of Eastern Shovelnose Rays (*Aptychotrema rostrata*) as a function of disc width (mm) and time-on-deck (minutes) at the mean trawl duration of 146 minutes caught during three sampling trips conducted in southern Queensland, Australia.

PTS did not differ significantly (P = 0.470) between sampling trips or between the control group (P = 0.932) and those caught in commercial trawls on the *C-Rainger*. The natural logarithm of total length was the best predictor of survival (P = 0.011), with survival increasing with total length (β = 3.227, S.E. = 1.272). Both the natural logarithm of trawl duration (β = –1.206, S.E. = 0.587) and time-on-deck (β = –0.050, S.E. = 0.022) were found to have a significant (P = 0.040 and P = 0.026, respectively) negative effect on survival (Figure 3). Mean overall PTS for *A. rostrata* was 86.8% (S.E. = 3.2%). Condition Index was found to have a significant (P = 0.01) positive effect on survival, suggesting that heathier animals on capture were more likely to survive (Table 10, page 49).

Objective 2 (discard estimation)

Modelling discard rates

Although the discard measures were based on several thousand observations, they were obtained during discrete temporal and spatial windows associated with various research projects, opportunistic sampling and short-term observer trips. Thus, while they can be used to derive mean discard rates for each sector and the effects of bycatch reduction devices (BRDs), the data frame time series is very patchy and unsuitable for quantifying long-term temporal trends in discard rates, other than for pre- and post BRD effects (i.e. before and after 2001 when the devices were mandated). Hence, the discard rates derived from the modelling for each sector are fixed through time, except for the BRD effects (Figure 4).

For all three methods, the reference level that sectors were compared against was the shallow water eastern king prawn sector. Testing for BRD effects within sectors was affected by the lack of non-BRD observations in the red spot king prawn and Moreton Bay sectors. Modelling discard rates in the stout whiting fishery was undertaken separately because the gear used in this sector is very different, with long sweeps designed to 'herd' the whiting. Furthermore, as the objective of the whiting fishery is to retain finfish, no other BRDs apart from turtle excluder devices (TEDs) are required in this sector.

For the retained catch method, discard rates [discards (kg) per retained catch (kg)] in the scallop sector were significantly lower than the shallow water eastern king prawn reference sector (Table 11, page 50, β = -14.28, $P < 0.05$). Overall, BRDs significantly reduced discard rates (β = -0.28, $P < 0.05$), although their effect was inconsistent across sectors. The scallop fishery had the largest reduction (β = -0.56, $P < 0.05$) in discard rates due to BRDs. The influence of sampling program type (i.e. commercial fishing versus research charters) was marginal ($P = 0.084$), indicating that discard rates obtained during charters were higher than those from commercial fishing.

For the effort method, discard rates (kg boat-day^{-1}) in all five sectors were significantly lower than the shallow water eastern king prawn fishery (Table 11, page 50). The influence of BRDs was generally not significant, except for the scallop fishery (β = -1.30 $P < 0.05$). Discard rates varied between sampling trips ($P < 0.05$).

For the swept area method, there were no significant differences in discard rates (kg ha^{-1}) between sectors (Table 11, page 50). Sampling program also had no significant effect (β = 11.32, $P = 0.136$). Overall, BRDs significantly reduced discard rates (β = -0.29, $P < 0.05$), although the effect was inconsistent across sectors. Discard rates were significantly increased in the deep water eastern king prawn (β = 0.41, $P < 0.05$) and tiger and endeavour prawn (β = 0.13, $P < 0.05$) sectors due to BRDs, but significantly reduced in the scallop fishery (β = -0.58, $P < 0.05$).

For the stout whiting sector, discard rates were significantly higher for Danish Seine compared to trawling for all three methods, although only marginally significant for the effort method (β = 1.05, $P = 0.075$) (Table 12). For the retained catch method, discard rates (kg kg^{-1} retained catch) were affected by lunar phase. The relatively large lunar advanced parameter (β = -1.05, $P < 0.05$) indicates that the discard rate declined to a minimum during the waxing lunar phase.

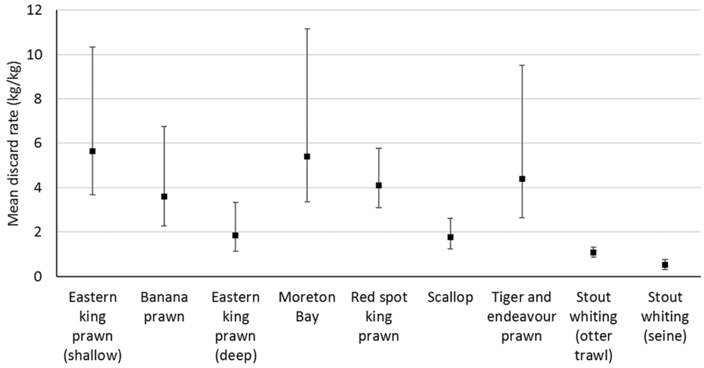
Adjusted mean discard rates for the retained catch method varied between 5.63 ($CI_{\alpha=0.05}$: 3.69 - 10.33) kg kg^{-1} in the shallow water eastern king prawn sector and 0.52 ($CI_{\alpha=0.05}$: 0.51 - 0.53) kg kg^{-1} in the Danish Seine stout whiting fishery, respectively (Figure 4a).

In contrast, the stout whiting (Figure 4b) sector had significantly higher discard rates at 2691 (S.E. 1143) kg boat-day^{-1} and 2312 (S.E. 763) kg boat-day^{-1} for Danish seine and fish trawl, respectively. The tiger and endeavour prawn sector had the highest discard rate among the remaining sectors at 1001 (S.E. 148) kg boat-day^{-1}, while Moreton Bay had the lowest at 208 (S.E. 60) kg boat-day^{-1}.

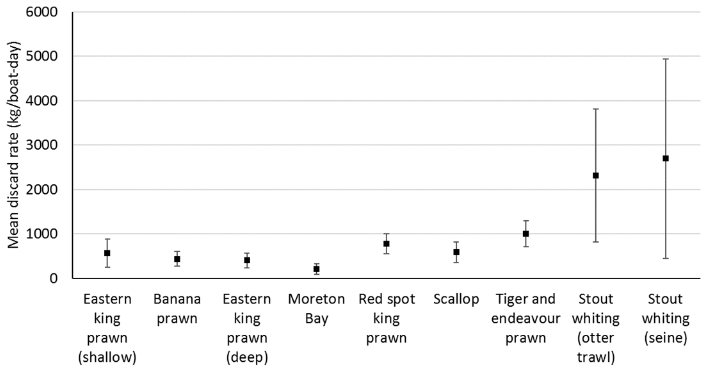
For the swept area method (Figure 4c) discard rates were relatively stable across sectors, with most sectors falling within the range of 5-10 kg ha^{-1}. The banana prawn sector had the highest discard rate at 9.93 (S.E. 1.80) kg ha^{-1} while the deep water eastern king prawn fishery had the lowest at 1.44 (S.E. 0.35) kg ha^{-1}.

Retained catch

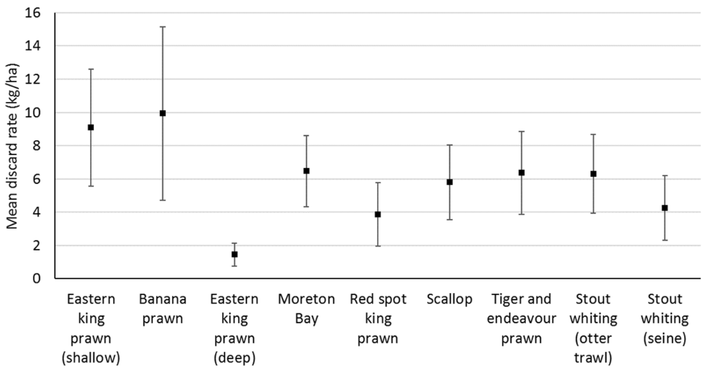
Monthly summations of the retained catch (t) in each sector were derived from the CFISH logbook database and are provided in Figure 11, page 51. In most sectors, the monthly retained catch, which is composed mainly of the principal target species with minor contributions from by-product species, displays marked seasonal cycles within each year. It is noteworthy that catches in the tiger and endeavour prawn fishery, which historically produced the highest catches in the QECOTF, have declined significantly over the last decade. In contrast, monthly catches in the deep water eastern king prawn fishery display a steady increase since 1988.



a) Bycatch per retained catch



b) Bycatch per boat day



c) Bycatch per swept area

Figure 4: Adjusted mean discard rates for sectors that comprise the Queensland East Coast Otter Trawl Fishery, including the stout whiting trawl and Danish seine fishery. Discard rates were derived using three methods based on a) retained catch, b) effort and c) swept area. Vertical bars are 95% confidence intervals.

Fishing effort

Monthly trends in fishing effort (boat-days) based on logbook data from January 1988 to December 2014 show strong seasonal patterns in each sector (Figure 12, page 52). The tiger and endeavour prawn fishery consistently generated the largest monthly fishing effort levels from 1988 to 2000, often exceeding 5000 boat-days per month. Following implementation of a fishing effort unitisation system in 2000, which was a key element of the *Fisheries (East Coast Trawl) Management Plan 1999*, many small vessels with little previous participatory history exited the fishery, collectively resulting in a marked decline in effort. This decline is also apparent in the banana prawn, scallop and Moreton Bay sectors. In contrast, effort levels in the deep water eastern king prawn fishery have remained relatively stable over the entire time series.

The global financial crisis in 2007 and 2008 was associated with reduced profitability in the QECOTF, possibly due to reductions in the demand and price of seafood, combined with elevated fuel costs. This period was also associated with a decline in fishing effort in several sectors, including the shallow water eastern king prawn and tiger and endeavour prawn sectors.

Swept area

Mean daily swept area rates (ha boat-day^{-1}) for each month and sector are provided in Figure 13, page 53 for the period from January 1988 to December 2014. The deep water eastern king prawn fishery has the highest swept area rate at approximately 200-250 ha boat-day^{-1}. This can be attributed to the longer total net head rope lengths permitted in this sector, combined with the slightly larger size and power of the vessels operating in it. Mean daily swept area rates (ha boat-day^{-1}) have remained relatively stable over the time series for most sectors. An exception is the shallow water eastern king prawn fishery where swept area rates have increased from about 60 ha boat-day^{-1} in 1988 to 120 ha boat-day^{-1} in 2014. This may be attributed to an increasing number of larger, more efficient vessels operating in this fishery, particularly since 2000.

Total discard estimates

In four of the eight sectors (i.e. tiger and endeavour prawn, red spot king prawn, scallop and Moreton Bay) the general trend in discards is a decline from 1988 to 2014 (Figure 14, page 54). In three sectors (i.e. shallow water eastern king prawn, banana prawn and stout whiting), there is no discernible trend, while in the deep water eastern king prawn fishery discards increased.

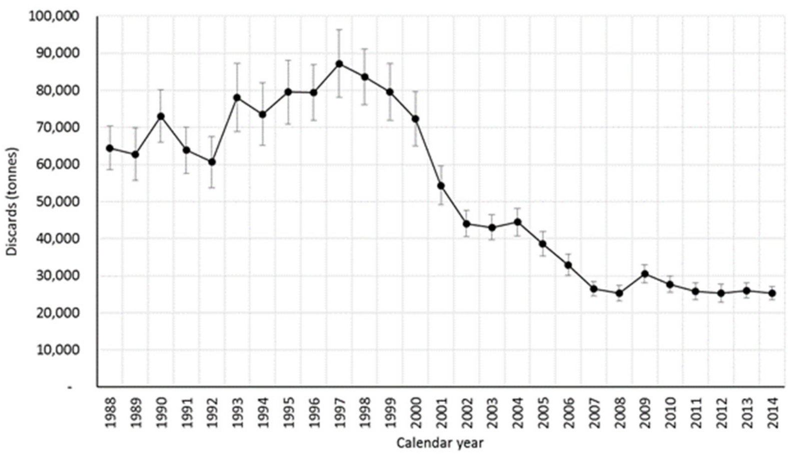


Figure 5: Total annual discard estimates (tonnes) for the Queensland east coast otter trawl fishery for all sectors from January 1988 to December 2014. Each annual estimate is the mean of the three estimates (retained catch method, effort method and swept area method).

The increase in the deep water eastern king prawn fishery is likely attributed to the steady increase in fishing effort and retained catch over the time series. This contrasts with the tiger and endeavour prawn sector where the decline in discards is likely attributable to the significant decline in effort. The marked decline in the scallop sector after 2000 is largely attributed to BRDs, which have been more effective in this sector than any other. Trends in discards for the banana prawn sector are relatively unstable and characterised by large fluctuations. This is likely attributed to the nature of banana prawn catch and effort, which are heavily affected by rainfall and freshwater flows.

In the 1980s and 1990s, the tiger and endeavour prawn sector consistently generated more discards annually than any other sector. Based on the effort and swept area methods, estimates often exceeded 30,000 t. In recent years, however, discards in this sector have remained consistently below 10,000 t (Figure 14, page 54). In recent years, discards from the eastern king prawn fishery have remained at about 10,000 t annually (combined both shallow and deep sectors), which is the highest amount of any sector.

By summing the sector means each year a long-term trend in total annual discards was derived for the whole QECOTF (Figure 5). This long-term trend indicates that total discards for the whole fishery peaked at 87,175 t (S.E. 9089) in 1997 and declined markedly over the following decade. Since 2011, annual total discards have remained relatively stable between 25,271 and 25,983 t.

Objective 3 (risk assessment)

For the purposes of the current study, the spatial extent of the trawl fishery south of the GBRMP encompasses 27,912km^2 (Figure 6a). Of the five sectors that comprise this area, the shallow water EKP sector is the largest at 16,151km^2 and the banana prawn sector is the smallest at 405km^2. Logbook data indicate that the spatial extent of each sector is relatively stable from one year to the next.

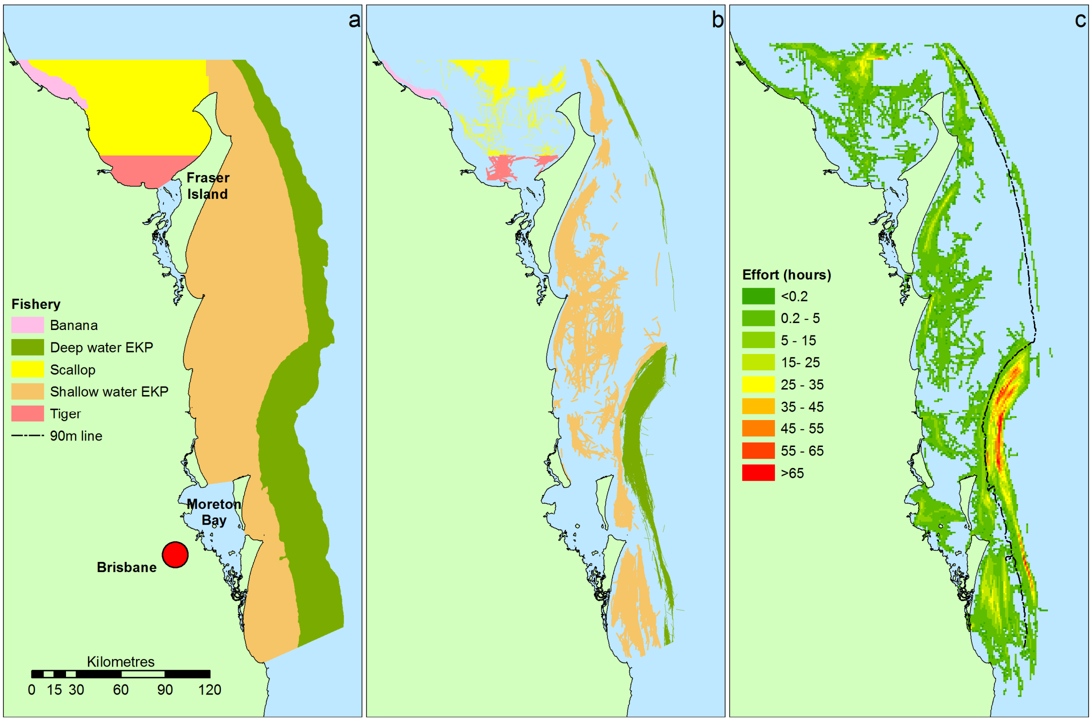


Figure 6: a) Spatial extent of the five sectors south of the Great Barrier Reef Marine Park, b) distribution of prawn and scallop trawl effort within each sector in 2015 generated using TrackMapper software, and c) trawl effort, in hours, at a resolution of 0.1 nm^2 (~0.343km^2) in 2015 from TrackMapper.

The deep water EKP sector has the largest trawled area at 11,158km^2, resulting from high levels of fishing effort (Figure 6b) and the size of the gear used (Table 2). Trawl effort was high in the area to the east of the Sunshine Coast in the deep water EKP sector and relatively high in the shallow water EKP sector east of Fraser Island, Double Island Point, Moreton Island and Southport (Figure 6c).

Table 5: Spatial extent of each sector and the trawled area within each sector (both in km^2) for the purposes of the current study.

Sector	Area	Trawled area
Banana	405	149
Shallow water EKP	16,151	5,230[1]
Tiger	889	23
Scallop	5,112	945
Deep water EKP	5,356	11,158
Total	27,912	17,505

[1]Includes 842km^2 and 837km^2 for Danish seine and fish trawl, respectively

Species distribution for the elasmobranchs ranged in size from 27,912km^2 for the Scalloped Hammerhead (*Sphyrna lewini*), which occurs throughout all sectors in the fishery, to 1,934km^2 for the Philippine Spurdog (*Squalus montalbani*), which occurs only in deeper waters (see Species distributions, page 61).

Of the 47 species assessed, only 18 had published life history parameters (Table 8, page 47). As such, the life history tool on Fishbase was the primary source for the metrics required to determine M in order to derive F_{msm}. A single estimate of F_{msm} was possible for 15 of the species assessed (Table 15, page 60) due to a lack of relevant data. F_{msm} ranged between 0.052 for *Squalus montalbani* and 0.320 for *Rhizoprionodon taylori*.

Gear efficiency was only estimated at the species level for the two most common elasmobranchs, *A. rostrata* and *T. testacea* (Table 13, page 58). For the most part, catch sampling was inadequate to quantify gear efficiency at the species level and, as such, species were grouped at increasingly higher classification levels until the models were able to produce satisfactory posterior distributions of catch, compared to the observed values. This was particularly the case for sharks, which were relatively uncommon in the sampling undertaken by Courtney *et al.* (2007b), Dodt (2005) and by Fisheries Queensland's Fishery Observer Program (Table 13, page 58). Further, to reduce Type II errors or 'false' zeros, data recorded as part of the opportunistic sampling conducted by Courtney *et al.* (2007b) were excluded from the analysis. Gear efficiency was generally lowest for the stout whiting fish trawl gear and highest for prawn trawls (see Supplementary results on page 48 and Table 13 on page 58).

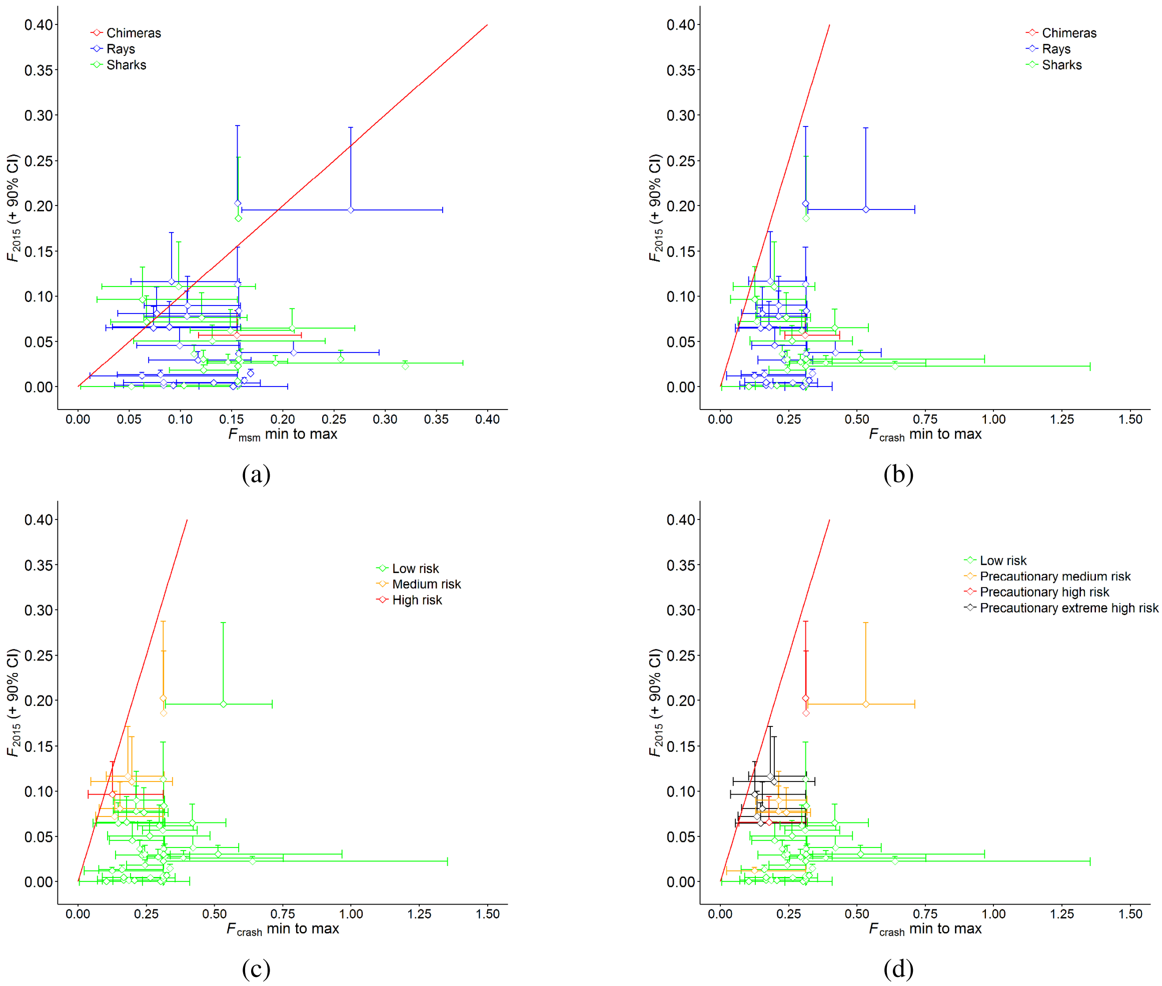


Figure 7: Comparison of estimated fishing mortality for 47 elasmobranchs with a) maximum sustainable fishing mortality, F_{msm}, as a function of classification, b) unsustainable fishing mortality, F_{crash}, as a function of classification, c) unsustainable fishing mortality, F_{crash}, as a function of risk category, and d) unsustainable fishing mortality, F_{crash}, as a function of precautionary risk category. The red line is where $F = F_{msm}$ or $F = F_{crash}$.

Published estimates of escapement (E) and post-trawl survival (S) were available for very few species (Table 13, page 58). We used escapement rates published by Zhou and Griffiths (2008) for seven species that are found in northern Australia and survival rates derived in the current study for *T. testacea* and *A. rostrata*. Survival estimates published by Stobutzki *et al.* (2002) and estimates derived from unpublished data collected during a previous FRDC research project (Courtney *et al.* 2007b) were also used.

Six species (*Dentiraja endeavouri*, *Parascyllium collare*, *Heterodontus galeatus*, *Bathytoshia lata*, *Urolophus bucculentus* and *Urolophus flavomosaicus*) were categorised as being at medium risk (i.e. $F_{lim} > F > F_{msm}$) and one (*Squalus megalops*) was categorised as being at high risk (i.e. $F_{crash} > F > F_{lim}$). Generally, these species had restricted distributions within the fishery's spatial extent that overlapped with significant levels of trawling (see Appendix 7 – Species distributions, page 61) resulting in fishing mortality estimates that exceeded sustainable levels (i.e. $F > F_{msm}$, see Figure 7). The instantaneous fishing mortality rate did not exceed unsustainable levels (i.e. $F < F_{crash}$) for any species (Figure 7b).

When assessing the uncertainty around the point estimates of both F and F_{msm}, all seven species found to be at medium or high risk were categorised at higher precautionary risk levels (Figure 7d). Additionally, five species (*Dentiraja endeavouri*, *Urolophus sufflavus*, *Hemitrygon fluvorium*, *Urolophus viridis* and *Sphyrna lewini*) went from low risk to precautionary medium risk, *Urolophus kapalensis* went from low risk to precautionary high risk and *Maculabatis astra* went from low risk to precautionary extreme high risk (Figure 7d).

Discussion

Objective 1 (elasmobranch post-trawl survival)

These results represent the first short-term post-trawl survival (PTS) estimates for elasmobranchs discarded from prawn trawls in the primary literature. Although two previous studies (Fennessy 1994; Stobutzki *et al.* 2002) discuss the at-vessel mortality of various sharks and rays, the current study is the first to maintain animals in holding tanks for an extended period after capture. The mean PTS of 86.8% for *A. rostrata* is at the upper bounds for batoids assessed using comparable methods (Ellis *et al.* 2017), while the survival of *T. testacea* was relatively low, especially for males (17.3%).

Species-specific differences in PTS from trawl gear are evident in previous studies of elasmobranchs. For example, the PTS of thornback skate (*Raja clavata*) was 80.8% in the Turkish bottom trawl fishery (Saygu and Deval 2014), compared to 20.6% for the brown skate (*Raja miraletus*). Similarly, Enever *et al.* (2009) reported PTS of 59% for *R. clavata*, compared to 33% for the cuckoo skate (*Leucoraja naevus*). In both studies, the higher PTS for *R. clavata* was attributed to its accentuated spines that provide improved physical protection compared to other species. Differences in morphology were obvious in the two species assessed in the current study: *A. rostrata* are covered in fine denticles (Last *et al.* 2016), while *T. testacea* are smooth and soft to touch. This likely afforded *A. rostrata* more protection against trawl capture and release than *T. testacea*. Further, the morphology of *A. rostrata* provided protection against the physical abrasions associated with confinement-dependent factors affecting survival during the holding period such as abrasion by the plastic tanks and piercing by the caudal sting of captive *T. testacea*. In contrast, a high proportion (>50%) of *T. testacea* had abrasions and injuries, particularly on their ventral surface which appeared red and irritated at the end of each containment period. This issue may have been alleviated with the addition of a sedimentary substrate, allowing animals to bury and avoid abrasive contact with the plastic tanks. Although the *Tom Marshall* was anchored in calm waters, the Gold Coast Broadwater is a busy waterway and the wash from numerous large recreational vessels caused the vessel to roll violently at times, affecting the environment in which the animals were housed. Further, ~25% of *T. testacea* showed signs of infection at the tag site, while no *A. rostrata* appeared affected. The addition of an anti-biotic and anti-fungal ointments to the tag wounds, such as those used by Courtney *et al.* (2001), may have reduced any infection but were deemed unnecessary given that only short-term survival was assessed. As such, captivity in the holding tanks likely contributed to the low PTS of *T. testacea* and the estimates derived in the current study should be considered as minimum for this reason.

The effect of holding animals in tanks is acknowledged as a source of bias when assessing PTS (Broadhurst *et al.* 2006; Mandelman and Farrington 2007a; Ellis *et al.* 2017). Ellis *et al.* (2017) suggest that captive stress, stocking densities and environmental conditions may affect post-release survival (PRS) estimates. Despite this, the use of tanks to hold captive animals is the most common method used to determine PTS for elasmobranchs in field-based studies (e.g. Kaiser and Spencer 1995; Revill *et al.* 2005; Rodríguez-Cabello *et al.* 2005; Enever *et al.* 2009; Benoît *et al.* 2010; Enever *et al.* 2010; Cicia *et al.* 2012; Depestele *et al.* 2014;

Saygu and Deval 2014). Apart from on-board tanks, researchers have used various methods to quantify the PTS of elasmobranchs such as at-vessel mortality (Stobutzki *et al.* 2002), qualitative health assessments (Benoît *et al.* 2010; Benoît *et al.* 2013), submerged holding pens located adjacent to fishing grounds (Mandelman and Farrington 2007a; Rulifson 2007; Mandelman *et al.* 2013), land-based tanks (Mandelman and Farrington 2007b; Cicia *et al.* 2012) and trawl simulation studies (Frick *et al.* 2010; Heard *et al.* 2014).

Each of these methods, however, has been shown to bias PTS estimates. For example, Frick *et al.* (2010) and Heard *et al.* (2014) tested the effects of crowding, air exposure and trawl duration on the survival of the three species of elasmobranch (*Heterodontus portusjacksoni*, *Mustelus antarcticus* and *Urolophus paucimaculatus*) in a laboratory and concluded that the PRS estimates derived in each study cannot be extrapolated to animals caught in the wild due to the absence of additional stressors such as temperature change. Additionally, at-vessel mortality has been used to assess PTS in prawn trawl fisheries (Fennessy 1994; Stobutzki *et al.* 2002): however, given the delayed effects of capture by trawl gear on survival (e.g. van Beek *et al.* 1990; Wassenberg and Hill 1993; Kaiser and Spencer 1995), this method would have likely yielded underestimates of PTS for *A. rostrata* and *T. testacea*. Further, Dudgeon *et al.* (2013) used passive acoustic telemetry to assess the site fidelity of the zebra shark (*Stegostoma fasciatum*) in south east Queensland. However, this method was deemed unsuitable to assess PTS in the current study given the time required to carefully perform invasive surgery at night, on a commercial vessel. Similarly, pop-up satellite archival tags (PSATs), used to estimate post-release survival of large elasmobranchs (e.g. Campana *et al.* 2016) were unsuitable given the small size of *T. testacea* and *A. rostrata*.

Comparatively few *T. testacea* and *A. rostrata* were categorised as dead or nearly dead (Category '1'Table 10 page 49) on capture with most mortalities occurring within 24 hours of capture regardless of trawl type (control vs. commercial). Condition Index was found to be a poor predictor of survival for *T. testacea* (Table 10). Only 38% of *T. testacea* with a Condition Index of '3'survived reinforcing the inadequacy of AVM as a reliable proxy for PTS for this species in particular, although this result may be compromised by the confinement-dependent effects discussed above. In contrast, 90.5% of *A. rostrata* with a Condition Index of '3' survived as did half of the animals given a Condition Index of '1'. Given that only two staff were responsible for assessing condition, it may have been beneficial to increase the number of condition categories (Ellis *et al.* 2017). For example, Benoît *et al.* (2010) used four categories to assess survival from a fish trawl in Canada. In the current study, the survival of animals presenting with a Condition Index of '1' had the most variable PTS (Table 10, page 49) and, as such, this portion of the study may have benefited from an extra category describing poor health such as that described by Benoît *et al.* (2010).

The housing of animals in on-board tanks precludes interaction with predators and scavengers (e.g. Enever *et al.* 2009; Mandelman *et al.* 2013). Therefore, PTS derived using this method may be underestimated. In the current study, bull sharks (*Carcharhinus leucas*) were observed feeding on the discards from the *C-Rainger* whilst the catch was being sorted. Further, blue swimmer crabs (*Portunus armatus*) and three-spot crabs (*P. sanguinolentus*) were regularly caught as bycatch throughout the three sampling trips, both of which are known scavengers caught by pot fishers in south east Queensland. Further research is required to determine the effect of predation and scavenging on the PTS of *T. testacea* and *A. rostrata*.

For both species, the size of the individual was the best predictor of PTS, with larger animals more likely to survive (Figure 2 and Figure 3). This is consistent with previous studies for elasmobranchs (Stobutzki *et al.* 2002; Enever *et al.* 2010; Depestele *et al.* 2014; Saygu and Deval 2014). For example, Saygu and Deval (2014) found that larger thornback skates (*Raja clavata*) and brown skates (*R. miraletus*) were more likely to survive at least 48 hours after capture in a Turkish trawl fishery. Similarly, Enever *et al.* (2010) reported that health score on capture increased with the size of skates (*Leucoraja naevus*, *Raja microocellata*, *R. brachyura*, *R. clavata and R. montagui*) caught by fish trawls in the United Kingdom, resulting in higher PTS. The size-related difference in survival has been attributed to reduced resilience of smaller animals to fatigue and injury (Davis 2002; Benoît *et al.* 2013). Benoît *et al.* (2013) suggested that smaller animals are less likely to survive capture because of a susceptibility to hypoxia due to a higher mass-specific metabolic rate and a higher energy cost for breathing.

Air exposure is an important predictor of PTS (Davis 2002; Broadhurst *et al.* 2006; International Council for the Exploration of the Sea 2014) and is a function of the time required to process the catch (Davis 2002). In the current study, time-on-deck affected the PTS of both *T. testacea* (Figure 2) and *A. rostrata* (Figure 3). Time-on-deck reflected total catch weight and trawl duration: increasing trawl duration resulted in higher catch weights and longer sorting times. The crew of the *C-Rainger* sorted the catch by making a space

(~0.75m^2) on the sorting tray before filling that space with a small amount of catch. Prawns were removed to buckets before any elasmobranchs were selected and passed to the observer for tagging. This process was repeated until all catch had been processed. The tagging procedure took approximately 5 – 10 s and, as such, the time-on-deck metric used in the current study is representative of commercial operations in the eastern king prawn fishery. A reduction in PTS resulting from increased air exposure is consistent with previous studies. Cicia *et al.* (2012) and Frick *et al.* (2010) reported lower survival for little skate (*Leucoraja erinacea*) and gummy sharks (*Mustelus antarcticus*), respectively, after increased levels of air exposure during laboratory-based experiments. Field studies, (Benoît *et al.* 2010; Benoît *et al.* 2012) found reduced survival for skates (Rajidae) with increasing exposure to air after capture by fish trawls in Canada. Cicia *et al.* (2012) provide a detailed description of the physiological response to air exposure and, along with Mandelman *et al.* (2013), show that elevated temperature gradients between air and water exacerbate the effects of air exposure. However, the temperature gradient was relatively constant across the three sampling trips in the current study, prohibiting analysis of this metric.

Female *T. testacea* were more likely to survive than males. Studies that have found gender-specific PTS in elasmobranchs invariably report that the survival of females is higher (Stobutzki *et al.* 2002; Laptikhovsky 2004; Enever *et al.* 2009; Mandelman *et al.* 2013). Enever *et al.* (2009) and Mandelman *et al.* (2013) suggest higher survival in females is a result of the thicker skin which provides protection against biting males during copulation. Further, Mandelman *et al.* (2013) hypothesise that the presence of claspers may lead to injuries for males. In the current study, female *T. testacea* were larger (Figure 2), confounding the effect of gender, given the generalised linear modelling (GLM) indicated that size was the best predictor of PTS in this species. Similarly, Stobutzki *et al.* (2002) found that the immediate PTS of female batoids (*Neotrygon leylandi*, *Maculabatis toshi* and *Gymnura australis*) was higher in a northern Australian prawn trawl fishery, noting that the males of most elasmobranchs are smaller. Interestingly, the PTS of *A. rostrata* was not gender-specific, nor were there significant size differences between the sexes of this species.
Tow duration had a negative effect on the PTS of *A. rostrata*. Where measured, increased tow duration has resulted in lower PTS for elasmobranchs (Fennessy 1994; Mandelman and Farrington 2007a; Enever *et al.* 2010; Mandelman *et al.* 2013). For example, Fennessy (1994) reported that shorter tows resulted in increased PTS of backwater butterfly rays (*Gymnura natalensis*) in a South African prawn trawl fishery. However, as discussed previously, the results from the current study show that there is correlation between tow duration, time-on-deck and catch weight. An inability to quantify the exact time an animal enters the trawl (Mandelman *et al.* 2013) somewhat compromises tow duration as a valid predictor of PTS. In the current study, tow duration was preferred to catch weight as a predictor of PTS due to difficulties in measuring catch weight accurately. Further, tow duration is a metric familiar to prawn trawl operators facilitating better communication of results to stakeholders.

Objective 2 (discard estimation)

Influence of BRDs

This study used generalised linear modelling to examine the potential influence of several factors on discard rates in the Queensland East Coast Otter Trawl Fishery (QECOTF). Stratifying the fishery into sectors significantly increased the amount of variation that could be explained, as two of the three methods (i.e. retained catch and effort methods, Table 11, page 50) detected significant differences in discard rates between sectors when compared to the base level (i.e. shallow water eastern king prawn sector). For the effort method, all sectors were associated with significant negative parameter values, indicating that discard rates (kg boat-day^{-1}) are relatively high in the shallow water eastern king prawn fishery.

Bycatch reduction devices (BRDs) generally resulted in significant reductions in discard rates, however, their effects varied across methods and sectors. It is noteworthy that no significant BRD effect was found for the effort method (Table 11). This is likely due to having to exclude the research charter data for this method because the amount of discards generated during a charter-day is unrepresentative of a commercial fishing boat-day. Hence, the only observations from nets with BRDs included for this analysis were devices used by commercial fishers during their normal fishing activities. Several different combinations of TEDs and other BRDs have been assessed in the QECOTF based on research charters, with significant reductions in discard rates demonstrated (Courtney *et al.* 2006; 2008; 2014). The lack of a significant BRD effect in the effort GLM (, page 50) may be due to commercial fishers using relatively ineffective devices, and/or their ineffective installation. The lack of a significant BRD effect for the effort method is also partially attributed to reduced experimental control during normal commercial fishing activities, resulting in reduced ability to detect such effects. This supports the need to test BRDs under tightly controlled conditions which are often not possible during commercial fishing activities.

BRDs used in the scallop fishery were found to significantly reduce discard rates, and resulted in the largest BRD effects (Table 11, $\beta = -0.56$, $P < 0.05$ for the retained catch method and $\beta = -0.58$, $P < 0.05$ for the swept area method). This strong effect is reflected in the decline in annual discards in the scallop fishery, especially for the swept area method, after 2000 when BRDs became mandatory in the QECOTF (Figure 13, page 53). Much of this influence can be attributed to TEDs excluding sponges (Porifera) which comprise over 60% of the weight of discards in this sector (Courtney *et al.* 2008). Sponges are not a dominant component of the discard assemblages for the other sectors.

Comparison of methods

For most sectors, the three methods produced considerably different discard estimates (Figure 14, page 54), with the largest differences in the order of 2-3 fold in some years, particularly prior to 2001. For the banana prawn sector it is noteworthy that the trends were relatively similar. This may be related to this species (*Fenneropenaeus merguiensis*) being the only schooling species in the QECOTF and caught predominantly during daylight in very shallow depths. Fishing effort in the banana prawn fishery may be a particularly good predictor of retained catch as well as discards.

For most sectors the discard trajectories merged after 2001. This is particularly noticeable in the scallop fishery and may be related to the strong BRD influence in this sector. In four of the eight sectors the retained catch method consistently resulted in the lowest discard estimates. In terms of ranking the methods, the effort method was probably the least reliable because it was based on the fewest observations (n=381 boat-days), due to the exclusion of the research charter data and the pooling of the commercial trawl data to boat-days. The pooling of observations to boat-days combined with only including commercial fishing observations also significantly reduced the model's ability to evaluate BRD effects. The swept area method probably produced the most reliable discard estimates, because it took into account more factors, such as net size and trawl duration. However, we acknowledge that most fisheries will not have access to such details. The retained catch method was the simplest of the three methods, but often produced the lowest discard estimates, possibly suggesting that it is the least conservative of the methods considered here.

Objective 3 (risk assessment)

The results of the risk assessment conducted as part of the current project represent the first attempt at quantifying the risk posed to any species by the trawl fishery south of the Great Barrier Reef Marine Park (GBRMP) in Queensland. Despite the deficiencies in some of the metrics used to determine risk in the current study (see discussion below), the SAFE method has been shown to be superior (Zhou *et al.* 2016) to qualitative methods in assessing risk such as the Productivity and Susceptibility Analysis (PSA) used by Stobutzki *et al.* (2001a). Generally, qualitative assessments are more precautionary, resulting in more species being classified as being at medium or high risk. This is the case for a qualitative risk assessment conducted by Pears *et al.* (2012). These authors assessed the ecological risk posed to 33 elasmobranchs in the GBRMP and found that 11 species (~33%) were at high risk (Figure 7). Nine of these species occur south of the GBRMP and were assessed as low risk (*A. rostrata*, *T. testacea*, *N. trigonoides*, *N. picta*, *D. endeavouri*, *M. astra*, *H. monopterygius* and *G*, *australis*) or medium risk (*U. flavomosaicus*) in the current study.

The approach of using the species' distribution to quantify A_x is known to underestimate fishing mortality, F_{2015}. Zhou *et al.* (2015) uses two other approaches to quantify species distribution to assess risk in the Joseph Bonaparte Gulf red-legged banana prawn (*Penaeus indicus*) fishery. These two approaches rely on dividing the fishery into cells, 6nm x 6nm in size, and using survey data to define the distribution of a species based on the grids where that species was present in research trawls. Fishing mortality for this species is then quantified based on the level of trawl effort within the cells where it occurs. It is assumed the species does not occur outside of these cells and, as such, these approaches tend to overestimate fishing mortality. Although this is the most conservative approach for estimating fishing mortality, the low frequency of occurrence of elasmobranchs sampled (Table 13, page 58) for the current study would have resulted in many species being categorised, erroneously, as being at high risk. This is a function of the sampling undertaken to determine species location in the current study: all trawls were conducted either on-board vessels during commercial fishing or during surveys conducted on commercial fishing grounds. This resulted in an unacceptable overestimation of fishing mortality. In contrast, the sampling undertaken in the

NPF includes trawls conducted outside of the fishery area, allowing for a more accurate description of species distribution compared to those in the current study.

The species distributions in the current study are quantified using several sources: Last and Stevens (2009), the Atlas of Living Australia (ALA)[1], catch data from Courtney *et al.* (2007b) and Kyne *et al.* (2005). Despite multiple sources, the area of each species distribution represents a significant source of uncertainty in the assessment of risk. For example, the ALA reports that *S. megalops* occurs in water depths between 30m and 750m: however, Last and Stevens (2009) say that the species occurs in shallow water in southern waters. We have, therefore, assumed the restricted, more conservative distribution for the purposes of the ERA in the current project. Ideally, regular fishery independent surveys should be carried out to determine the distribution of each species in the area of interest.

A lack of high-resolution trawl effort information, such as that used to assess risk in the current study, is a significant limitation in qualitative assessments like that undertaken by Pears *et al.* (2012) and others. For example, these authors reported that the QECOTF posed a high risk to the pale tropical skate (*Dipturus apricus*) despite trawling occurring in the less than 1% of the species' distribution (195-605m). Furthermore, Last and Stevens (2009) report that *D. apricus* occurs mainly in waters between 300 and 500m, in which case the QECOTF poses no risk to this species as trawling does not occur in these depths.

All but one (*H. galeatus*) of the species assessed as being at medium or high risk occur predominantly in the deep water EKP sector (Appendix 7 – Species distributions, page 61). Specifically, these species' distributions are restricted to areas that have high levels of trawl effort (see Figure 6c), resulting in high fishing mortality rates (i.e. F_{2015}). Further, these species are relatively long-lived and slow growing (Table 8, page 47), characteristics that make elasmobranchs susceptible to overfishing (Dulvy *et al.* 2008). For example, the only species at high risk in the current study, *S. megalops*, has a published growth rate of k = 0.034yr^{-1} and an L_∞ = 82.9cm (Braccini *et al.* 2007). This species also lives in excess of 25 years and has a high age-at-maturity (~19 years: Braccini *et al.* 2006).

Of the 47 species assessed in the current study, life history characteristics are available for 18 (~38%). Of these 18 species, life history characteristics were derived from samples collected within the trawl fishery for only seven species (*H. fluvorium*, *L. macrorhinus*, *M. astra*, *M. walkeri*, *N. picta*, *N. trigonoides* and *S. montalbani*). This represents an avenue of research that requires urgent attention as growth is known to differ both spatially (e.g. Moulton *et al.* 1992) and temporally (e.g. Carlson and Baremore 2003). For example, the age and growth of *S. megalops* were derived from samples collected on demersal trawl and shark gill-net vessels operating in the Southern and Eastern Scalefish and Shark Fishery in waters off south-eastern Australia. As such, this species may grow faster in the warmer waters of southern Queensland resulting in an increased resilience to fishing mortality (i.e. F_{msm}). Improvements to, and validation of, life history characteristics have been identified as limitations in previous ERA studies (Zhou *et al.* 2016).

Along with a lack of life history data, the lack of data quantifying escapement from turtle exclusion devices (TEDs) is an area of research requiring attention. For example, the seven species found to be at medium or high risk had no published information on escapement from TEDs. These devices were introduced in Queensland between 1999 and 2001 to reduce the impact on sea turtles. The uptake of TEDs in tropical prawn trawl fisheries has likely led to beneficial flow-on effects (Jordan *et al.* 2013): the mechanical separation of catch (Broadhurst 2000), essentially a function of the bar spacing (or mesh size in the case of soft/flexible TEDs) of the device and the size of the animal encountering the device, prohibits the entry of large animals into the codend. This has led to significant reductions in the number of large elasmobranchs captured by tropical prawn trawl fisheries (e.g. Robins-Troeger *et al.* 1995; Brewer *et al.* 2006; Willems *et al.* 2016). However, there are very few studies detailing the effects of TEDs and BRDs in the primary literature where elasmobranchs were the focus, with data on these species collected only on an opportunistic basis. This had led to reportage regarding the effects of TEDs and BRDs on the catch rates of elasmobranchs based on relatively low sample sizes with resultant uncertainty and unreliability of results (e.g. Courtney *et al.* 2006; Queirolo *et al.* 2011; Jordan *et al.* 2013). Further, species differentiation is often absent, with individuals grouped to genus, family or order (Oliver *et al.* 2015).

These issues were especially evident during the 1990s. As TEDs and BRDs became mandatory in prawn trawl fisheries, their effects on target catch and discards were the focus of significant research efforts,

[1] http://www.ala.org.au/

particularly in Australia (see review by Broadhurst 2000). As a result, studies during this period were motivated by the need to inform fishers and managers of devices that satisfied legislative requirements regarding turtle exclusion whilst maintaining the catch rates of target species. Numerous studies from the 1990s reported the effects of TEDs and BRDs on prawn and discard catch rates (e.g. Kendall 1990; Isaksen *et al.* 1992; Rulifson *et al.* 1992; Broadhurst *et al.* 1996; Broadhurst *et al.* 1997; Kennelly *et al.* 1998), while others also confirmed the exclusion of turtles (Robins-Troeger 1994; Robins-Troeger *et al.* 1995; Brewer *et al.* 1998; McGilvray *et al.* 1999). Most studies during the 1990s were conducted on known trawl grounds in an effort to replicate commercial conditions (Robins-Troeger 1994; Broadhurst *et al.* 1997; Robins and McGilvray 1999). This resulted in sufficient quantities of both target species and bycatch to enable robust analyses from a relatively small number of trawls, especially where paired comparisons were employed. Given that interactions with elasmobranchs in prawn trawls are relatively rare (e.g. Fennessy 1994; Kyne *et al.* 2002; Fennessy and Isaksen 2007; Wakefield *et al.* 2016), analyses regarding the effect of TEDs and BRDs on these species were largely absent. Generally, most elasmobranchs were grouped in with the discarded portion of the bycatch, with mention made of a relatively small number of large individuals (e.g. Kendall 1990; Robins-Troeger *et al.* 1995; Brewer *et al.* 1998; McGilvray *et al.* 1999; Robins and McGilvray 1999).

The escapement rates (see Table 13, page 50) used in the current study were all taken from a study by Zhou and Griffiths (2008). These authors appropriated escapement rates derived from several studies conducted in the NPF (Brewer *et al.* 1998; Brewer *et al.* 2004; Brewer *et al.* 2006). The escapement rates are applicable in the current study given the required bar spacing for both the NPF and QECOTF is 12cm maximum. Numerous species assessed in the current study that have no published escapement rates would certainly be excluded by TEDs due to their size. For example, the scalloped hammerhead (*Sphyrna lewini*) has an escapement rate of $E = 0$ despite the fact that it reaches 3.5m, while both the giant guitarfish and the bottlenose wedgefish grow to 2.7m and 3.0m, respectively, resulting in escapement rates of $E = 1$ (Zhou and Griffiths 2008). Further, Raborn *et al.* (2012) estimated that the introduction of TEDs reduced the catch rate of bonnethead sharks (*Sphyrna tiburo*), which have a maximum size of 1.2m, by 31%. Brewer *et al.* (2006) reported reductions of 86% and 94% for larger sharks and rays (>1m total length or disc width), respectively. As such, any species growing larger than 1 m would be subjected to some level of exclusion. Further work is required to ensure realistic escapement rates are used in quantifying fishing mortality in future risk assessments.

Apart from the PTS rates for *A. rostrata* and *T. testacea*, quantified as part of the current study (see Results, page 12), the survival rates used in the ecological risk assessment were immediate or at-vessel survival (Table 13, page 58). Stobutzki *et al.* (2002) published survival rates for six species common to both the NPF and the fishery area of the current study, while preliminary PTS of four species was estimated using data recorded as part of a FRDC-funded research project (Courtney *et al.* 2007b). These estimates should be regarded as a maximum PTS given they represent the survival of animals on capture and ignore any delayed effects known to affect survival (e.g. van Beek *et al.* 1990; Wassenberg and Hill 1993; Kaiser and Spencer 1995). As discussed earlier, the paucity of PTS studies in prawn trawl fisheries is most likely due to the cost and logistical constraints of field-based experiments needed to quantify post-release survival (Musyl *et al.* 2011; Benoît *et al.* 2012; Benoît *et al.* 2013; Dapp *et al.* 2016). However, the results from the survival studies conducted as part of the current study demonstrate that PTS of elasmobranchs is variable but not necessarily zero. The need to be conservative necessitates the use of $S = 0$ where no survival information is available but future risk assessments should include an average survival derived from a meta-analysis of PTS studies or other such research. The seven species assessed as being at medium or high risk have no published PTS estimates.

Trawl effort in the QECOTF has decreased since 2000. This coincided with the implementation of the *Fisheries (East Coast Trawl) Management Plan 1999*, which included effort unitisation and a 2-for-1 new boat policy. These measures have seen nominal effort decrease in all sectors of the QECOTF (Figure 12): however, effort in the deep water EKP fishery has been relatively constant in the last ten years. The removal of small vessels from the EKP fishery has resulted in an increase in vessel mean fishing power so that, despite a decrease in nominal effort, catch rates have increased since 2000 (O'Neill *et al.* 2014). As a result, those elasmobranchs that are restricted in distribution to deeper waters have been subjected to similar levels of fishing effort and fishing mortality for at least a decade.

The results of the ERA represent localised issues resulting from high levels of fishing effort in the area south of Noosa. All of the species assessed as being at medium or high risk have distributions that extend well beyond the confines of the fishery assessed. As such, it is sensible to assess the risk posed to these species across their entire distribution. For example, the SAFE method could be used for the species impacted by the QECOTF in all of Queensland, including the Great Barrier Reef Marine Park. Further, the EKP fishery extends well into New South Wales and data from this jurisdiction can be included, and the impacts assessed across a much wider area.

Conclusion

Queensland's East Coast Otter Trawl Fishery (QECOTF) is the largest trawl fishery in Australia with 289 vessels accessing the fishery in 2015. The QECOTF is a multi-sector fishery targeting prawns, scallops, Moreton Bay bugs, squid and stout whiting with a GVP of ~$AUS81 million in 2015.

In 2000, the Queensland government implemented a range of management arrangements designed to increase profitability and mitigate the environmental impacts of the fishery. These objectives were achieved through a range of measures including effort unitisation and the introduction of turtle excluder devices (TEDS) and other bycatch reductions devices (BRDs). These measures were necessary given the increasing concern regarding the effects of prawn trawling, particularly within the World Heritage-listed Great Barrier Reef.

The results of the current study demonstrate that the management changes that occurred in the early 2000s have had a significant effect on the discards produced in the QECOTF. Discards production peaked in 1997 at 87,175 tonnes before decreasing to ~45,000 tonnes in 2002. Since this time, discards have decreased further to about 25,200 tonnes per annum for the period 2011-2014. This reduction in discards is likely attributed to a concomitant reduction in nominal fishing effort resulting from numerous factors including: a reduction in the number of vessels, increased fuel prices, decreasing or stable product prices and reduced access to fishing grounds.

The effects of TEDs and BRDs are variable between sectors, with the devices being most effective in the scallop fishery. This is a function of the composition of the discards associated with each sector. For example, large sponges comprise a high proportion of the discards from the scallop sector, all of which are excluded by TEDs, resulting in significant reductions in total discards from this sector. In contrast, the discards from the shallow water EKP sector are mainly composed of small fish and crabs which pass through the bars of the TEDs used but lack the swimming ability to locate and exit the trawl via BRDs, resulting in only modest reductions in discards.

Elasmobranchs (i.e. sharks and rays) are one component largely unaffected by TEDs and BRDs in the Eastern King Prawn (EKP) sector. These species are characterised by late maturity, few offspring, long life spans and slow growth, making them vulnerable to overexploitation. While the introduction of TEDs has gone some way to decreasing the ecological risk posed to large elasmobranchs by prawn trawling, numerous studies have shown that the catch rates of smaller species remain unaffected. In the current project, the only species assessed as being at high risk was the Piked Spurdog (*Squalus megalops*), a small (<64cm TL) spurdog restricted to areas of high effort within the trawl fishery south of the GBRMP. Six further species were found to be at medium risk, all of which have restricted distributions that coincide with areas of relatively high levels of fishing effort.

Two of the species found to be at medium risk, *B. lata* and *H. galeatus* are large and, as such, a high proportion of individuals encountering TEDs would likely be excluded. Despite this, the escapement rates for these species in the current project was $E = 0$ because there are no published estimates in the scientific literature. These results highlight the need for better escapement estimates to be used in future ERAs. Similarly, the lack of published PTS estimates for the species assessed in the current project requires urgent attention. Ten of the estimates used are at-vessel survival and likely overestimate the PTS of these species. The survival estimates derived in the current study suggest that PTS is variable in elasmobranchs but rarely zero. As such, given the low likelihood of PTS estimates being published for all species, mean PTS from meta-analyses should be used in future ERAs.

The mean PTS of 86.8% for *A. rostrata* is at the upper bounds for batoids assessed using comparable methods in the primary literature. In contrast, a lower proportion of *T. testacea* survived, with the higher survival of females most likely due to their thicker skin which provides protection from males during mating. In excess of 45 species of elasmobranch have the potential to interact with the EKP fishery in southern Queensland. Of these, the scalloped hammerhead (*Sphyrna lewini*) is considered to be Endangered according to the IUCN Red List (Baum *et al.* 2007), while the Endeavour skate (*Dentiraja endeavouri*) and the bluegrey carpetshark (*Brachaelurus colcloughi*) are classified as Vulnerable. As such, assessing the catch and the PTS of these and other species is required to ensure that current levels of fishing effort in the EKP fishery are sustainable in the longer-term.

Further research is required for the seven species assessed as being at medium or high risk in the current study. Life history parameters derived for these and other species from individuals sampled within the fishery area would reduce uncertainty around the sustainable fishing mortality (F_{msm}) estimates, while all seven species have no published estimates of escapement or survival. Similarly, more detailed distribution information for these species would be appropriate to reduce uncertainty around fishing mortality estimates (F_{2015}). Further, the escapement rates published for elasmobranchs in southern Queensland show that TEDs are ineffective for the majority of species: decreases in the bar space of TEDs would go some way to reducing the catch of elasmobranchs in Queensland. As such, research is required to determine the appropriate bar spacing to reduce the catch of elasmobranchs while maintaining the catch rate of target species, particularly Moreton Bay bugs which are becoming more important given the status of Queensland's sea scallop fishery (Yang *et al.* 2016).

Implications

Objective 1 − Quantify the survival of elasmobranchs (i.e. sharks and rays) that are caught incidentally in Queensland prawn trawl nets and discarded

The post-trawl survival estimates derived in the current study represent the first for elasmobranchs derived from short-term confinement in on-board tanks. As such, the derived estimates increase the scientific knowledge in this important field. The survival estimates provide trawl fishers and fishery managers with evidence that elasmobranch survival is variable but not necessarily poor as is assumed in some previous qualitative risk assessments. Where possible, the trawl industry should facilitate further low-cost research to determine PTS in elasmobranchs.

Objective 2 − Quantify reductions in bycatch over the last 20-30 years in the Queensland East Coast Otter Trawl Fishery and describe how these have come about e.g. Fleet reduction, gear technology

The decrease in discards from 87,175 tonnes in 1997 to ~25,200 tonnes in 2014 equates to a ~71% reduction and is likely to be deemed favourably by the broader Australian community, conservation agencies, the Great Barrier Reef Marine Park Authority and the Queensland commercial fishing industry and processors. However, it is also apparent that much of the decline is due to reductions in fleet size and fishing effort, which are generally indicative of a declining industry.

Access to export markets is contingent on Fisheries Queensland meeting discard reduction goals through the Wildlife Trade Operation process undertaken by the Federal Department of the Environment and Energy (DOEE) as part of the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act). Specifically, Fisheries Queensland is required to develop and implement measures to reduce discards in the Queensland East Coast Otter Trawl Fishery (QECOTF). The results from the current study provide clear evidence that Fisheries Queensland has directly addressed this requirement through a range of management measures including effort reduction and the introduction of TEDs and BRDs. This accreditation allows Queensland trawl fishers and seafood processors to export product to overseas markets, increasing the profitability of the fishery. The declining trend in discards indicates that impacts of the fishery on the majority of non-target bycatch species has declined markedly.

Similarly, access to fishing areas within the Great Barrier Reef Marine Park (GBRMP) is conditional on meeting the requirements of the Great Barrier Reef Marine Park Act, including the overarching objective of long-term protection and conservation. About 50% of the QECOTF's catch and effort occur in the GBRMP and management of the fishery in the Park is accredited according to guidelines stipulated under the EPBC Act. One of the guiding principles of the EPBC Act is to "promote ecologically sustainable development through the conservation and ecologically sustainable use of natural resources". The significant reductions in discards provide evidence that the QECOTF is being managed under this guiding principle. According to the Great Barrier Reef Outlook Report 2014, a lack of data represents a source of uncertainty in quantifying the discards. This project directly assesses this lack of data and provides estimates for discards for each sector of the QECOTF.

The reductions in discards quantified in the current project were primarily due to concurrent reductions in trawl effort. The analyses performed as part of the current project demonstrate that the effectiveness of turtle excluder devices (TEDs) and bycatch reduction devices (BRDs) varies significantly between sectors, with the devices having the most impact in the scallop sector. The exclusion of research sampling from the data used to generate the discards as a function of effort (i.e. per boat-day) indicates that the BRDs used by commercial fishers in the early 2000s were effective only in the scallop sector. Although this represents a significant implication from this research, Fisheries Queensland has implemented changes to BRD legislation to improve the effectiveness of devices used by fishers. These changes include the mandatory use of square mesh codend BRDs in the scallop sector, installing BRDs closer to codend drawstrings and the removal of several ineffective BRDs. It is reasonable, therefore, to expect higher discard rates using the effort method than those presented in the current project.

Of the prawn sectors assessed, the shallow water Eastern King Prawn (EKP) sector was found to have high discard rates for all three methods. This sector is the most visible and discards washing up on Sunshine Coast beaches[1] represents a significant challenge for fishery managers. Consequently, the poor exclusion rates for BRDs in this sector require attention.

Objective 3 – Assess the risk that trawling poses to the sustainability of high risk bycatch species, including elasmobranchs, from the Queensland East Coast Otter Trawl Fishery

The ERA conducted as part of the current project satisfies one of the recommendations specified by the DOEE in granting Wildlife Trade Operation accreditation to the QECOTF. This accreditation allows fishers operating in the QECOTF to export products to overseas markets. Further, this accreditation implies that impacts on species affected by the fishery are sustainable. This is the case for the majority (~85%) of elasmobranchs assessed in the current project: that is, the trawl fishery south of the GBRMP poses only low ecological risk to 40 species of elasmobranchs, including the two most common species, *T. testacea* and *A. rostrata*.

Seven species, however, were assessed as being at medium or high risk. As such, steps are required to mitigate this risk, particularly for the high risk species, *Squalus megalops*. Given that effort levels in the deep water EKP sector have remained relatively constant, a reduction in ecological risk can only result from improved estimates of fishing mortality, via escapement and survival estimates, as well as improved life history parameters. An important implication of this work, therefore, is that important metrics used to assess risk are limited for elasmobranchs in southern Queensland which requires immediate attention.

The quantitative method used to assess the risk posed to elasmobranchs by the QECOTF is preferable to qualitative methods used elsewhere. The transfer of knowledge between CSIRO and DAF regarding the Sustainability Assessment for Fishing Effects (SAFE) method of risk assessment represents a significant outcome from the project. This, combined with the improvements made to TrackMapper, provides DAF with the capacity to perform risk assessments for a range of trawl-caught species outside the scope of the current project. A recent review of fisheries management in Queensland[2] indicated that ERAs should be conducted on a range of non-target species to identify impacts on the broader ecosystem and the SAFE method could be employed for this purpose. Further, the SAFE method requires the input of very few people and, as such, represents an efficient and cost-effective way of assessing the risk posed by the QECOTF when compared to qualitative assessments.

No escapement from TEDs (i.e. $E = 0$) and no survival (i.e. $S = 0$) is a function of two separate issues: 1) no published estimates; and 2) published rates being equal to zero. Both of these issues occurred in the current project. Cost and logistical restraints prevent the assessment of these metrics in field-based studies and, as such, methods should be developed for quantifying escapement and survival for species with no published estimates. This could be achieved by using available data for similar species or using mean estimates from a meta-analysis of published rates. Where possible, Fisheries Queensland should provide resources for this work.

The best available information has been used to generate species distributions quantified in the current study. However, the spatial extent of species distributions presented require validation through at-sea sampling. This is a limitation of the methods used. All measures should be used to generate accurate species distributions in the area south of the Great Barrier Reef Marine Park. Further, all steps should be taken to include the entire QECOTF in future ERAs to quantify the impacts across the fishery. Such an ERA would provide information regarding the sustainability of elasmobranchs within the Great Barrier Reef Marine Park and the GBRMPA should provide support and resources for such a study.

[1] https://www.sunshinecoastdaily.com.au/news/from-coolum-to-cooloola-whats-causing-these-dead-f/3145777/
[2] https://publications.qld.gov.au/dataset/f6af65c5-f1f6-48cf-8937-a74ebe467acb/resource/fef6ebad-2317-4c0a-99e1-585598bf1a1e/download/green-paper-on-fisheries-managment-reform-july-2016.pdf

Recommendations

1) As with all ERAs, data quality is a source of uncertainty in assessing the risk posed to catch components of trawl fisheries. This is especially relevant where the components are wholly discarded for regulatory reasons and very little data are available: five of these species that were concluded to be at medium or high risk have no published life history characteristics. Where possible, steps should be taken to improve life history data and distribution information for all species assessed, particularly for those deemed as being at medium or high risk.

2) The ERA should be extended to the entire east coast of Queensland and, if possible, south into northern New South Wales. The results from the current project suggest that localised fishing pressure may be having a detrimental impact on seven species, all of which are distributed across a much wider area than the fishery jurisdiction assessed. The assessment undertaken by (Pears *et al.* 2012) indicated that the QECOTF posed a high risk to 11 species of elasmobranchs and this assessment should be updated with the SAFE, or similar, quantitative methods.

3) The use of $E = 0$ or $S = 0$ is both inaccurate and simplistic where no published information exists on these metrics. No published information exists regarding the escapement (E) and PTS (S) of the seven species assessed at medium or high risk. This represents a significant source of uncertainty to the risk categorisation of these species. Future ERAs should include escapement and survival estimates for similar species or mean estimates derived from a meta-analyses of published escapement and survival rates.

4) Following on from 3, above, resources should be devoted to quantifying the effects of reducing the bar spacing used in TEDs in the QECOTF. A reduction in bar spacing will result in the retention of fewer elasmobranchs, thereby reducing fishing mortality. However, a reduction may also result in a loss of marketable product, especially blue swimmer crabs (*Portunus armatus*), Moreton Bay bugs (*Thenus* spp.) and Balmain bugs (*Ibacus* spp.), which should also be quantified. Where possible, a cost-benefit analysis should be used to determine the appropriate bar spacing across the fishery. A decrease in bar spacing will also result in further reductions in non-elasmobranch discards.

5) Validation of species distribution information is required. A comprehensive trawl survey of the area south of the GBRMP using multiple gear types (prawn trawl, scallop trawl, fish trawl and Danish seine) would confirm the distributions generated as part of this project and improve the estimates of the respective gear efficiency estimates.

6) Provided recommendations 1 − 3 can be implemented, an ERA for elasmobranchs and other nominated species (e.g. Syngnathids, Moreton Bay bugs) should be undertaken at a minimum of every five years to ensure the impacts of the QECOTF are sustainable in the longer term. Subsequent ERAs should incorporate any relevant updated input data such as escapement or survival estimates. This process should include some automation so that annual effort data can be extracted from TrackMapper and fishing mortality estimated for each species in a timely manner. A work group should be convened to decide a list of priority species to be assessed annually made up of shark and ray researchers, conservationists, GBRMPA staff, fishery managers, fishers and DAF researchers.

7) Estimating and monitoring of the production of discards in the QECOTF would be improved if a fishery-observer program was re-introduced to obtain additional and updated measures of discard catches, prawn/target species catches and fishing gear details (i.e. net sizes, BRD details and trawl duration, etc.).

Extension and Adoption

The results of this research have been extended to Fisheries Queensland through the Steering Committee meetings held as part of the project. Additionally, the PI has been in contact with Fisheries Queensland staff throughout the course of the project. The trawl industry has been updated via the presence of David Sterling, of Moreton Bay Seafood Industry Association (MBSIA), on the Steering Committee. Minutes from the first Steering Committee meeting were also sent to the Queensland Seafood Industry Association (QSIA), who declined to be part of the Steering Committee. The QSIA will be sent a copy of the final report.

It is expected that three scientific manuscripts will be generated from the project: 1) results of the post-trawl survival work (Objective 1) has been completed and will be submitted to Marine and Freshwater Research before 30 May 2017; 2) the reductions in discards (Objective 2) is close to completion and will be submitted to ICES Journal of Marine Science before 30 June 2017; and 3) results from the ERA using data from 2016/17, scheduled to be submitted by 30 June 2018.

A second media release describing the reduction in discards generated by the QECOTF will be released through DAF's communication unit. The media release will be disseminated widely. Further, the PI has been accepted as a PhD candidate at James Cook University and, as such, results of this work will be published in the submitted thesis.

This project has produced the first statistically robust estimates of total discards from the QECOTF, including long term trends. It is expected that the results from this project will be used by Fisheries Queensland to demonstrate the impacts of the QECOTF and the effects of the management measures implemented in the early 2000s. The reduction in discards quantified provides evidence to the GBRMPA and the federal Department of the Environment and Energy that Fisheries Queensland and the trawl operators have made significant advances in reducing the impact of the QECOTF on non-target species and the ecosystems in which it operates. This will result in continued access to fishing grounds within the GBRMP and also to lucrative overseas markets through the WTO process.

Project coverage

A project media release authorised by FRDC on 16 July 2016 was distributed to news outlets on 8 July 2016. A small article appeared in the News Mail from Bundaberg on 8 July 2016, below. Further, the PI was interviewed by Naomi Lynch from Grant Broadcasters. The interview was aired on 12 July 2016 across the Grant network of radio stations including:

- Zinc 96.1FM, Sunshine Coast;
- Hitz 93.9FM, Bundaberg;
- 4RO, Rockhampton;
- 4MK and Star 101.9 FM, Mackay;
- Star 106.3 FM, Townsville; and
- Star 102.7 FM and 4CA, Cairns.

Media release

17 June 2016

Research underway on Queensland’s east coast trawl fishery

A collaborative research project is underway to quantify the amount of discards in the Queensland trawl fishery from 1988 to 2014. The project involving the Queensland Department of Agriculture and Fisheries (DAF) and the Commonwealth Scientific and Industrial Research Organisation (CSIRO) is co-funded by the Federal Government’s Fisheries Research and Development Corporation (FRDC).

Project leader and DAF scientist, Matthew Campbell, said data had been collected as part of previous research projects and during the Queensland government’s Fishery Observer Program.

“The data contains information about the weight of the target catch and discards, as well as other parameters such as net size, hours fished and trawl speed,” Mr Campbell said.

“Using information from around 4000 measurements of discards by fishery observers and scientists, we are able to estimate the total tonnage of discards produced in the fishery from 1988 to 2014.

“Three separate analyses will be undertaken to determine discards as a function of catch, effort and swept area.

“Project staff can then assess the effects of various management changes on the amount of discards including the introduction of turtle excluder devices (TEDs) and bycatch reduction devices (BRDs), reductions in fishing effort and changes in fleet dynamics.

“We can also assess the accuracy of each method and provide advice as to the best method for calculating measures of discards in this and other trawl fisheries.”

Mr Campbell said the mandatory use of TEDs and BRDs in the early 2000s combined with fleet reductions as a result of licence buy-backs has likely led to significant reductions in the amount of non-target species returned to the sea.

“The reduction in discards achieved as a result of management changes, however, remains undetermined,” he said.

Mr Campbell said the project team will also conduct quantitative ecological risk assessments for some species of sharks and rays. “A recent ecological assessment suggested that prawn and scallop trawling may impose a high risk to several species of sharks and rays within the Great Barrier Reef Marine Park.

“These species exhibit life history characteristics such as slow growth and low fecundity, making their populations more susceptible to fishing impacts.

“Sampling aimed at collecting critical information for the risk assessments, including measuring the survival of sharks and rays after they are returned to the sea, is currently underway.

“We expect to provide stakeholders with estimates of trawl fishing effort levels that ensure the long-term sustainability of several species of sharks and rays that are common on Queensland’s trawl grounds.

“Trawl fishery managers are required to ensure the sustainability of all species that are affected by the trawl fishery, not just the targeted species such as prawns, scallops and Moreton Bay bugs.”

The project is due to be completed by September 2017.

Contact: Matthew Campbell at the EcoSciences Precinct, Dutton Park on 07 3255 4229 or matthew.campbell@daf.qld.gov.au.

RESEARCH: Prawn trawler.

East coast trawl research

A PROJECT is underway to quantify the amount of discards in the Queensland trawl fishery to protect sharks and rays.

The project, involving the Queensland Department of Agriculture and Fisheries and CSIRO, is co-funded by the Federal Government.

DAF scientist, Matthew Campbell, said data was collected as part of previous research and during the Queensland Government's Fishery Observer Program.

"The data contains information about the weight of the target catch and discards, as well as other parameters such as net size, hours fished and trawl speed," Mr Campbell said.

The project will be completed by September.

Figure 8: Clipping from the News Mail, Bundaberg on 8 July 2016.

Project materials developed

At least two scientific journal articles will be submitted before 30 June 2017. These will be appended to the final report.

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Appendices

Appendix 1 – Project staff

Matthew Campbell (DAF), Tony Courtney (DAF), Na Wang (DAF), Marl McLennan (DAF), Shijie Zhou (CSIRO) and Sean Maberly (DAF)

Appendix 2 – Intellectual Property

No intellectual property has been generated from this project.

Appendix 3 – Sample code

Below is the 'R' code used to run WinBUGS to determine Q_x in the ERA section on page 9:

```
library(R2WinBUGS)

pois.inits <- function() {
 list(mu = 150,
    q = rep(0.1, 0.1, length(unique(all7$gear))),
       Den = rep(950,950,length(unique(all7$ymg))),
       den2 = rep(950,950,length(all7$catch))
 )
}

#####___Poisson Model____####
sink("poisson model.txt")
cat("
  model {
  mu ~ dlnorm(1, 0.1)
  for (k in 1:4) { #Change to number of gear types assessed
  q[k] ~ dbeta(1,1)
   }

  for (i in 1:78) { #Change to number of year x month x gear combinations
       Den[i] ~ dpois(mu)
  }

  for (i in 1:805) { #change according to length of data
       den2[i] ~ dpois(Den[ymg[i]]) #ymg: year.month.grid index
       N[i] <- den2[i] * swpA[i]
       catch[i] ~ dbin(q[gear[i]], N[i])
       #      pred.c[i] ~ dbin(q[gear[i]], N[i])
       est.c[i]<- q[gear[i]] * N[i]
  }
  Den.mean<- mean(Den[])

}
",fill = TRUE)
sink()

params <- c("q", "mu", "est.c")


poisson <- bugs(data = mylio7, # change according to species, genus, family, etc
      inits    = pois.inits,
      parameters = params,
      model    = "poisson model.txt",
      n.thin    = 1,
      n.chains   = 3,
      n.burnin   = 190000,
      n.iter    = 200000,
      debug = TRUE,
      bugs.directory = MyWinBugsDir)
```

Appendix 4 – Scientific and common names of species assessed

Table 6: Species names and common names of the 47 species evaluated in the current ecological risk assessment. Shark names follow Last and Stevens (2009) and ray names follow Last *et al.* (2016).

Species	Common name
Aetomylaeus nichofii	Banded Eagle Ray
Aptychotrema rostrata	Eastern shovelnose Ray
Asymbolus analis	Grey Spotted Catshark
Asymbolus rubiginosus	Orange Spotted Catshark
Bathytoshia brevicaudata	Smooth Stingray
Bathytoshia lata	Brown Stingray
Brachaelurus colcloughi	Colclough's Shark
Carcharhinus amboinensis	Pigeye Shark
Carcharhinus brevipinna	Spinner Shark
Chiloscyllium punctatum	Grey Carpetshark
Dentiraja australis	Sydney Skate
Dentiraja endeavouri	Endeavour Skate
Dipturus melanospilus	Blacktip Skate
Figaro boardmani	Sawtail Shark
Glaucostegus typus	Giant Guitarfish
Gymnura australis	Australian Butterfly Ray
Hemigaleus australiensis	Australian Weasel Shark
Hemitrygon fluvorium	Estuary Stingray
Heterodontus galeatus	Crested Hornshark
Hydrolagus ogilby	Ogilby's Ghostshark
Hypnos monopterygius	Coffin Ray
Loxodon macrorhinus	Sliteye Shark
Maculabatis astra	Blackspotted Whipray
Maculabatis toshi	Brown Whipray
Mustelus walkeri	Eastern Spotted Gummy Shark
Neotrygon leylandi	Painted Maskray
Neotrygon picta	Speckled Maskray
Neotrygon trigonoides	Coral Sea Maskray
Orectolobus maculatus	Spotted Wobbegong
Orectolobus ornatus	Ornate Wobbegong
Parascyllium collare	Collar Carpetshark
Rhizoprionodon acutus	Milk Shark
Rhizoprionodon taylori	Australian Sharpnose Shark
Rhynchobatus australiae	Bottlenose Wedgefish
Sphyrna lewini	Scalloped Hammerhead
Squalus grahami	Eastern longnose Spurdog
Squalus megalops	Piked Spurdog
Squalus montalbani	Philippine Spurdog
Squatina albipunctata	Eastern Angelshark
Stegostoma fasciatum	Zebra Shark
Trygonoptera testacea	Common Stingaree
Trygonorrhina fasciata	Eastern Fiddler Ray
Urolophus bucculentus	Sandyback Stingaree
Urolophus flavomosaicus	Patchwork Stingaree
Urolophus kapalensis	Kapala Stingaree
Urolophus sufflavus	Yellowback Stingaree
Urolophus viridis	Greenback Stingaree

Appendix 5 – Data sources

Table 7: Information used to generate the species distributions for use in the ecological risk assessment. Depth is in metres.

Species	Depth range	Reference(s)	Latitude range	Reference(s)
Aetomylaeus nichofii	0 – 115	4, 1	Hervey Bay, north	4
Aptychotrema rostrata	0 – 100	1, 2, 3	Halifax Bay, QLD, south	1
Asymbolus analis	25 – 200	1, 4	Cape Moreton, south	1
Asymbolus rubiginosus	25 – 200	1	Cape Moreton, south	1, 2
Bathytoshia brevicaudata	0 – 150	1	Maroochydore, south	1
Bathytoshia lata	0 – 360	1	Double Island Point, south	2
Brachaelurus colcloughi	0 – 100	1, 2	Gladstone, south	1
Carcharhinus amboinensis	0 – 100	1	Moreton Bay, north	1
Carcharhinus brevipinna	0 – 75	1	Jervis Bay, north	1, 2
Chiloscyllium punctatum	0 – 90	1, 2, 3	Sandon River, north	1, 2
Dentiraja australis	22 – 325	1	Double Island Point, south	2
Dentiraja endeavouri	120 – 290	1, 2	Fraser Island, south	1, 2
Dipturus melanospilus	240 - 700	1, 2	Broken Bay, north	1, 2
Figaro boardmani	150 – 640	1	Noosa, south	1, 2
Glaucostegus typus	0 – 100	1, 2	Forster, north	1, 2
Gymnura australis	0 – 250	1, 2	Broken Bay, north	1, 2
Hemigaleus australiensis	0 – 130	1, 2	Brunswick Heads, north	1, 2
Hemitrygon fluvorium	0 – 30	1, 2	Repulse Bay, south	1, 2
Heterodontus galeatus	0 – 90	1	Mooloolaba, south	1, 3
Hydrolagus ogilby	160 - 700	1, 3	Cairns, south	1
Hypnos monopterygius	0 – 220	1, 2	Heron Island, south	1, 2, 4
Loxodon macrorhinus	0 – 100	1, 2	Moreton Bay, north	1, 2
Maculabatis astra	0 – 140	1, 2	Moreton Bay, north	1
Maculabatis toshi	0 – 20	2	Clarence River, north	1
Mustelus walkeri	50 – 400	1	Moreton Island, north	1
Neotrygon leylandi	10 – 90	1, 4	Hervey Bay, north	1
Neotrygon picta	0 – 100	1	Hervey Bay, north	1
Neotrygon trigonoides	0 – 90	1, 2	Port Stephens, NSW, north	1, 2
Orectolobus maculatus	0 – 220	1, 4	Swains Reefs, south	1
Orectolobus ornatus	0 – 100	1	Sydney, north	1, 2
Parascyllium collare	20 - 175	1	Mooloolaba, south	1
Rhizoprionodon acutus	0 – 200	1	Fraser Island, north	1
Rhizoprionodon taylori	0 – 100	1	Double Island Point, north	2
Rhynchobatus australiae	0 – 60	1	Crowdy Head, NSW, north	1, 2
Sphyrna lewini	0 – 285	1	Sydney, north	1
Squalus grahami	220 – 450	1	Bermagui, NSW, north	1
Squalus megalops	30 – 580	1, 2	Whitsundays, south	1
Squalus montalbani	290 – 670	1	Townsville, south	1
Squatina albipunctata	35 – 415	1	Cairns, south	1
Stegostoma fasciatum	0 – 50	2	Montague Island, north	1
Trygonoptera testacea	0 – 90	2, 3	Sandy Cape, south	1, 3
Trygonorrhina fasciata	0 – 100	1	Victoria to Noosa	1, 2
Urolophus bucculentus	65 – 265	1	Stradbroke Island, south	1
Urolophus flavomosaicus	60 – 320	1	Caloundra, north	1
Urolophus kapalensis	10 – 130	1, 2.	To Cape Moreton	1, 2
Urolophus sufflavus	45 – 320	1	Cape Moreton, south	1
Urolophus viridis	60 – 300	1	Stradbroke Island, south	1

1 – Last and Stevens (2009); 2 – Atlas of living Australia; 3 – Catch data from Courtney *et al.* (2007b); 4 – Kyne *et al.* (2005)

Table 8: Metrics used to estimate natural mortality (M) and resulting F_{msm} for use in the ecological risk assessment. L_∞ and k are the von Bertalanffy growth parameters, L_{max} is the maximum reported length, t_{max} is the maximum reported age and t_{mat} is the age-at-maturity. Lengths and widths are in centimetres, ages are in years.

Species	L_{max}/W$_{max}$	L_∞	k	t_{max}	t_{mat}	References
Aetomylaeus nichofii	65	67.3	0.22	13	3	Fishbase (accessed 26/09/2016)
Aptychotrema rostrata	120	103	0.16	18	3.8	Fishbase (accessed 23/09/2016)
Asymbolus analis	90	92.7	0.14	20.5	4.4	Fishbase (accessed 26/09/2016)
Asymbolus rubiginosus		41	0.28	10.2	2.5	Fishbase (accessed 26/09/2016)
Bathytoshia brevicaudata	430	432.1				Fishbase (accessed 26/09/2016)
Bathytoshia lata	400	402.4				Fishbase (accessed 26/09/2016)
Brachaelurus colcloughi	76	78.5				Fishbase (accessed 26/09/2016)
Carcharhinus amboinensis		267.2	0.145	26.4	4.9	Tillett *et al.* (2011)
Carcharhinus brevipinna		200	0.14	13.8	2.6	Carlson and Baremore (2005)
Chiloscyllium punctatum	132	135.2				Fishbase (accessed 26/09/2016)
Dentiraja australis	50	52	0.29	9.9	2.3	Fishbase (accessed 26/09/2016)
Dentiraja endeavouri	32.1	33.6	0.38	7.5	1.9	Fishbase (accessed 26/09/2016)
Dipturus melanospilus	77.7	80.2	0.19	15.1	3.3	Fishbase (accessed 26/09/2016)
Figaro boardmani	61	63.2	0.2	14.3	3.3	Fishbase (accessed 26/09/2016)
Glaucostegus typus		277.0	0.3	41.5	7.7	White *et al.* (2014)
Gymnura australis	73	75.5				Fishbase (accessed 26/09/2016)
Hemigaleus australiensis	110	113				Fishbase (accessed 26/09/2016)
Hemitrygon fluvorium		147.5	0.03	25	13.4	Pierce and Bennett (2011)
Heterodontus galeatus	152	155.3	0.05	58.1	11.9	Fishbase (accessed 26/09/2016)
Hydrolagus ogilby	88	90.7	0.2	14.4	3.1	Fishbase (accessed 26/09/2016)
Hypnos monopterygius	70	72.4				Fishbase (accessed 26/09/2016)
Loxodon macrorhinus		85.3	0.32	6.5	1.4	Gutteridge *et al.* (2013)
Maculabatis astra		82.18	0.073	29	8.67	Jacobsen and Bennett (2011)
Maculabatis toshi	86	88.7				Fishbase (accessed 26/09/2016)
Mustelus walkeri		224.5	0.033	18	3.9	Rigby *et al.* (2016b)
Neotrygon leylandi	25	26.3				Fishbase (accessed 26/09/2016)
Neotrygon picta		36.05	0.08	18	13.94	Jacobsen and Bennett (2010)
Neotrygon trigonoides	70	44.2	0.08	14	16.4	Jacobsen and Bennett (2010)
Orectolobus maculatus		163	0.09	14.5	2.8	Huveneers *et al.* (2013)
Orectolobus ornatus		99.9	0.14	13.8	2.6	Huveneers *et al.* (2013)
Parascyllium collare	85	87.6				Fishbase (accessed 26/09/2016)
Rhizoprionodon acutus		86.1	0.63	14.4	1.8	Harry *et al.* (2010)
Rhizoprionodon taylori		73.25	1.013	4.3	1	Simpfendorfer (1993)
Rhynchobatus australiae		204.5	0.41	41.5	7.6	White *et al.* (2014)
Sphyrna lewini		319.7	0.249	28.5	4.1	Chen *et al.* (1990)
Squalus grahami	60.2	62.4	0.1	28.6	6.5	Fishbase (accessed 26/09/2016)
Squalus megalops		82.9	0.034	25	19.1	Braccini *et al.* (2007)
Squalus montalbani		63.24	0.0071	28	21.8	Rigby *et al.* (2016a)
Squatina albipunctata	98.5	101.3	0.19	15.1	3.2	Fishbase (accessed 26/09/2016)
Stegosoma fasciatum	354	356.8				Fishbase (accessed 26/09/2016)
Trygonoptera testacea	61	49	0.09	32	7.5	Fishbase (accessed 23/09/2016)
Trygonorrhina fasciata	108	112.9	0.13	26.6	4	Izzo and Gillanders (2008)
Urolophus bucculentus	80	82.6	0.06	47.8	10.4	Fishbase (accessed 26/09/2016)
Urolophus flavomosaicus	59	61.2	0.08	35.7	8.2	Fishbase (accessed 26/09/2016)
Urolophus kapalensis	52	45	0.09	32	20	Fishbase (accessed 23/09/2016)
Urolophus sufflavus	42	43.8	0.1	28.4	6.8	Fishbase (accessed 26/09/2016)
Urolophus viridis	44	45.8	0.1	28.5	6.8	Fishbase (accessed 26/09/2016)

Appendix 6 – Supplementary results

Objective 1

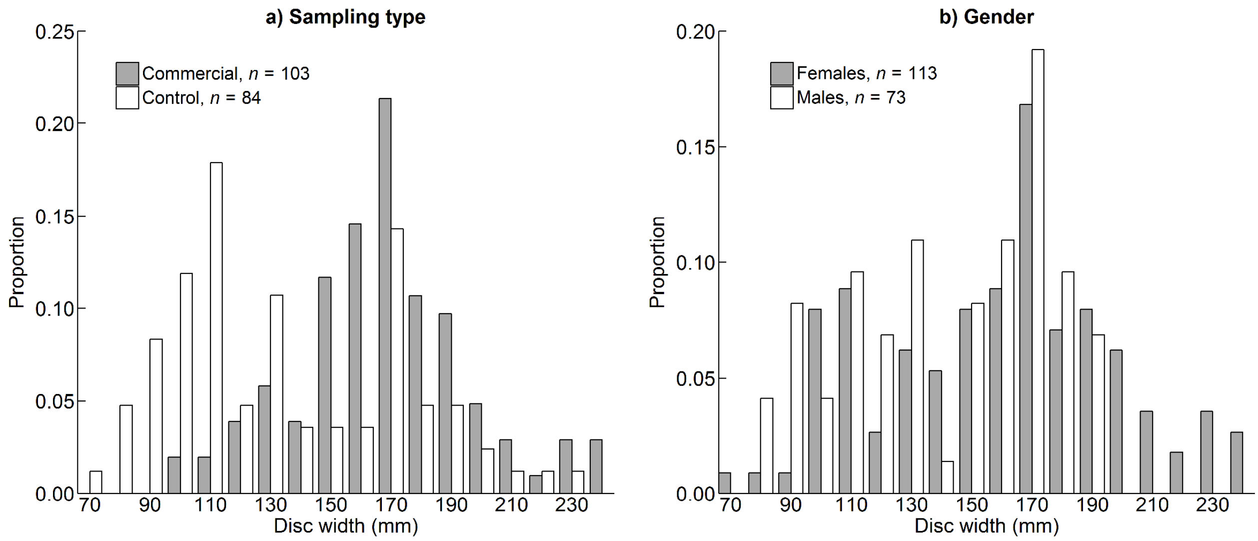


Figure 9: Length frequency distributions for 187 Common Stingarees (*Trygonoptera testacea*) caught during three post-release survival experiments as a function of a) sampling type (control vs. commercially trawled) and b) gender. Note: gender was not recorded for five individuals.

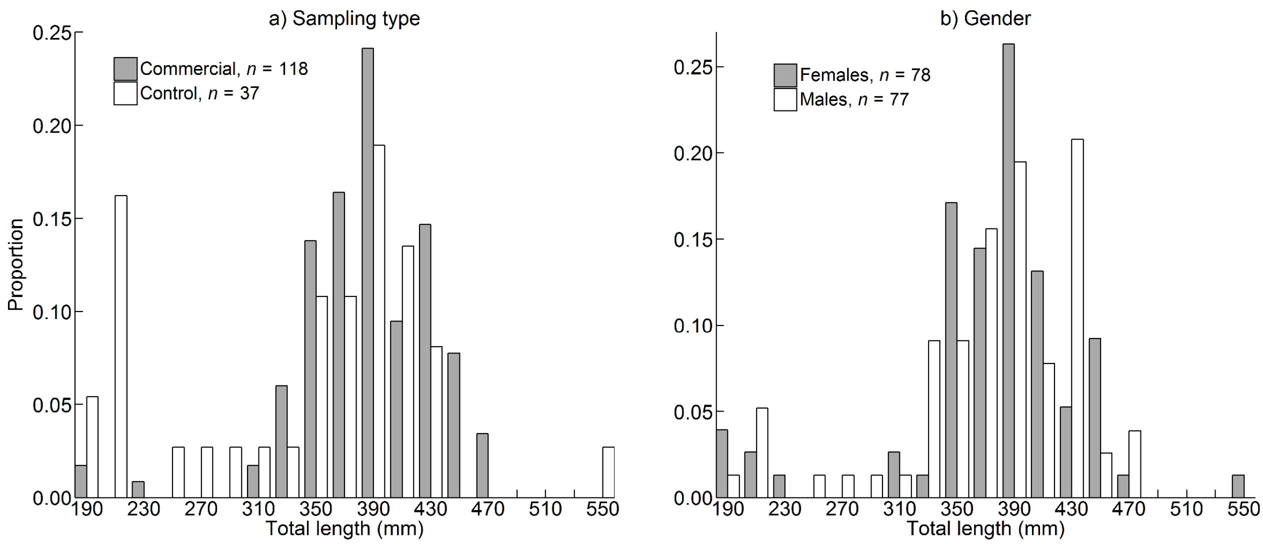


Figure 10: Length frequency distributions for 155 Eastern Shovelnose Rays (*Aptychotrema rostrata*), caught during three post-release survival experiments, as a function of a) sampling type (control vs. commercially trawled) and b) gender.

Table 9: Number of individual *Trygonoptera* testacea and *Aptychotrema rostrata* caught by each vessel during the three post-release survival experiments conducted in 2016. Also shown is mean trawl duration and their standard errors (SE), in minutes.

Trip	Night	*Tom Marshall* (controls)			*C-Rainger* (commercially trawled)		
		No. trawls (duration ± SE)	*A. rostrata*	*T. testacea*	No. trawls (duration ± SE)	*A. rostrata*	*T. testacea*
1	1	20 (14 ± 0.71)	13	35	3 (158 ± 29.52)	12	0
	2	-	-	-	4 (145 ± 4.12)	14	0
2	1	11 (11 ± 0.24)	20	57	5 (81 ± 3.46)	67	52
	2	-	-	-	3 (87 ± 0.38)	2	45
3	1	5 (11 ± 0.20)	4	27	3 (66 ± 1.15)	23	6

Table 10: Post-trawl survival of *Trygonoptera testacea* and *Aptychotrema rostrata* as a function of Condition Index, described by Enever *et al.* (2009) as: 1 = Dead or nearly dead, no body movement, slight movement of spiracles; 2 = limp wing and/or wing movement; some spiracle movement; and 3 = vigorous wing and/or body movement; rapid spiracle movement. Also shown are the number of individuals from each species in each category.

Condition index	*T. testacea*		*A. rostrata*	
	PTS (S.E)	*n*	PTS (S.E)	*n*
1	18.2 (11.6)	11	50.0 (14.4)	12
2	32.4 (4.6)	105	85.1 (3.5)	101
3	38.0 (5.8)	71	90.5 (4.5)	42

Objective 2

Table 11: Parameter estimates and standard errors from three GLMs (retained catch, fishing effort and swept area) for the banana prawn, tiger and endeavour prawn, deep water (>50 fathoms) eastern king prawn, red spot king prawn and scallop sectors. The reference level for fishing sector was the shallow (<50 fathoms) water eastern king prawn fishery.

Sector	Retained catch			Effort			Swept area		
	β	S.E.	Pr(>∣t∣)	β	S.E.	Pr(>∣t∣)	β	S.E.	Pr(>∣t∣)
Banana prawn	6.14	6.58	0.351	-2.18	0.57	<0.05	1.74	4.13	0.674
Eastern king prawn (deep)	-30.87	17.75	0.082	-1.89	0.46	<0.05	-11.80	10.68	0.269
Moreton Bay	-46.60	26.83	0.082	n/a	n/a	n/a	-21.24	15.95	0.183
Red spot king prawn	8.65	7.83	0.269	-1.96	0.55	<0.05	3.56	4.88	0.465
Scallop	-14.28	6.65	<0.05	-1.42	0.62	<0.05	-6.04	5.42	0.265
Tiger and endeavour prawn	-33.14	20.96	0.114	-1.52	0.60	<0.05	-13.99	12.60	0.267
BRDs	-0.28	0.07	<0.05	0.16	0.27	0.553	-0.29	0.05	<0.05
Sampling program (Charter)	21.96	12.72	0.084	n/a	n/a	n/a	11.32	7.58	0.136
Lunar phase	-38.55	27.94	0.168	-0.08	0.10	0.468	-15.86	17.02	0.352
Lunar phase advanced	22.94	12.59	0.068	-0.11	0.10	0.266	10.45	7.43	0.159
Banana prawn with BRDs	-0.20	0.10	<0.05	-0.25	0.35	0.478	-0.02	0.08	0.814
Eastern king prawn (deep) with BRDs	0.28	0.09	<0.05	-0.10	0.31	0.745	0.41	0.07	<0.05
Scallop with BRDs	-0.56	0.08	<0.05	-1.30	0.35	<0.05	-0.58	0.06	<0.05
Tiger and endeavour prawn with BRDs	0.20	0.07	<0.05	-0.21	0.29	0.469	0.13	0.05	<0.05
Trawl shot number			<0.05						<0.05
Trip						<0.05			

Table 12: Parameter estimates and standard errors from the stout whiting sector GLMs. The reference level was the stout whiting trawl fishery.

Sector	Retained catch			Effort			Swept area		
	β	S.E.	Pr(>∣t∣)	β	S.E.	Pr(>∣t∣)	β	S.E.	Pr(>∣t∣)
Danish seine	2.68	0.04	<0.05	1.05	0.58	0.075	0.86	0.38	<0.05
Lunar	-0.15	0.04	<0.05	-0.43	0.72	0.549	-0.69	0.45	0.124
Lunar advanced	-1.05	0.03	<0.05	-0.83	0.63	0.192	-0.72	0.40	0.071
Trip			<0.05			<0.05			
Trawl shot number							-6.04	5.42	<0.05

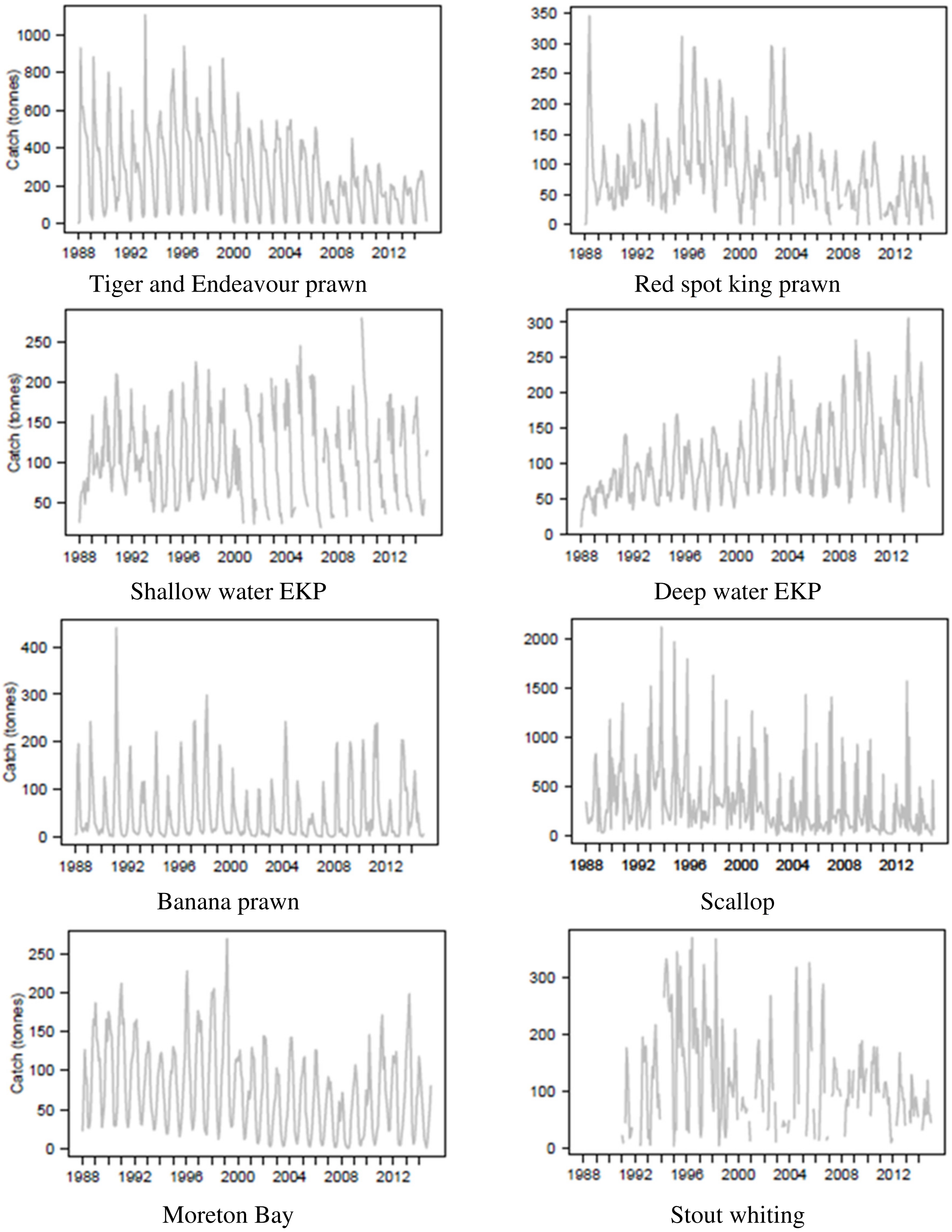


Figure 11: Monthly summation of retained catch in each sector from January 1988 to December 2014. Noteworthy trends include the long-term decline in catch from the tiger and endeavour prawn sector, and the increasing catch from the deep water eastern king prawn fishery. Scallop catch is in adductor muscle meat weight.

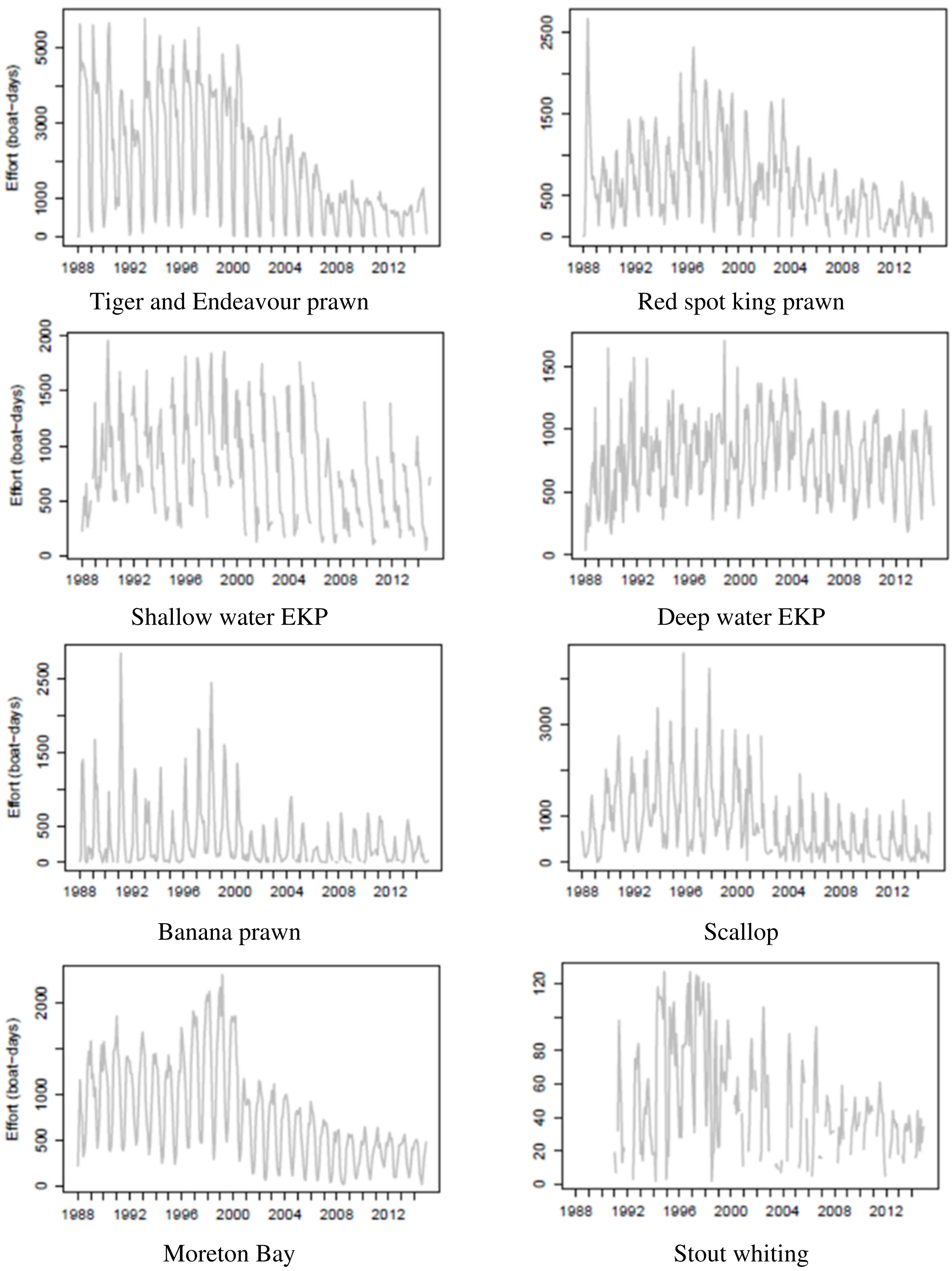


Figure 12: Monthly summation of fishing effort (boat-days) in each sector from January 1988 to December 2014. Fishing effort has declined in most sectors since the logbook program commenced in 1988, especially in the tiger and endeavour prawn, scallop and Moreton Bay sectors.

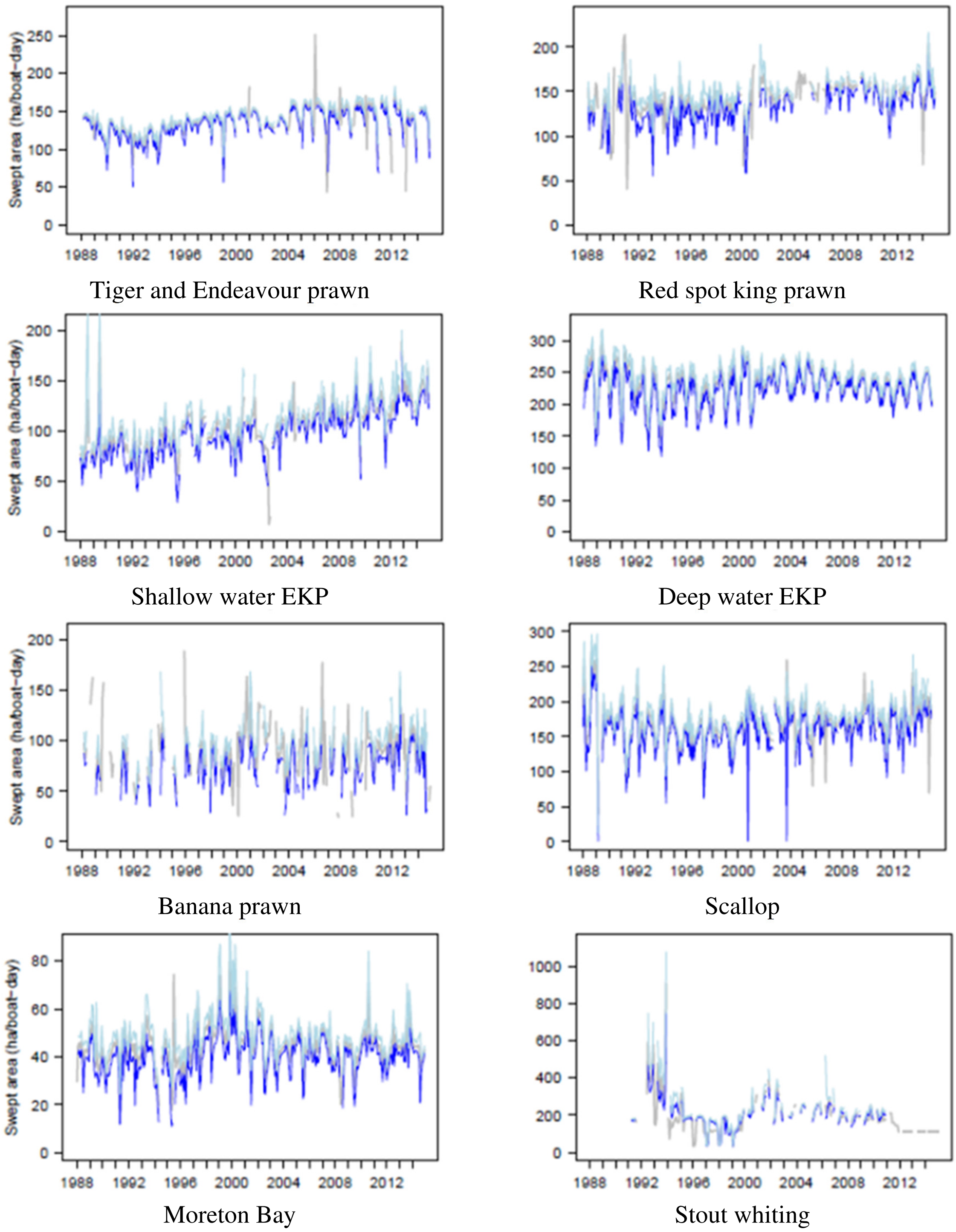


Figure 13: The mean daily swept area rate (ha boat-day-1) for each month and sector from January 1988 to December 2014. The upper (light blue) and lower (dark blue) 95% confidence intervals are based on 100,000 bootstrap samplings from different vessels in each sector for each month. Where swept area estimates were available from fewer than five vessels, no confidence intervals are provided.

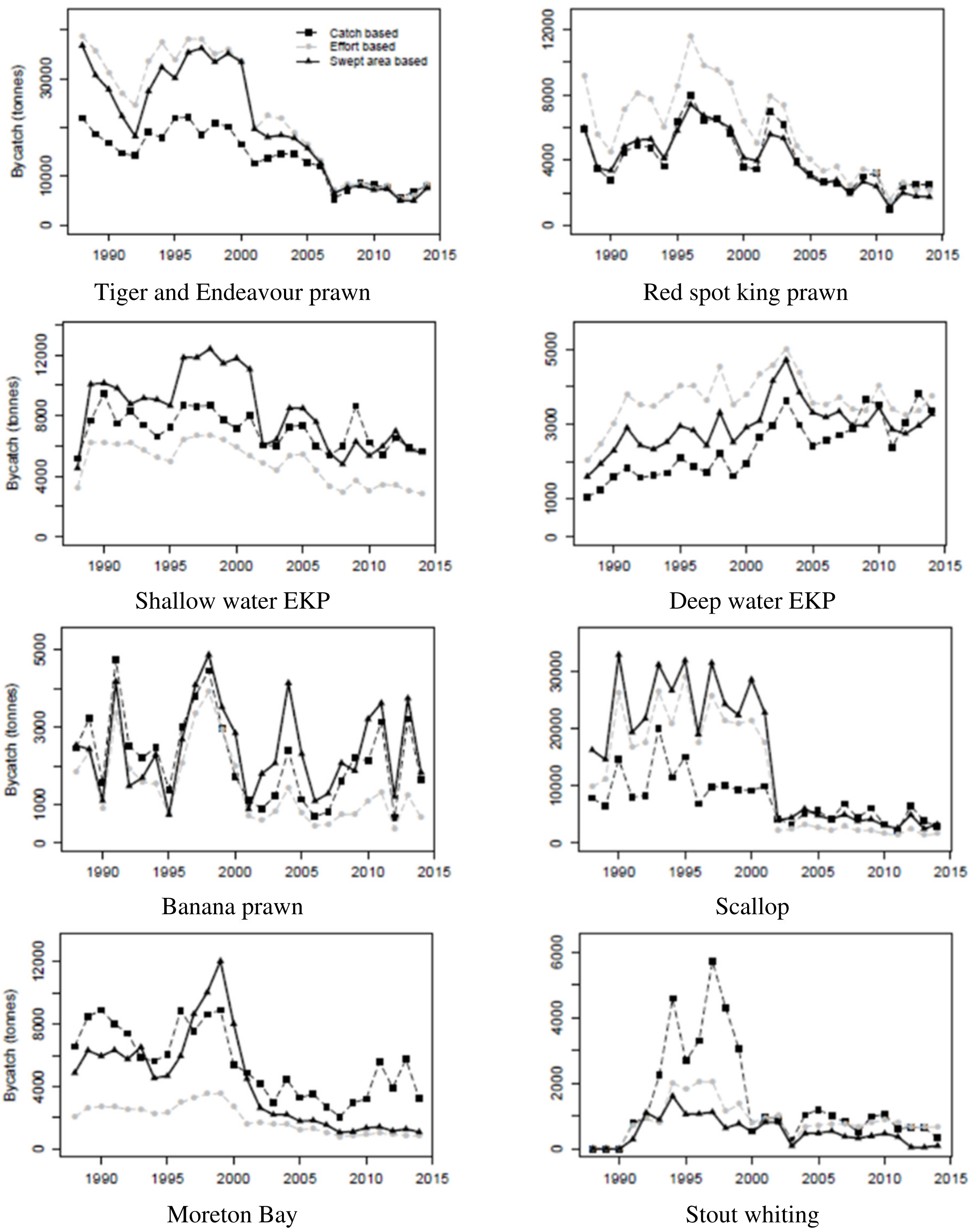
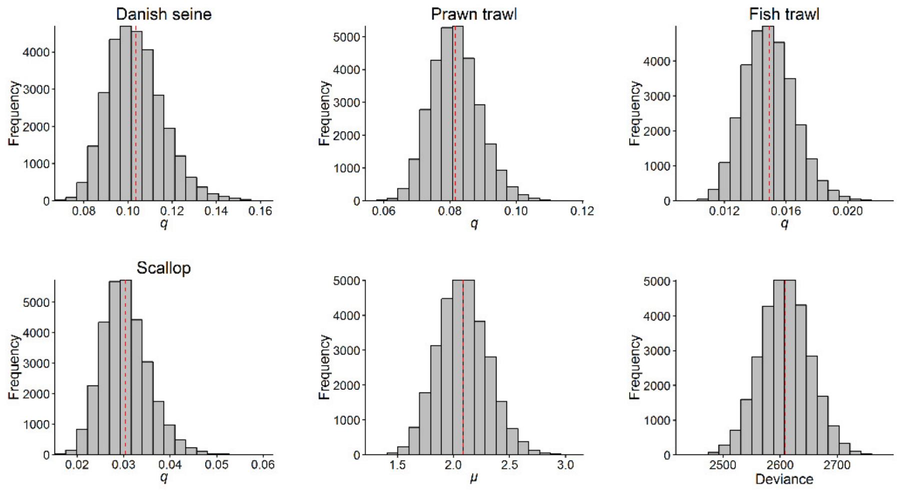
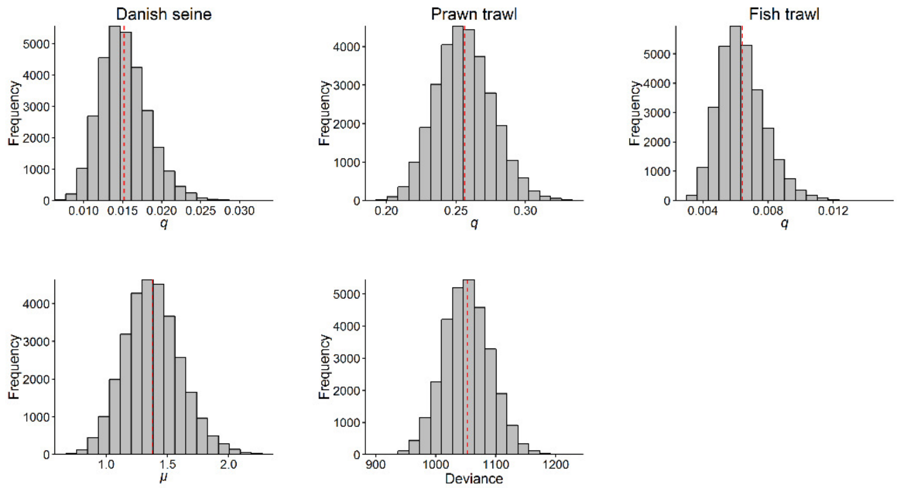


Figure 14: Total annual discard estimates (t) for the Queensland east coast otter trawl fishery sectors from January 1988 to December 2014, using three methods (retained catch, effort and swept area).

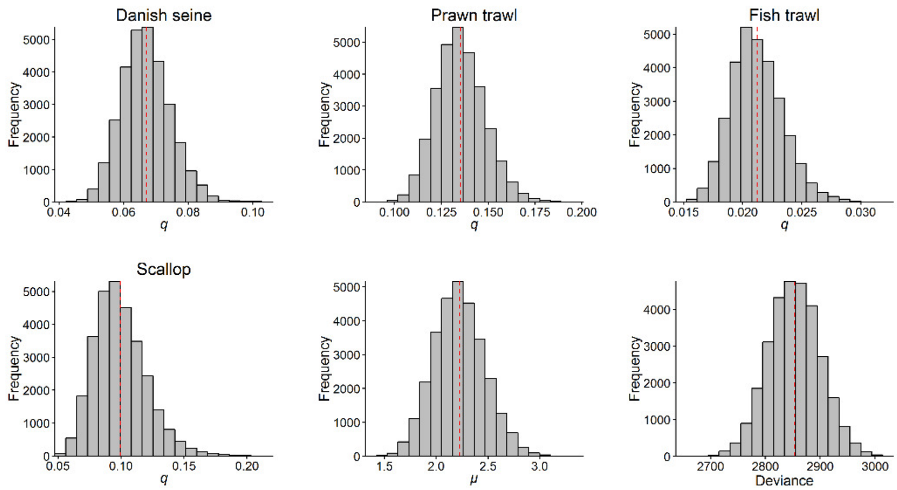
Objective 3



a) *Aptychotrema rostrata*

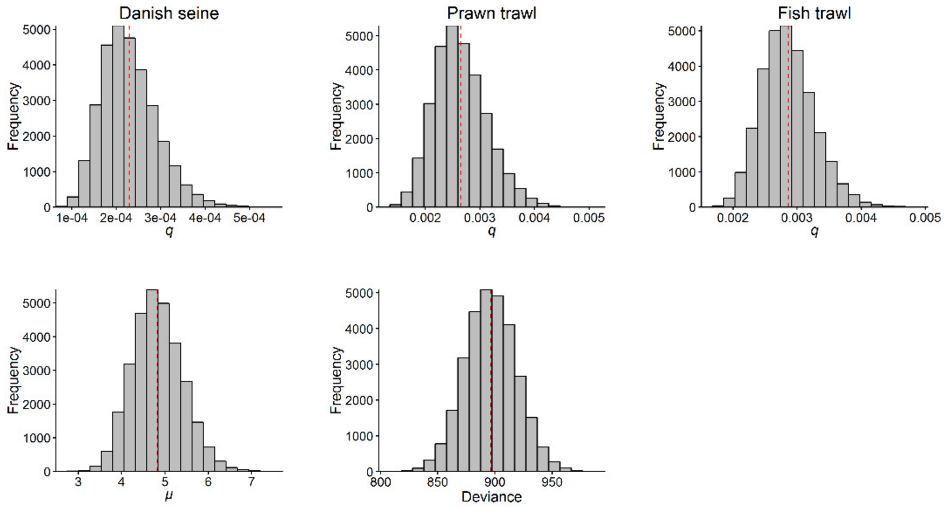


b) *Trygonoptera testacea*

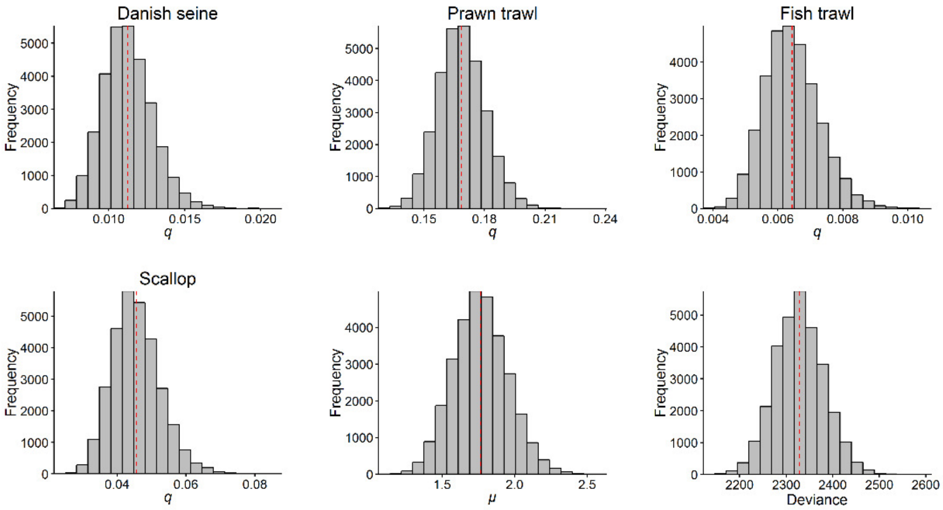


c) Batoidea

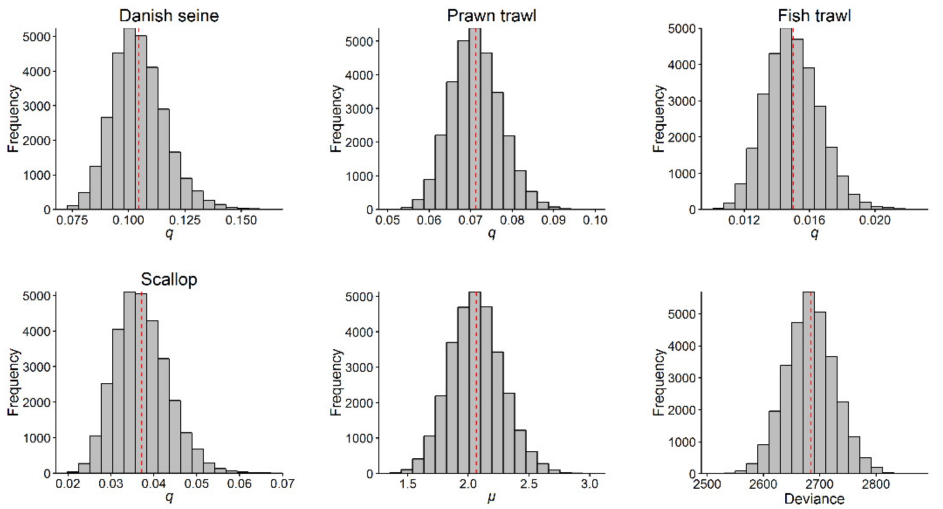
Figure 15: Posterior distribution of parameters estimated using the WinBUGS code on page 44 for a) *Aptychotrema rostrata*, b) *Trygonoptera testacea*, and c) Batoids. μ represents the mean density from at-sea sampling and q is the derived efficiency of the respective gear types. Red vertical lines represent the mean values.



a) Carcharhinids



b) Myliobatiformes



c) Rhinopristiformes

Figure 16: Posterior distribution of parameters estimated using the WinBUGS code on page 44 for a) Carcharhinids, b) Myliobatiformes, and c) combined Rhinopristiformes. μ represents the mean density from at-sea sampling and q is the derived efficiency of the respective gear types. Red vertical lines represent the mean values.

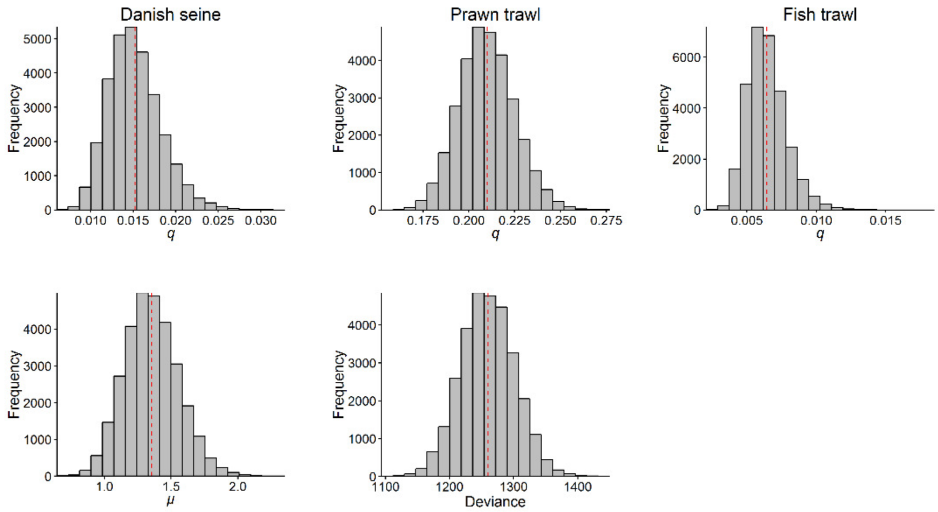


Figure 17: Posterior distribution of parameters estimated using the WinBUGS code on page 44 for all Urolophids. μ represents the mean density from at-sea sampling and q is the derived efficiency of the respective gear types. Red vertical lines represent the mean values.

Table 13: Metrics used to calculate instantaneous fishing mortality rate, *F*, for use in the ecological risk assessment. *n* is the number of individuals caught from 1,175 trawl shots during research surveys and observer-based sampling. 'Group' represents the classification level of each species at which the gear efficiency, *Q*, value was quantified where: S = Species, M = Myliobatiformes, R = Rhinopristiformes, U = Urolophidae, B = Batoidea, A = All sharks, rays and skates. *Q* is the mean gear efficiency, *E* is the escapement probability and *S* is the proportion surviving capture.

Species	*n*	Group	*Q* Danish seine	*Q* Prawn trawl	*Q* Fish trawl	*Q* Scallop	*E*	*S*
Aetomylaeus nichofii	18	M	0.0114	0.1551	0.0067	0.0571	0	0
Aptychotrema rostrata	3933	S	0.1037	0.0820	0.0150	0.0306	0	0.8[1]
Asymbolus analis	31	A	0.0669	0.1282	0.0260	0.1024	0	0.875[3]
Asymbolus rubiginosus	21	A	0.0669	0.1282	0.0260	0.1024	0	0.539[3]
Bathytoshia brevicaudata	46	M	0.0114	0.1551	0.0067	0.0571	0.23[2]	0
Bathytoshia lata	16	M	0.0114	0.1551	0.0067	0.0571	0	0
Brachaelurus colcloughi	8	A	0.0669	0.1282	0.0260	0.1024	0	0
Carcharhinus amboinensis	1	A	0.2266	0.1282	0.0260	0.1024	0	0
Carcharhinus brevipinna	7	A	0.0669	0.1282	0.0260	0.1024	0	0
Chiloscyllium punctatum	46	A	0.0669	0.1282	0.0260	0.1024	0.27[2]	0
Dentiraja australis	6	A	0.0669	0.1282	0.0260	0.1024	0	0
Dentiraja endeavouri	128	A	0.0669	0.1282	0.0260	0.1024	0	0.242[3]
Dipturus melanospilus	6	A	0.0669	0.1282	0.0260	0.1024	0	0
Figaro boardmani	30	A	0.0669	0.1282	0.0260	0.1024	0	0.824[3]
Glaucostegus typus	16	R	0.1046	0.0713	0.0150	0.0372	0	0
Gymnura australis	17	M	0.0114	0.1551	0.0067	0.0571	0	0.59[4]
Hemigaleus australiensis	114	A	0.0002	0.0026	0.0029	0.0000	0	0.38[4]
Hemitrygon fluvorium	46	M	0.0114	0.1551	0.0067	0.0571	0	0
Heterodontus galeatus	13	A	0.0669	0.1282	0.0260	0.1024	0	0
Hydrolagus ogilby	1	A	0.0669	0.1282	0.0260	0.1024	0	0
Hypnos monopterygius	62	B	0.0670	0.1352	0.0212	0.0996	0	0
Loxodon macrorhinus	37	A	0.0669	0.1282	0.0260	0.1024	0	0
Maculabatis astra	2	M	0.0114	0.1551	0.0067	0.0571	0	0
Maculabatis toshi	2	M	0.0114	0.1551	0.0067	0.0571	0.42[2]	0.47[4]
Mustelus walkeri	25	A	0.0669	0.1282	0.0260	0.1024	0	0
Neotrygon leylandi	85	M	0.0114	0.1551	0.0067	0.0571	0	0.41[4]
Neotrygon picta	4	M	0.0114	0.1551	0.0067	0.0571	0	0
Neotrygon trigonoides	385	M	0.0114	0.1551	0.0067	0.0571	0.23[2]	0
Orectolobus maculatus	7	A	0.0669	0.1282	0.0260	0.1024	0.39[2]	0
Orectolobus ornatus	3	A	0.0669	0.1282	0.0260	0.1024	0	0
Parascyllium collare	28	A	0.0669	0.1282	0.0260	0.1024	0	0
Rhizoprionodon acutus	12	A	0.0669	0.1282	0.0260	0.1024	0	0.18[4]
Rhizoprionodon taylori	12	A	0.0669	0.1282	0.0260	0.1024	0	0
Rhynchobatus australiae	16	R	0.1046	0.0713	0.0150	0.0372	0.39[2]	0.9[4]
Sphyrna lewini	53	A	0.0669	0.1282	0.0260	0.1024	0	0
Squalus grahami	0	A	0.0669	0.1282	0.0260	0.1024	0	0
Squalus megalops	0	A	0.0669	0.1282	0.0260	0.1024	0	0
Squalus montalbani	7	A	0.0669	0.1282	0.0260	0.1024	0	0
Squatina albipunctata	5	A	0.0669	0.1282	0.0260	0.1024	0.39[2]	0
Stegostoma fasciatum	5	A	0.0669	0.1282	0.0260	0.1024	1	0
Trygonoptera testacea	1270	S	0.0152	0.2561	0.0064	0.0000	0	0.2[1]
Trygonorrhina fasciata	3	R	0.1046	0.0713	0.0150	0.0372	0	0
Urolophus bucculentus	3	U	0.0052	0.1031	0.0019	0.0141	0	0
Urolophus flavomosaicus	1	U	0.0052	0.1031	0.0019	0.0141	0	0
Urolophus kapalensis	121	U	0.0052	0.1031	0.0019	0.0141	0	0
Urolophus sufflavus	1	U	0.0052	0.1031	0.0019	0.0141	0	0
Urolophus viridis	1	U	0.0052	0.1031	0.0019	0.0141	0	0

[1]Current study; [2]Zhou and Griffiths (2008); [3]Peter Kyne unpublished data; [4]Stobutzki *et al.* (2002)

Table 14: Area over which each species is distributed south of the Great Barrier Reef Marine Park (A_x), trawled area within each species distribution as a function of sector and the total trawled area within each species distribution (all in km^2).

Species	A_x	Trawled area							
		Banana	Deep Water	Scallop	Shallow Water	Tiger	Danish seine	Fish trawl	Total
Aetomylaeus nichofii	6,882	149	1	945	85	23	0	0	1,203
Aptychotrema rostrata	23,027	149	275	945	3,551	23	842	837	6,621
Asymbolus analis	3,822	0	2,056	0	1,828	0	411	537	4,832
Asymbolus rubiginosus	3,841	0	2,013	0	1,826	0	411	385	4,634
Bathytoshia brevicaudata	5,622	0	3,060	0	2,082	0	478	480	6,101
Bathytoshia lata	9,749	0	10,222	0	2,159	0	515	556	13,452
Brachaelurus colcloughi	23,027	149	275	945	3,551	23	842	837	6,621
Carcharhinus amboinensis	22,328	149	270	945	3425	23	818	837	6,467
Carcharhinus brevipinna	21,515	149	0	945	2,970	23	842	837	5,766
Chiloscyllium punctatum	22,556	149	0	945	3,551	23	842	833	6,343
Dentiraja australis	8,137	0	1,499	0	544	0	412	423	2,878
Dentiraja endeavouri	4,221	0	8,285	0	0	0	0	0	8,285
Dipturus melanospilus	3,019	0	1	0	0	0	0	0	1
Figaro boardmani	3,050	0	3,726	0	0	0	0	0	3,726
Glaucostegus typus	22,556	149	0	945	3,551	23	842	837	6,347
Gymnura australis	27,363	149	11,158	945	3,551	23	842	837	17,505
Hemigaleus australiensis	25,832	149	10,927	945	3,551	23	842	833	17,270
Hemitrygon fluvorium	8,854	149	0	401	295	23	182	213	1,262
Heterodontus galeatus	4,355	0	0	0	2,065	0	470	458	2,993
Hydrolagus ogilby	5,397	0	2,337	0	0	0	0	0	2,337
Hypnos monopterygius	26,579	149	11,135	945	3,551	23	842	837	17,482
Loxodon macrorhinus	22,328	149	270	945	3,425	23	842	824	6,478
Maculabatis astra	24,507	149	5,861	945	3,551	23	842	837	12,208
Maculabatis toshi	5,321	149	0	46	84	23	21	49	372
Mustelus walkeri	13,278	0	10,191	0	965	0	4	26	11,187
Neotrygon leylandi	4,920	0	0	943	0	16	0	0	959
Neotrygon picta	6,534	149	0	945	0	23	0	0	1,117
Neotrygon trigonoides	22,556	149	0	945	3,551	23	842	833	6,343
Orectolobus maculatus	26,094	149	11,038	945	3,551	23	842	837	17,385
Orectolobus ornatus	23,027	149	275	945	3,551	23	842	837	6,621
Parascyllium collare	5,804	0	5,859	0	2,062	0	453	434	8,807
Rhizoprionodon acutus	14,058	149	269	945	2,085	23	218	201	3,889
Rhizoprionodon taylori	16,982	149	328	945	1,393	23	327	282	3,447
Rhynchobatus australiae	17,970	149	0	945	2,357	23	840	815	5,129
Sphyrna lewini	27,912	149	11,158	945	3,551	23	842	837	17,505
Squalus grahami	2,504	0	22	0	0	0	0	0	22
Squalus megalops	20,791	0	11,158	544	3,256	0	660	624	16,243
Squalus montalbani	1,934	0	0	0	0	0	0	0	0
Squatina albipunctata	20,085	0	11,158	544	3,256	0	660	624	16,243
Stegostoma fasciatum	12,882	149	0	945	1,440	23	836	785	4,177
Trygonoptera testacea	16,151	0	0	0	3,466	0	842	833	5,141
Trygonorrhina fasciata	9,571	0	272	0	2,590	0	624	638	4,124
Urolophus bucculentus	2,066	0	968	0	615	0	0	15	1,597
Urolophus flavomosaicus	6,173	0	6,626	0	196	0	3	3	6,828
Urolophus kapalensis	3,230	0	178	0	1,826	0	427	409	2,840
Urolophus sufflavus	2,806	0	966	0	1,414	0	101	102	2,584
Urolophus viridis	2,145	0	966	0	614	0	0	15	1,595

Table 15: Relevant metrics used to assess the ecological risk posed to elasmobranchs and risk categories derived from the current study. Also shown is the precautionary risk categories derived. See the Methods section on page 9 for details.

Species	F_{msm}	$\min(F_{\text{msm}})$	F	F + 90%CI	Risk	Precautionary risk
Aetomylaeus nichofii	0.169	-	0.014	0.019	Low	Low
Aptychotrema rostrata	0.133	0.096	0.004	0.005	Low	Low
Asymbolus analis	0.123	0.089	0.018	0.024	Low	Low
Asymbolus rubiginosus	0.209	0.159	0.065	0.086	Low	Low
Bathytoshia brevicaudata	0.156	-	0.113	0.153	Low	Low
Bathytoshia lata	0.156	-	0.202	0.287	Medium	Prec. High
Brachaelurus colcloughi	0.157	-	0.030	0.040	Low	Low
Carcharhinus amboinensis	0.113	-	0.036	0.046	Low	Low
Carcharhinus brevipinna	0.146	-	0.027	0.036	Low	Low
Chiloscyllium punctatum	0.156	-	0.023	0.030	Low	Low
Dentiraja australis	0.211	0.158	0.038	0.051	Low	Low
Dentiraja endeavouri	0.266	0.160	0.194	0.291	Low	Prec. Medium
Dipturus melanospilus	0.151	0.118	0.000	0.000	Low	Low
Figaro boardmani	0.159	0.127	0.028	0.042	Low	Low
Glaucostegus typus	0.084	0.045	0.018	0.023	Low	Low
Gymnura australis	0.157	-	0.037	0.051	Low	Low
Hemigaleus australiensis	0.156	-	0.001	0.001	Low	Low
Hemitrygon fluvorium	0.062	0.012	0.012	0.015	Low	Prec. Medium
Heterodontus galeatus	0.067	0.032	0.072	0.100	Medium	Prec. Extreme High
Hydrolagus ogilby	0.155	0.118	0.057	0.083	Low	Low
Hypnos monopterygius	0.157	-	0.084	0.115	Low	Low
Loxodon macrorhinus	0.257	0.157	0.031	0.040	Low	Low
Maculabatis astra	0.074	0.027	0.065	0.087	Low	Prec. Extreme High
Maculabatis toshi	0.157	-	0.003	0.004	Low	Low
Mustelus walkeri	0.098	0.023	0.111	0.159	Medium	Prec. Extreme High
Neotrygon leylandi	0.162	-	0.007	0.010	Low	Low
Neotrygon picta	0.081	0.038	0.013	0.018	Low	Low
Neotrygon trigonoides	0.084	0.036	0.001	0.002	Low	Low
Orectolobus maculatus	0.131	0.054	0.050	0.068	Low	Low
Orectolobus ornatus	0.122	-	0.030	0.040	Low	Low
Parascyllium collare	0.157	-	0.186	0.256	Medium	Prec. High
Rhizoprionodon acutus	0.193	0.126	0.026	0.034	Low	Low
Rhizoprionodon taylori	0.320	0.156	0.022	0.028	Low	Low
Rhynchobatus australiae	0.093	0.045	0.001	0.002	Low	Low
Sphyrna lewini	0.121	0.064	0.076	0.104	Low	Prec. Medium
Squalus grahami	0.103	0.064	0.001	0.002	Low	Low
Squalus megalops	0.063	0.019	0.097	0.133	High	Prec. Extreme High
Squalus montalbani	0.052	0.003	0.000	0.000	Low	Low
Squatina albipunctata	0.149	0.109	0.062	0.085	Low	Low
Stegostoma fasciatum	0.156	-	0.006	0.007	Low	Low
Trygonoptera testacea	0.099	0.058	0.046	0.066	Low	Low
Trygonorrhina fasciata	0.117	0.069	0.030	0.039	Low	Low
Urolophus bucculentus	0.077	0.039	0.081	0.109	Medium	Prec. Extreme High
Urolophus flavomosaicus	0.091	0.052	0.117	0.172	Medium	Prec. Extreme High
Urolophus kapalensis	0.089	0.034	0.066	0.094	Low	Prec. High
Urolophus sufflavus	0.107	0.065	0.090	0.122	Low	Prec. Medium
Urolophus viridis	0.106	0.064	0.078	0.107	Low	Prec. Medium

Appendix 7 – Species distributions

Following are the distribution of each species assessed in the ERA. Also shown is the distribution of trawl effort with each distribution.

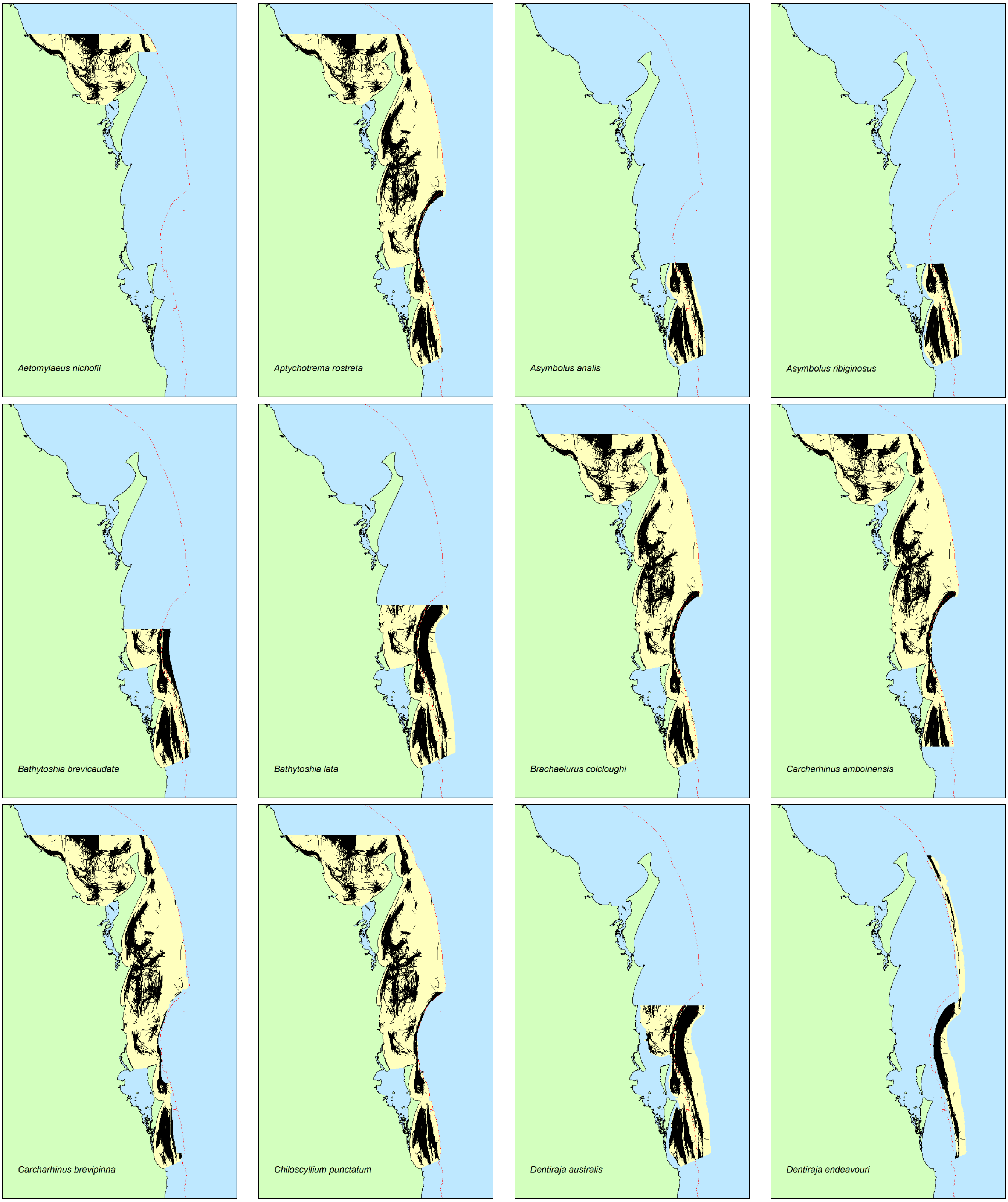


Figure 18: Spatial distribution of 12 of the species assessed in the ERA. Also shown is the distribution of trawl effort within each distribution. The red lines represent the 91m (50 fathom) depth contour.

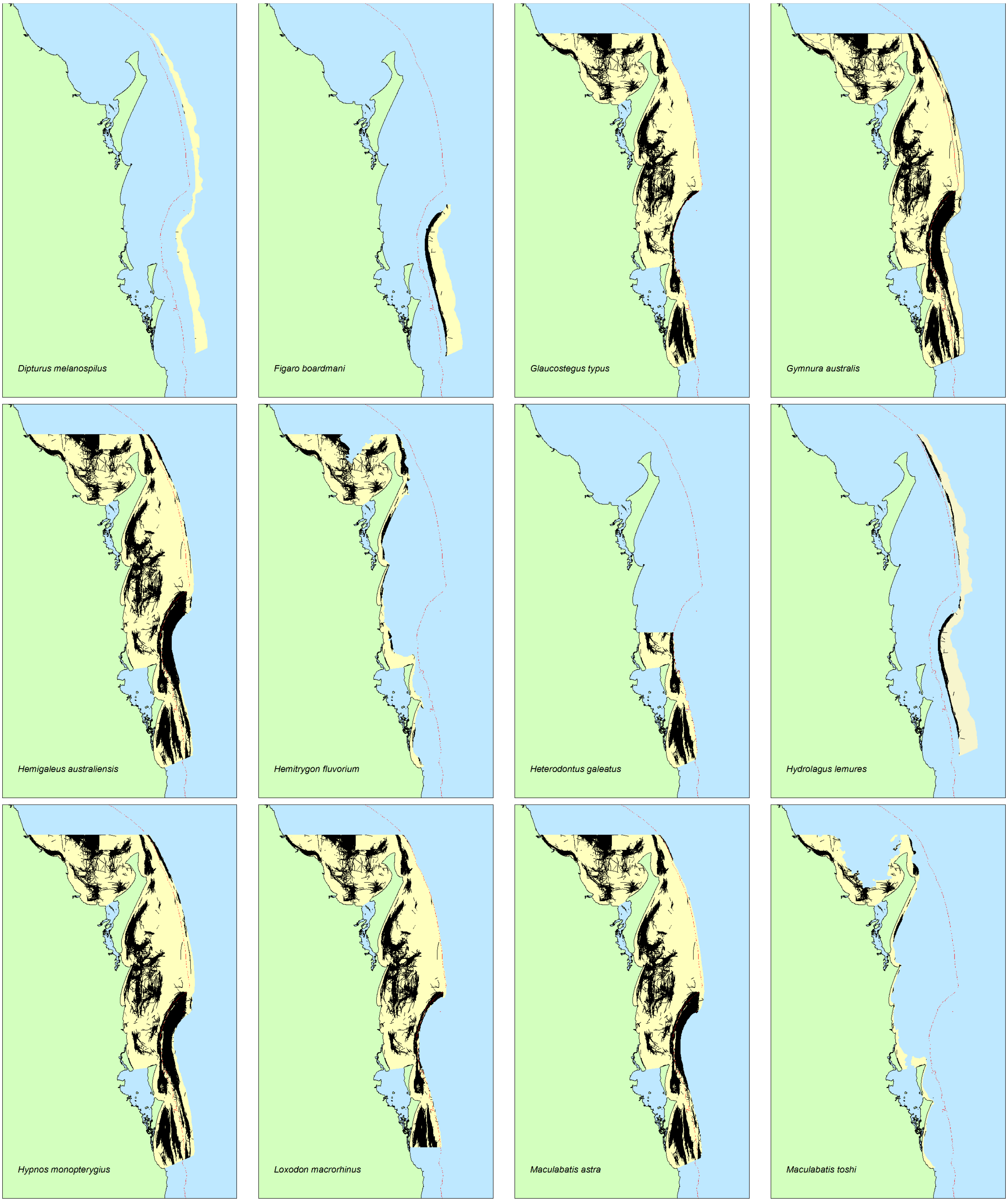


Figure 19: Spatial distribution of 12 of the species assessed in the ERA. Also shown is the distribution of trawl effort within each distribution. The red lines represent the 91m (50 fathom) depth contour.

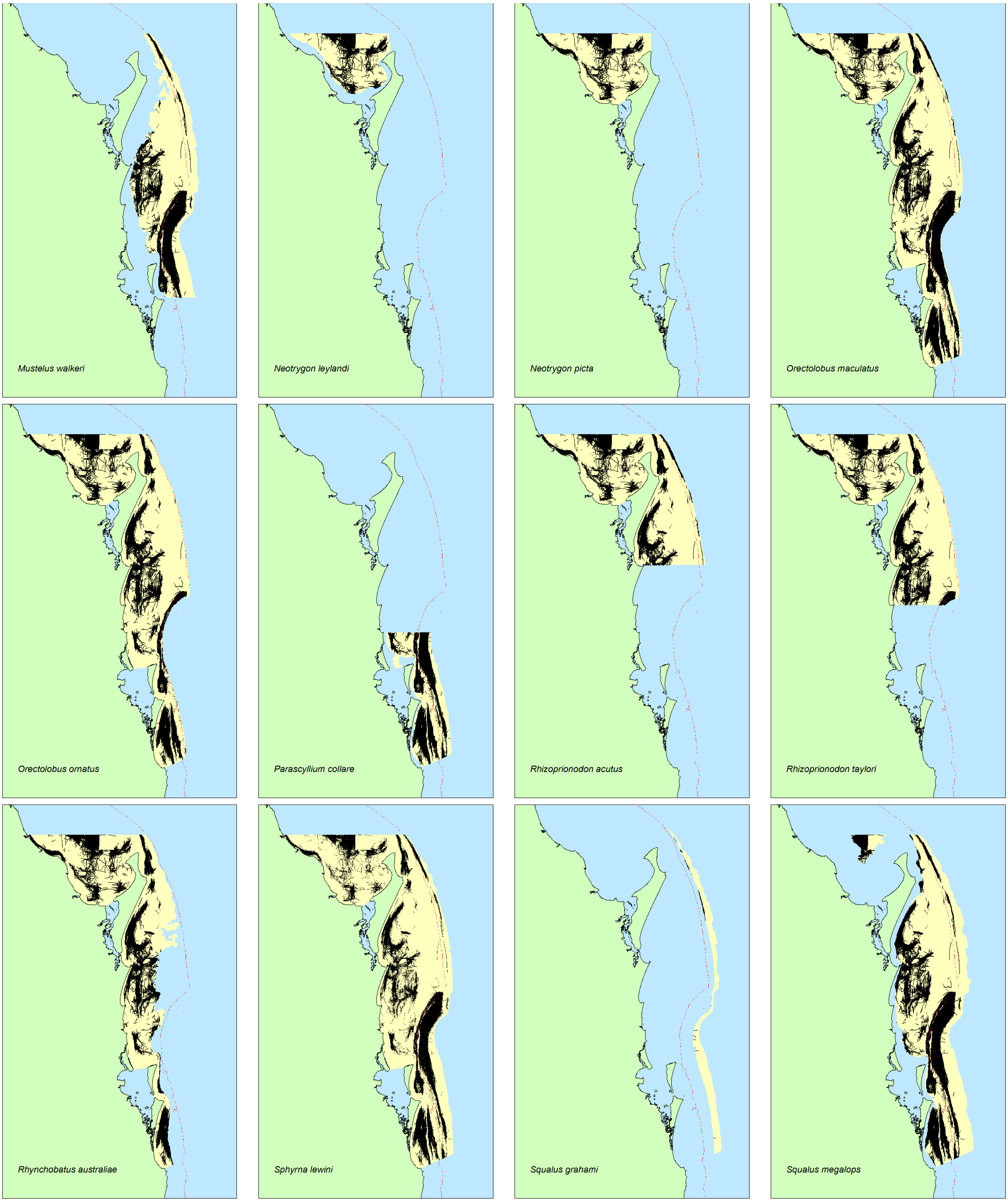


Figure 20: Spatial distribution of 12 of the species assessed in the ERA. Also shown is the distribution of trawl effort within each distribution. The red lines represent the 91m (50 fathom) depth contour.

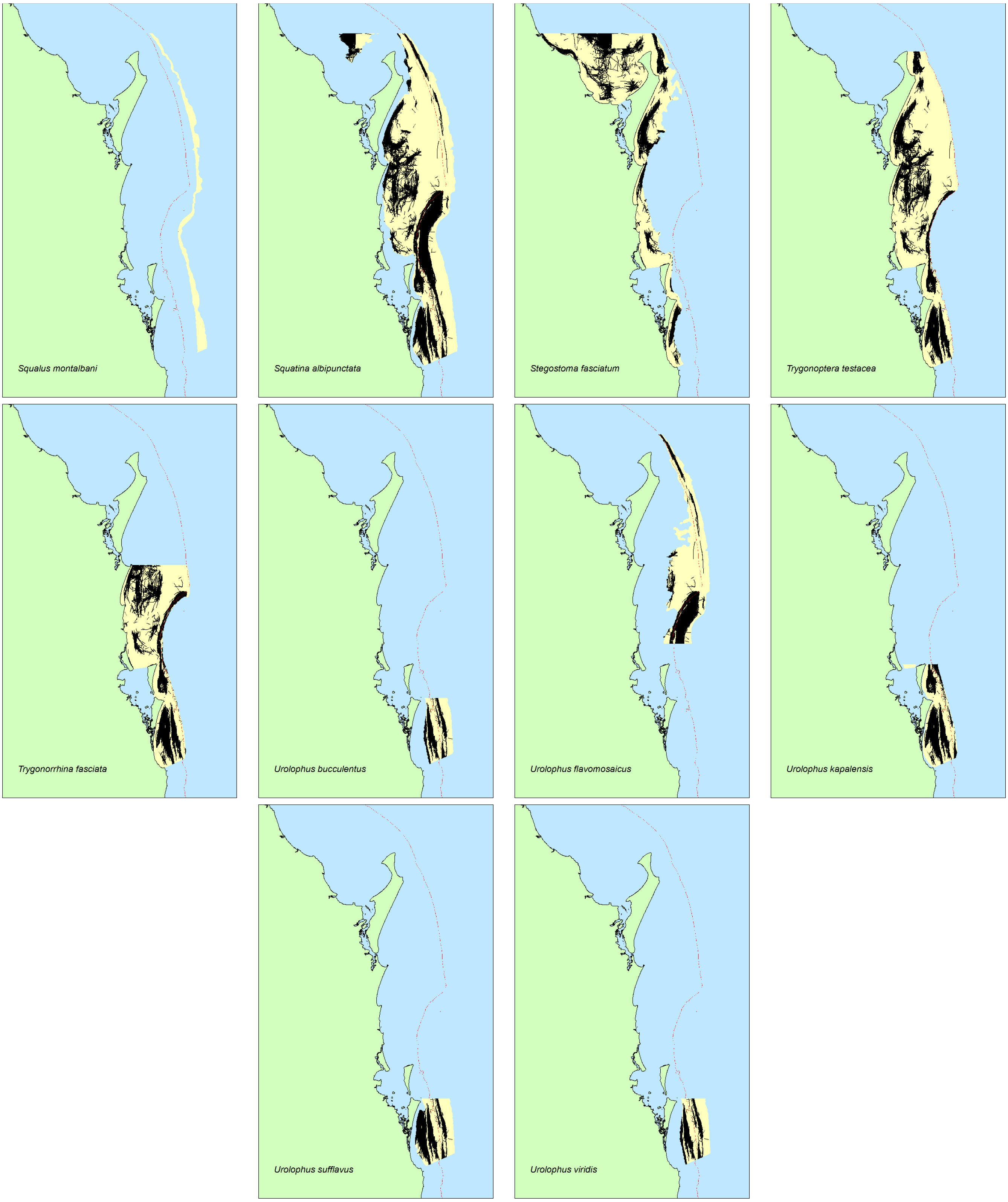


Figure 21: Spatial distribution of ten of the species assessed in the ERA. Also shown is the distribution of trawl effort within each distribution. The red lines represent the 91m (50 fathom) depth contour.

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