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ASSESSING ECOLOGICAL RISK POSED TO COMMON RAYS BY PRAWN TRAWLING

Thesis submitted by

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This thesis is the result of the support provided by my father, John. Dad took on the responsibility of taking care of three kids under seven and this document is testament to his commitment and dedication to raising kids alone. I will be forever grateful to be raised in a place like Urunga, NSW, which instilled a love of the ocean that continues almost five decades later.

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Abstract

Chondrichthyans (i.e., sharks, rays, and chimaeras) are susceptible to over-exploitation due to life history characteristics such as slow growth, late maturity, and low fecundity. Approximately one-third of all chondrichthyans have an elevated risk of extinction due to capture in fisheries targeting other species. Chondrichthyans are caught incidentally in penaeid trawls which are characterised by small mesh sizes and are towed on the seafloor. Previous research has shown that small chondrichthyans are found in the discarded portion of catches from the Queensland east coast otter trawl fishery (QECOTF), Australia's largest penaeid-trawl fishery.

The current study quantified the ecological risk posed to 48 chondrichthyan species discarded by trawlers operating in the QECOTF south of the Great Barrier Reef Marine Park (GBRMP, >24.5°S) in the 2019 fishing year. Risk was assessed using the Sustainability Assessment for Fishing Effects (SAFE) method, a quantitative approach that compares the instantaneous fishing mortality rate (F) of each species to its maximum sustainable fishing mortality (F_{msm}). Fishing mortality is a function of the area trawled within each species' distribution, offset by the survival of discarded animals and escape from trawls via turtle excluder devices (TEDs). The SAFE analysis indicated that the level of fishing effort in the 2019 fishing year posed low risk to the long-term sustainability of all but two of the 48 species assessed. Overfishing of two species, *Squalus megalops* and *Dentiraja australis*, was occurring. Management changes introduced in the early 2000s resulted in significant reductions in nominal fishing effort in the study area. This, combined with the mandated use of TEDs and the prohibition of the retention of chondrichthyan products, has led to a reduction in the fishing mortality of chondrichthyans in the study region and throughout the QECOTF. Fishing impacts can be further mitigated by reducing maximum bar space, the distance between adjacent vertical bars in a single grid hard TED, to increase the escape of chondrichthyans. Additionally, fishers can increase post-release survival of chondrichthyans by reducing trawl duration and returning individuals to the sea as soon as practicable to reduce exposure to air.

The majority of the 48 species assessed in this study lack life history information, which is required to estimate maximum sustainable fishing mortality (F_{msm}). As such, growth was estimated to improve the risk categorisation for *Aptychotrema rostrata*, the most common chondrichthyan found in penaeid-trawl discards in the study area (see Table 16). Growth parameters were derived from vertebral sections and estimated in a Bayesian framework with informative priors, to minimise bias resulting from the under-sampling of older, larger animals. The number of length-at-age observations was increased via back-calculation. A total of 212 length-at-age observations were used to estimate the growth parameters which, after back-calculation, increased to 1112 length-at-age measures. The von Bertalanffy growth function (VBGF) was found to best fit the *A. rostrata* length-at-age data. With the sexes combined, the estimated VBGF parameters were $L_\infty = 923$ mm TL, $L_0 = 193$ mm TL and $k = 0.08$ year^{-1}. Estimates of

L_∞ and L_0 were higher for females (1141 and 193 mm, respectively), compared to males (813 and 187 mm, respectively). The growth coefficient for females ($k = 0.05$ year^{-1}) was half that of males ($k = 0.10$ year^{-1}).

The escape of chondrichthyans from penaeid trawls, via TEDs, was estimated using data collected by observers on-board commercial trawlers operating in Australia's northern prawn fishery (NPF) during 2001 (Chapter 4). Generalised linear mixed modelling was used to quantify factors affecting the escape of Carcharhiniformes, Myliobatiformes, Orectolobiformes and Rhinopristiformes. Fish size, bar space, TED orientation (top- or bottom-shooter), grid size and grid shape were among the factors tested. More than 6,200 individuals were caught during the sampling conducted in the NPF to quantify escape via TEDs. The catch of large elasmobranchs was lower from nets containing TEDs: increasing fish size was found to result in higher escape for all taxonomic orders. Top-shooter TEDs increased the escape of Carcharhiniformes, while bottom-shooter TEDs facilitated greater escape of Myliobatiformes. Grid orientation had no effect on the escape of Orectolobiformes or Rhinopristiformes. Decreasing bar space was found to increase the escape of the Australian blacktip shark (*Carcharhinus tilstoni*). The TEDs facilitated the escape of several species of conservation interest including the scalloped hammerhead (*Sphyrna lewini*) and zebra shark (*Stegostoma fasciatum*). The rostrum of the narrow sawfish (*Anoxypristis cuspidata*), however, inhibited escape of this globally important species.

To improve estimates of the risk posed to the two most common chondrichthyans caught by trawlers operating in the QECOTF, the post-trawl survival (PTS) rate of the common stingaree (*Trygonoptera testacea*) and the eastern shovelnose ray (*A. rostrata*), was quantified experimentally, using on-board tanks to house animals up to three days post-capture. A total of 155 *A. rostrata* and 187 *T. testacea* were assessed for PTS. The experiments revealed that *A. rostrata* were more resilient to trawl catch-and-release than *T. testacea*. For both species, survival was found to increase with size, whereas increasing time on deck resulted in lower survival. Female *T. testacea* were found to be more resilient than males, and increased tow duration resulted in lower survival for *A. rostrata*. The mean (± s.e.m.) PTS for female and male *T. testacea* was 33.5 ± 6.0% and 17.3 ± 5.5% respectively, compared with a mean PTS for *A. rostrata* of 86.8 ± 3.2%.

The risk assessment results were limited by a lack of life history information for 26 of the species assessed and this requires attention to improve future assessments. Region-specific life history metrics are also desirable. Future assessments will benefit from improved estimates of escape via TEDs and the survival of released individuals, particularly for those species for which these metrics are currently lacking.

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1.1 Introduction

Approximately 55% of the catch from penaeid-trawl fisheries is discarded (Gilman *et al.*, 2020; Perez Roda *et al.*, 2019). Penaeid trawls are poorly selective (Broadhurst, 2000) due to the small mesh size used in their construction and the susceptibility to capture of the species with which penaeids cohabit in tropical and sub-tropical demersal environments. Penaeid-trawl discards are comprised of hundreds of species (Courtney *et al.*, 2006; Tonks *et al.*, 2008), including species of conservation interest such as sea turtles (Brewer *et al.*, 1998; Robins–Troeger *et al.*, 1995). Consequently, significant research efforts have focused on quantifying and reducing discards since the 1990s (Kennelly, 2020). Initially, research efforts were directed at reducing the incidental capture of sea turtles (*Chelonia mydas*, *Eretmochelys imbricata*, *Lepidochelys kempii*, *Dermochelys coriacea* and *Caretta caretta*), which were categorised as Endangered in the United States under the Endangered Species Act of 1973. This work resulted in the development of turtle excluder devices (TEDs), primarily in the south-east United States, which are now mandatory in many penaeid-trawl fisheries world-wide. In 1999, the Australian Government listed penaeid trawling as a Key Threatening Process to sea turtles under the *Environment Protection and Biodiversity Conservation Act* (EPBC) *1999* (Eayrs *et al.*, 1997; Gullett, 2003; Robins *et al.*, 1999). This categorisation implied that penaeid trawling adversely affected the survival or abundance of sea turtles in waters north of 28°S, which was addressed by the introduction of legislation requiring the mandatory use of TEDs in Australia's Northern Prawn Fishery (NPF) and Queensland East Coast Otter Trawl fishery (QECOTF).

The introduction of TEDs in penaeid-trawl fisheries has also been found to reduce the catch of some chondrichthyans (sharks, rays, and chimaeras) found in penaeid-trawl discards (Brewer *et al.*, 2006; Noell *et al.*, 2018; Willems *et al.*, 2016). Chondrichthyans have been the subject of increased concern in the last two decades due to an elevated risk of extinction (Dulvy *et al.*, 2014), primarily as a result of capture in fisheries targeting other species (Dulvy *et al.*, 2021). Chondrichthyan life history strategies include late maturity, low fecundity, long life spans and slow growth (Dulvy *et al.*, 2008; James *et al.*, 2015), which make this species group vulnerable to over-exploitation (Ellis *et al.*, 2008). Research has shown that chondrichthyans caught by prawn trawls are predominantly batoids and small demersal sharks (Courtney *et al.*, 2006; Ellis *et al.*, 2017; Robins and McGilvray, 1999; Shepherd and Myers, 2005; Stobutzki *et al.*, 2002). Importantly, the batoids caught by penaeid trawls include highly threatened species such as sawfish (Brewer *et al.*, 2006; Brewer *et al.*, 1998), guitarfish (Garcıa–Caudillo *et al.*, 2000), wedgefish (Brewer *et al.*, 2006; Fennessy, 1994; Robins and McGilvray, 1999) and skates (Kyne *et al.*, 2002; Rigby *et al.*, 2016c).

Chondrichthyans are found in the discarded portion of catches in the QECOTF, a multi-sector trawl fishery which operates between Cape York (10°42'S, 142°03'E) and the Queensland/New South Wales border (28°10'S, 153°33'E). Trawl fishers target penaeid prawns (*Melicertus* spp., *Penaeus* spp., *Metapenaeus* spp.), squid (Teuthoidea), Moreton Bay bugs (*Thenus parindicus* and *T. orientalis*), saucer scallops (Pectinidae: *Ylistrum balloti*) and stout whiting (*Sillago robusta*). Fishers can also retain limited amounts of permitted (byproduct) species including portunid crabs (*Portunus armatus* and *P. sanguinolentus*), Balmain bugs (*Ibacus* spp), and cuttlefish (*Sepia* spp.). Logbook data indicate that 287 vessels fished 32,955 days, making it Australia's largest penaeid-trawl fishery, and reported approximately 6,667 tonnes of product sold at both domestic and international markets in 2020. It has been estimated that >25,000 t of catch is discarded annually by the QECOTF (Wang *et al.*, 2020), which equates to a discard rate of ~77%, representing ~28.5% of discards from all Australian commercial fisheries combined (Kennelly, 2020).

Discards generated by the QECOTF have been widely researched in Queensland (e.g. Courtney *et al.*, 2008; Courtney *et al.*, 2014; Courtney *et al.*, 2006; Wang *et al.*, 2020). This research has demonstrated that chondrichthyans are caught incidentally throughout the fishery (Kyne *et al.*, 2002). However, the possession of shark products by fishers operating in the QECOTF was prohibited in 2001 and, since then, fishers have been required to return all chondrichthyans to the sea as quickly as practicable. As such, species-specific estimates of total catch are lacking, which hinders the assessment of population status of the chondrichthyans impacted by the QECOTF. Consequently, ecological risk assessment (ERA) is the most common method of assessing the population status of data-poor catch components, particularly the chondrichthyans, in this and other trawl fisheries (e.g. Baje *et al.*, 2021; Braccini *et al.*, 2006; Stobutzki *et al.*, 2002). Qualitative ERAs compare the likelihood of capture for each species to its resilience to fishing impacts (Astles *et al.*, 2009; Baje *et al.*, 2021; Hobday *et al.*, 2011). Two qualitative ERAs have been used to assess the population status of chondrichthyans caught by penaeid trawls in Queensland. Pears *et al.* (2012) assessed the risk posed to a number of species groups and habitats by the QECOTF within the Great Barrier Reef Marine Park (GBRMP, see Figure 9). These authors reported that the fishery posed high risk to the sustainability of 11 chondrichthyan species based on high levels of interaction with the fishery, poor or very poor post-release survival, and ineffective TEDs. Jacobsen *et al.* (2018) found that QECOTF posed a high risk to 15 chondrichthyans species in the area south of the GBRMP for similar reasons.

An alternative to the qualitative approaches to assess risk is the Sustainability Assessment for Fishing Effects (SAFE), which was developed to provide a quantitative measure of fishing impacts (Zhou *et al.*, 2015; Zhou and Griffiths, 2008; Zhou *et al.*, 2009; Zhou *et al.*, 2011). The SAFE method has been used to assess the risk to species discarded in major fisheries managed by the Australian Commonwealth Government and inshore fisheries in New Zealand. 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. Although SAFE is less data-intensive than a quantitative stock assessment, life history information and relevant metrics, such as post-trawl survival (PTS) and escape via TEDs, are required, in addition to data on the distribution of the species and fishing effort.

The life history information of chondrichthyans interacting with the QECOTF is not well known (Harry *et al.*, 2011; Kyne *et al.*, 2021). Although some studies have been conducted within the bounds of the QECOTF (e.g. Gutteridge *et al.*, 2013; Jacobsen and Bennett, 2010; Pierce and Bennett, 2009), growth and age-at-maturity information is scant in the primary literature for the majority of chondrichthyans in Queensland. For the most part, life history characteristics of chondrichthyans have been quantified by students in the last decade undertaking post-graduate research (e.g. Jacobsen and Bennett, 2011; White *et al.*, 2014) due to a lack of research funding for basic biological research. The lack of life history information represents a significant impediment to assessing the population status of chondrichthyans and, as a result, proxies are often used to estimate a species' resilience to fishing (Kyne *et al.*, 2021). Improvements to, and validation of, life history characteristics have been identified as a method of reducing error when assessing risk (Zhou *et al.*, 2016).

The PTS of chondrichthyans discarded from penaeid-trawl catches is poorly understood (Braccini *et al.*, 2012; Dapp *et al.*, 2016; Oliver *et al.*, 2015; Willems *et al.*, 2016). Ellis *et al.* (2017) reviewed 79 studies detailing the post-release survival of chondrichthyans and reported most studies in the primary literature were conducted in pelagic longline fisheries, while 21 were trawl-related (including beam trawl and scallop dredge). The PTS of chondrichthyans from penaeid trawls were the subject of only two studies (Fennessy, 1994; Stobutzki *et al.*, 2002), which quantified immediate or at-vessel survival which fails to account for any delayed effects known to affect survival (e.g. Kaiser and Spencer, 1995; Van Beek *et al.*, 1990; Wassenberg and Hill, 1993). The paucity of PTS studies in trawl fisheries is due to the cost and logistical constraints of field-based experiments needed to quantify this metric (Benoît *et al.*, 2012; Benoît *et al.*, 2013; Dapp *et al.*, 2016; Musyl *et al.*, 2011). Most of the trawl-based field studies assessing the PTS of chondrichthyans have been conducted in northern hemisphere fish trawls (e.g. Mandelman *et al.*, 2013; Mandelman and Farrington, 2007a; Revill *et al.*, 2005; Rodríguez–Cabello *et al.*, 2005) and have shown that PTS is highly variable, even between species (Ellis *et al.*, 2017).

For the chondrichthyans interacting with the QECOTF, escape rate via TEDs is largely unknown. Although TEDs have been shown to reduce the capture of large chondrichthyans from penaeid-trawl catches (Brewer *et al.*, 2006; Willems *et al.*, 2016), rigorous at-sea testing of TEDs demonstrated that the devices were ineffective for the two most common chondrichthyans caught in the QECOTF, the Eastern Shovelnose Ray (*Aptychotrema rostrata*) and the Common Stingaree (*Trygonoptera testacea*) (Courtney *et al.*, 2008; Courtney *et al.*, 2006). Escape was inestimable for any other chondrichthyan species due to low sample sizes (Courtney *et al.*, 2007). Bar space is the primary driver of escape and is correlated negatively with escape (Belcher and Jennings, 2011; Garstin and Oxenford, 2018; Noell *et al.*, 2018). The current maximum bar space of TEDs used in the QECOTF of 120 mm allows smaller chondrichthyans to pass through the TED and into the codend (Kyne *et al.*, 2002), resulting in capture, air exposure on the vessel deck, increased trauma and, hence, likely elevated fishing mortality.

In Australia, all export fisheries are required to demonstrate that management arrangements are ecologically sustainable for both target and discarded species. Failure to do so can result in the revocation of export permits, prohibiting access to lucrative international markets. Ecological risk assessments have been used to demonstrate the sustainability of chondrichthyan species discarded from trawls, for which catch data are lacking (Baje *et al.*, 2021; Braccini *et al.*, 2006; Pears *et al.*, 2012; Stobutzki *et al.*, 2002).

1.2 Project objectives

This study was developed as part of the research project "*Estimating the impacts of management changes on bycatch reduction and sustainability of high-risk bycatch species in the Queensland East Coast Otter Trawl Fishery*". The research was conducted between July 2015 and January 2018 and was funded by Fisheries Research and Development Corporation (FRDC, Project no. 2015/014), and co-funded by the Commonwealth Scientific and Industrial Research Organisation (CSIRO) and the Queensland Department of Agriculture and Fisheries (QDAF).

The objectives of 2015/014 were: 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 QECOTF 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 QECOTF.

The overarching objective of the current study was to determine the ecological risk, using the SAFE approach, to 48 chondrichthyan species caught in the southern portion of the QECOTF (>24.5°S), in the 2019 fishing year (1 November 2018 to 20 September 2019, Helidoniotis *et al.*, 2020). In this case, ecological risk refers to the risk that a population is subjected to levels of fishing mortality that are unsustainable in the long term. To improve estimates of risk, the following objectives were also developed: 1) Quantify the post-release survival of *A. rostrata* and *T. testacea*, 2) Quantify growth and age-at-maturity of *A. rostrata*, and 3) Quantify the escape of chondrichthyans from penaeid trawls, via TEDs.

1.3 Source of data

Biological samples for the research were obtained as part of FRDC project number 2015/014 (Campbell *et al.*, 2017). This research provided the post-release survival data (Chapter 5), and samples from which growth and age-at-maturity were estimated (Chapter 3). Further samples of *A. rostrata* were obtained from a commercial fisher, Michael Pinzone, Master of the stout whiting Danish seine vessel, FV *San Antone II*. Data collected by Brewer *et al.* (2006) for the Australian Commonwealth northern prawn trawl fishery (NPF) were reanalysed to determine the escape of chondrichthyans from penaeid trawls, via TEDs (Chapter 4). Data from Courtney *et al.* (2007), Dodt (2005) and Rowsell and Davies (2012) were used to calculate fishing mortality in Chapter 6. Life history information was obtained from various studies to determine maximum sustainable fishing mortality (F_{msm}) (Chapter 6) and the reader is directed to Table 15, on page 137, for relevant citations.

1.4 Thesis outline

Chapter 2 is a review of the studies in the primary literature that assess factors affecting the escape of chondrichthyans, via TEDs, and the survival of discarded chondrichthyans caught by trawls.

Chapter 3 provides estimates of growth and age-at-maturity of *A. rostrata* using vertebrae extracted from individuals collected in southern Queensland. Growth parameters are estimated in a Bayesian framework and back-calculation methods are used to increase the number of length-at-age observations. This chapter provides the first estimates of these metrics, which are used to inform the species' resilience to penaeid trawling assessed in Chapter 6.

Chapter 4 estimates the escape of chondrichthyans from penaeid trawls, via TEDs. Data collected in the Northern Prawn Fishery (NPF) are re-analysed to determine the factors affecting escape. Models are developed to estimate the escape of Carcharhiniformes, Myliobatiformes, Orectolobiformes and Rhinopristiformes, the outputs from which are used to estimate fishing mortality in the ecological risk assessment in Chapter 6.

Chapter 5 investigates the post-trawl survival of *A. rostrata* and *T. testacea*, estimated via field studies using on-board tanks. These are the first short-term (i.e., 3 days) post-trawl survival estimates of chondrichthyans discarded from penaeid trawls. The factors affecting post-trawl survival are discussed. The outputs of this chapter are used to calculate fishing mortality for these species in Chapter 6.

Chapter 6 assesses the risk posed to 48 chondrichthyans by the QECOTF in Queensland, south of the Great Barrier Reef Marine Park. The approach known as Sustainability Assessment for Fishing Effects (SAFE) is used to quantify risk. Deficiencies in data inputs are discussed, as are factors likely to affect the sustainability of chondrichthyans in future. Further, the effects of management changes implemented since 2000 on the risk posed by the QECOTF are examined.

Chapter 7 concludes the thesis by discussing the implications of the research. Remaining knowledge gaps and future research priorities are identified.

Chapters 3–5 have been published in the scientific literature, and Chapter 6 has been submitted for publication. These chapters have been included verbatim herein. Digital Object Identifiers have been included at the start of the appropriate chapters. The list of published journal articles are as follows:

Chapter 3: Campbell, M.J., McLennan, M.F., Courtney, A.J., and Simpfendorfer, C.A. (2021) Life-history characteristics of the eastern shovelnose ray, *Aptychotrema rostrata* (Shaw, 1794), from southern Queensland, Australia. Marine and Freshwater Research 72(9), 1280–1289. https://doi.org/10.1071/MF20347;

Chapter 4: Campbell, M.J., Tonks, M.L., Miller, M., Brewer, D.T., Courtney, A.J., and Simpfendorfer, C.A. (2020) Factors affecting elasmobranch escape from turtle excluder devices (TEDs) in a tropical penaeid-trawl fishery. Fisheries Research 224, 105456. https://doi.org/10.1016/j.fishres.2019.105456;

Chapter 5: Campbell, M.J., McLennan, M.F., Courtney, A.J., and Simpfendorfer, C.A. (2018) Post-release survival of two elasmobranchs, the eastern shovelnose ray (*Aptychotrema rostrata*) and the common stingaree (*Trygonoptera testacea*), discarded from a prawn trawl fishery in southern Queensland, Australia. Marine and Freshwater Research 69(4), 551–561. https://doi.org/10.1071/MF17161

2. A review of factors affecting the escape, via turtle excluder devices, and post-trawl survival of chondrichthyans caught in otter trawls

Plate 2: *Asymbolus rubiginosus* on the sorting tray of a penaeid trawler in southern Queensland. Photo by Peter Kyne.

2.1 Introduction

Tropical penaeid (shrimp or prawn) trawling is recognised as a poorly selective form of fishing (Griffiths *et al.*, 2006) and accounts for 27.3% of the world's fisheries discards (Kelleher, 2005). Penaeids cohabitate with a range of species, many of which are susceptible to capture by penaeid trawls (Andrew and Pepperell, 1992). As such, the discarded portion of penaeid-trawl catches is comprised of hundreds of species (e.g. Kennelly *et al.*, 1998; Stobutzki *et al.*, 2001b; Tonks *et al.*, 2008) and includes species of conservation interest such as sea turtles (Brewer *et al.*, 1998; Robins–Troeger *et al.*, 1995; Wallace *et al.*, 2010; Watson and Seidel, 1980). Concerns regarding the effects of discarding on ecosystems are recognised globally (Broadhurst, 2000; James *et al.*, 2015) and date back at least several decades (Andrew and Pepperell, 1992). Consequently, quantifying and mitigation of discards has been the subject of significant research efforts since the early 1990s (Broadhurst *et al.*, 2006; Kelleher, 2005).

Throughout the 1990s, a variety of strategies were developed to reduce discards. Potential management strategies formulated include: time and area closures, marine parks and sanctuaries, prohibiting the retention of selected species, and education (Alverson, 1999; Broadhurst, 2000; FAO, 2016). However, bycatch reduction devices (BRDs) and turtle excluder devices (TEDs) have been the most-researched method of reducing discards. Although the introduction of these devices sparked considerable controversy among fishers (Robins *et al.*, 1999; Tucker *et al.*, 1997), alternative measures such as spatial and/or temporal closures were less desirable given the resulting economic losses (Broadhurst, 2000).

Chondrichthyans (i.e., sharks, rays and chimaeras) are one component of penaeid-trawl discards that have received increasing attention in the last two decades (Dulvy *et al.*, 2017). Chondrichthyan life histories include late maturity, few offspring, long life spans and slow growth (Dulvy *et al.*, 2008; James *et al.*, 2015) making them vulnerable to overexploitation (Brander, 1981; Ellis *et al.*, 2008). In fact, it has been estimated that one-third of chondrichthyans are threatened with an elevated risk of extinction due to capture in fisheries targeting other species (Dulvy *et al.*, 2021). Research has shown that chondrichthyans caught by penaeid trawls are predominantly batoids and small demersal sharks (Courtney *et al.*, 2006; Ellis *et al.*, 2017; Robins and McGilvray, 1999; Shepherd and Myers, 2005; Stobutzki *et al.*, 2002), although larger pelagic sharks (e.g. Carcharhinids) are caught by penaeid and fish trawls that are large and/or fast moving (e.g. Brewer *et al.*, 2006; Jaiteh *et al.*, 2014; Raborn *et al.*, 2012; Wakefield *et al.*, 2016; Zeeberg *et al.*, 2006). Importantly, the batoids caught by penaeid trawls include highly threatened species such as sawfishes (Brewer *et al.*, 2006; Brewer *et al.*, 1998), guitarfish (Garcıa–Caudillo *et al.*, 2000), wedgefish (Brewer *et al.*, 2006; Fennessy, 1994; Robins and McGilvray, 1999) and skates (Kyne *et al.*, 2002; Rigby *et al.*, 2016c).

Penaeid trawling has been responsible for the decline in some coastal chondrichthyan populations (Graham *et al.*, 2001; Shepherd and Myers, 2005). However, research has shown that efforts to remove some of these species from penaeid trawls by TEDs and BRDs have been ineffective (e.g. Brewer *et* *al.*, 2006; Courtney *et al.*, 2008; Shepherd and Myers, 2005; Willems *et al.*, 2016). For example, Griffiths *et al.* (2006) reported that TEDs had no effect on the catch rates of 44 of the 56 chondrichthyans caught by penaeid trawlers operating in Australia's NPF. Given these results, the survival of chondrichthyans discarded after capture is an important metric when assessing the ecological risk posed to chondrichthyans by penaeid trawling (Griffiths *et al.*, 2006; Stobutzki *et al.*, 2002; Zhou and Griffiths, 2008; Zhou *et al.*, 2009). Despite this, the post-release survival of penaeid trawl-caught chondrichthyans is poorly understood (James *et al.*, 2015; Oliver *et al.*, 2015; Willems *et al.*, 2016): only two studies in the primary literature detail the post-release survival of chondrichthyans from penaeid trawls (Fennessy, 1994; Stobutzki *et al.*, 2002).

The absence of metrics such as post-release survival and escape from TEDs and BRDs in the scientific literature commonly leads to an over-estimate of fishing mortality. This results in the assessment of elevated ecological risk posed to chondrichthyans by penaeid trawling. For example, Pears *et al.* (2012) assumed that no *A. rostrata* survived trawl capture in a qualitative ecological risk assessment of the penaeid-trawl fishery within the boundaries of Australia's Great Barrier Reef Marine Park and, consequently, found that penaeid trawling posed high ecological risk to this species. However, subsequent research has shown that the post-trawl survival of *A. rostrata* is high, highlighting the need for post-trawl survival research. This is particularly the case where penaeid trawling has been shown to pose high risk to a particular species.

The objective of this chapter is to review the primary literature to determine the factors that affect two important metrics used to assess the ecological risk posed to chondrichthyans by penaeid trawls: escape via TEDs and post-trawl survival. The review has been limited to articles in the primary scientific literature, primarily using the SCOPUS abstract and citation database to search for keywords, and their combinations, such as: shark, ray, chondrichthyan, elasmobranch, shrimp, prawn, TED, BRD, post-release survival and post-release mortality. Care has been taken to modernise species names.

2.2 TEDs in penaeid-trawl fisheries: a short history

Although single grid hard TEDs have been used to exclude teleosts (e.g. Broadhurst and Kennelly, 1996; Isaksen *et al.*, 1992; Kennelly, 1999) and jellyfish (Broadhurst *et al.*, 1997; Robins *et al.*, 1999) in fisheries around the world, the uptake of these devices has primarily been driven by the need to reduce the capture of sea turtles. The United States classified six species of sea turtle as Endangered under the Endangered Species Act of 1973. In the late 1970s, researchers began experimenting with barriers in trawl nets in an attempt to exclude turtles (Broadhurst, 2000; Jenkins, 2012). Initially, these devices consisted of panels of mesh installed at the mouth of the net, prohibiting turtles from entering the gear. However, the panels were inefficient, clogging with debris, resulting in penaeid loss and other issues (Broadhurst, 2000; Seidel, 1979; Watson *et al.*, 1985; Watson and Seidel, 1980).

Experimentation in the exclusion of turtles using hard grids began in 1980 (Watson *et al.*, 1985). The National Marine Fisheries Service (NMFS), a part of the National Oceanic and Atmospheric Administration (NOAA), developed the NMFS TED, a large (91 cm x 107 cm x 76 cm) frame constructed from ~9.5mm (3/8 inch) galvanised pipe or fibreglass rod (Watson *et al.*, 1985). A grid, made up of several deflector bars 76mm–152mm (3–6 inches) apart, inclined at 45° directed turtles upward and out a hinged trap door (see Watson *et al.* 1985 for an illustration). The NMFS TED also incorporated design modifications that allowed finfish to exit the trawl. Initial research showed that the NMFS TED reduced bycatch by 51% without any significant loss of target penaeids (Watson *et al.*, 1985; Watson and Seidel, 1980). At the same time, the NMFS TED reduced turtle catch rates from 4.6 to 0.5 turtles per hour (Watson and Seidel, 1980).

Over the subsequent decade, NMFS developed several iterations of their TED, including smaller devices for use in inshore areas. Concurrently, penaeid-trawl fishers developed their own devices which were a mixture of hard (grids installed just forward of the codend) and soft (panels of mesh installed in the body of the net) TEDs (Kendall, 1990). At this time, NMFS began testing the effectiveness of industry-developed TEDs to ensure the devices excluded turtles within a given time period determined by SCUBA divers recording the time it took turtles to escape: 2 minutes initially, increasing to a 5 minute limit after the first year of testing (Jenkins, 2012). Throughout the 1990s, NMFS continued to develop, and encouraged industry-development of, TEDs for use in the south-east United States penaeid-trawl fisheries, releasing technical documents describing improvements made by gear researchers, fishers and netmakers (Mitchell *et al.*, 1995).

The 1990s saw TEDs develop worldwide, driven primarily by the decision of the United States to prohibit the importation of penaeids and penaeid products from jurisdictions that failed to adequately protect sea turtles (Epperly, 2003; Gullett, 2003). This prompted many countries to implement the use of suitable TEDs in tropical penaeid-trawl fisheries where turtle interaction was likely. Although issues arose when the United States began introducing TED legislation in the 1980s, Tucker et al. (1997) reported that the introduction of effective TED technology in other countries in the mid- to late-1990s was less controversial. While this may have been the case for fishery managers and the community, fishers were still reluctant to use the devices prior to mandatory legislation. For example, Robins et al. (1999) state that the use of TEDs in Queensland, Australia, was limited to only a few individuals prior to legislation requiring the use of the devices. Increasing pressure from conservation groups and governments provided further impetus for TED uptake worldwide. For example, the Australian Government listed penaeid trawling as a Key Threatening Process to sea turtles under the Environment Protection and Biodiversity Conservation Act (EPBC) 1999 (Eayrs *et al.*, 1997; Gullett, 2003; Robins *et al.*, 1999). This categorisation implied that penaeid trawling adversely affected the survival or abundance of sea turtles in waters north of 28°S. Such a categorisation required threat abatement in the form of the mandatory use of TEDs in Australia's two largest penaeid fisheries: the Northern Prawn Fishery (NPF) and the Queensland East Coast Otter Trawl fishery (ECOTF) (Hall and Mainprize, 2005).

The United States specifically required TEDs that quickly removed turtles from penaeid trawls (Gullett, 2003). Prescriptive regulations regarding TED dimensions, such as escape hole size, bar spacing and escape flap dimensions, ensured the prompt exclusion of turtles. State Department and National Marine Fisheries Service personnel travelled to countries exporting to the United States to inspect TEDs used. A high incidence of non-compliance resulted in the removal of export accreditation. Although strict compliance led to the uptake of effective TEDs, local issues necessitated modifications by industry to minimise penaeid loss. That is, TEDs that excluded turtles in south-eastern United States fisheries may not necessarily be effective in other areas. For example, the Wicks TED, developed by a Queensland netmaker, used a dual-frame system to alter bar space depending on the bycatch encountered in Moreton Bay, Queensland (Robins *et al.*, 1999). A grid with a bar space of 64 mm is permanently attached to the net extension, excluding turtles. A second grid with offset deflector bars can then be cable-tied to the first to reduce the bar space to approximately 30 mm to exclude large quantities of jellyfish (*Catostylus mosaicus*).

2.3 Escape of chondrichthyans, via TEDs, from penaeid trawls

The uptake of TEDs in tropical penaeid-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 chondrichthyans captured by tropical penaeid-trawl fisheries (e.g. Brewer *et al.*, 2006; Robins–Troeger *et al.*, 1995; Willems *et al.*, 2016). However, there are very few studies detailing the effects of TEDs and other BRDs in the primary literature where chondrichthyans 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 chondrichthyans based on relatively low sample sizes with resultant uncertainty and unreliability of results (e.g. Courtney *et al.*, 2006; Jordan *et al.*, 2013; Queirolo *et al.*, 2011). Further, species differentiation is often absent, with individuals grouped to genus, family or order (Oliver *et al.*, 2015).

The dearth of information on the effects of TEDs and other BRDs was especially evident during the 1990s. As TEDs and other types of BRDs became mandatory in penaeid-trawl fisheries, their effects on target catch and discards were a focus of research, particularly in Australia (Kennelly, 2020). 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 other BRDs on penaeid and discard catch rates (e.g. Broadhurst *et al.*, 1996; Broadhurst *et al.*, 1997; Isaksen *et al.*, 1992; Kendall, 1990; Kennelly *et al.*, 1998; Rulifson *et al.*, 1992), while others also confirmed the exclusion of turtles (Brewer *et al.*, 1998; McGilvray *et al.*, 1999; Robins–Troeger, 1994; Robins–Troeger *et al.*, 1995). Most studies during the 1990s were conducted on trawl grounds in an effort to replicate commercial conditions (Broadhurst *et al.*, 1997; Robins–Troeger, 1994; Robins and McGilvray, 1999). This resulted in sufficient quantities of both target species and total discards to enable robust analyses from a relatively small number of trawls, especially where paired comparisons were used.

Given that interactions with chondrichthyans in penaeid trawls are relatively rare (Fennessy, 1994; Fennessy and Isaksen, 2007; Kyne *et al.*, 2002; Wakefield *et al.*, 2016), analyses regarding the effect of TEDs and BRDs on these species were largely absent. Generally, in these studies, most chondrichthyans were grouped in with the discarded portion of the bycatch, with the authors noting a relatively small number of large individuals (e.g. Brewer *et al.*, 1998; Kendall, 1990; McGilvray *et al.*, 1999; Robins–Troeger *et al.*, 1995; Robins and McGilvray, 1999). For example, Robins–Troeger *et al.* (1995) conducted 93 paired tows in southern Queensland using the AusTED, a flexible, plastic-coated wire grid and reported significant reductions in discards while the catch rates of targeted penaeids (*Melicertus plebejus*, *Metapenaeus endeavouri*, *M. ensis*, *Penaeus esculentus*) and sand crabs (*Portunus armatus*) were unaffected. The device also significantly reduced the catch rates of green turtles (*Chelonia mydas*) and large stingrays: however, only one species of ray, *Neotrygon* spp., was reported and no information was provided on the number or size of the animals retained by the control net.

Chondrichthyan bycatch from penaeid trawls received increasing attention in the early 2000s (FAO, 1999; FAO, 2000; Garcıa–Caudillo *et al.*, 2000; Stobutzki *et al.*, 2002). As such, studies on penaeid-trawl bycatch published in the primary literature after this time have highlighted the effects of TEDs and other BRDs on the catch rates of chondrichthyans where appropriate. Researchers have used progressively more complex methods to analyse the effects of TEDs and other BRDs on catch rates. During the 1990s, paired comparisons were analysed using T-tests (e.g. Broadhurst and Kennelly, 1995; Broadhurst and Kennelly, 1996; Kendall, 1990) and generalised linear models (GLMs) in the early 2000s (e.g. Courtney *et al.*, 2006). Since this time, researchers have incorporated random effects in the analyses of the effects of TEDs and BRDs via mixed models (e.g. Gorman and Dixon, 2015; Lomeli and Wakefield, 2013; Millar *et al.*, 2004; Wakefield *et al.*, 2016; Willems *et al.*, 2016).

For the most part, penaeid-trawl operations employ multiple nets. This facilitates the testing of multiple codend treatments concurrently. Although TEDs and BRDs can be tested in one net using alternate tows, mesh covers over escape areas (e.g. Eayrs *et al.*, 2007), trouser trawls (Broadhurst and Kennelly, 1995) or via video analysis (Jaiteh *et al.*, 2014; Wakefield *et al.*, 2016), most studies report the results from two separate nets towed simultaneously (e.g. Brewer *et al.*, 2006; Brewer *et al.*, 1998; Broadhurst and Kennelly, 1996; Courtney *et al.*, 2014; Courtney *et al.*, 2006; Gorman and Dixon, 2015; Heales *et* *al.*, 2008; Kendall, 1990; McGilvray *et al.*, 1999; Robins–Troeger, 1994; Robins and McGilvray, 1999; Robins *et al.*, 1999). Courtney *et al.* (2008) were able to test four treatments simultaneously on a quad-rigged vessel operating in the Queensland saucer scallop (*Ylistrum balloti*) fishery. Using these more complicated methods, researchers are able to standardise for a range of spatial and temporal effects in order to isolate the effect of the devices tested.

The lack of studies detailing chondrichthyan bycatch in penaeid trawls was highlighted in a review by Molina and Cooke (2012). These authors reported that, of 103 studies published on shark bycatch, only eight (7.77%) were related to penaeid trawling, with most focusing on the effects of pelagic fish longlining. Given that capture in fisheries targeting other species is the primary cause of overfishing for chondrichthyans (Dulvy *et al.*, 2021), the lack of research in this area represents a significant gap in scientific knowledge. Although this review is focused on the effects of TEDs in penaeid trawls, following is a review of factors that have been found to affect the exclusion of chondrichthyans from TEDs from all trawl gears.

2.3.1 Factors affecting the exclusion of chondrichthyans by TEDS

The mechanical separation of a chondrichthyan encountering a TED is largely a function of the animal's size and shape. The likelihood of passing through a TED is decreased if an animal is larger in any dimension than the TED's bar spacing. However, despite the obvious importance of fish size on escape via TEDs, this factor is rarely discussed in the primary literature. For the most part, when quantifying discards at sea, researchers make an arbitrary decision as to what is a 'large' individual (i.e., where the individual is larger in all dimensions than the bar space) and report the exclusion rates of individuals of this size only (e.g. Courtney *et al.*, 2008; Gorman and Dixon, 2015). Very few studies report the effects of TEDs on 'small' chondrichthyans (e.g. Brewer *et al.*, 2006; Fennessy and Isaksen, 2007; Willems *et al.*, 2016). Further, the definition of a 'large' chondrichthyan is inconsistent among researchers with the following measures used in the primary literature: >1 m (Brewer *et al.*, 2006; Gorman and Dixon, 2015); >5 kg (Brewer *et al.*, 1998); >10 L (Courtney *et al.*, 2008); >50 cm (Willems *et al.*, 2016); and >70 cm and >30 cm for sharks and rays, respectively (Fennessy and Isaksen, 2007). It is assumed that, provided the TED's escape hole, escape hole cover and grid angle are suitable, the device will exclude a high proportion of these large individuals. A summary of the literature cited is given in Table 8.

2.3.1.1 Size of the chondrichthyan

For any shark or ray encountering a TED, the likelihood that an individual is excluded is determined by its trunk diameter or disc height, respectively, and the bar spacing of the TED. For example, Raborn *et al.* (2012) reported that the Atlantic sharpnose shark (*Rhizoprionodon terraenovae*) never reached a trunk diameter of 100 mm, the maximum bar space allowed in the Gulf of Mexico penaeid-trawl fishery, leaving the authors with the expectation that the introduction of TEDs had no effect on the catch rates of this species. In contrast, blacknose sharks (*Carcharhinus acronotus*) and bonnethead sharks (*Sphyrna* *tiburo*) reach a diameter of 10 cm at a total length of 82 cm and 77 cm, respectively. Given this result, Raborn *et al.* (2012) estimated that the introduction of TEDs reduced the catch rate of *C. acronotus* and *S. tiburo* by 94% and 31%, respectively.

These results are consistent with Brewer *et al.* (2006), who reported relatively poor exclusion of sharks from TEDs in the NPF. This study is one of the few that specifically detail the effects of TEDs and BRDs on chondrichthyans of all sizes. In a meta-analysis of catches from nets fitted with a range of TEDs and/or BRDs, the devices reduced the capture of sharks and rays by 17.7% and 36.3%, respectively, compared to control nets (no TED/BRD fitted). When a subset of larger sharks and rays (>1 m total length or disc width) were analysed, reductions of 86% and 94%, respectively, were achieved and only 4.9% and 25% for animals <1 m. The authors, however, did not discuss the merits of the various devices tested during their study and, as such, it is not possible to make inferences about the factors that influenced the rates of escape reported.

Brewer *et al.* (2006) reported that TEDs excluded only 13.3% of sharks compared to control nets, while TED+BRD combinations excluded 17.7% of sharks. This was due to the poor exclusion of small species, primarily whitecheek (*C. dussumieri*), milk (*Rhizoprionodon acutus*) and weasel (*Hemigaleus australiensis*) sharks. For each of these species , all individuals caught were <1 m (Brewer *et al.*, 2004). The devices tested in the Brewer *et al.* (2006) study were effective on only three species of shark: Australian blacktip sharks (*Carcharhinus tilstoni*), grey carpetsharks (*Chiloscyllium punctatum*) and zebra sharks (*Stegastoma fasciatum*). Of the remaining 14 species of shark caught during the study, none were found to be affected by the TEDs. Brewer *et al.* (2006) reported higher reduction rates for batoids. Overall, nets fitted with a TED+BRD combination excluded 36.3% of rays compared to control nets, while a net fitted with a TED-only reduced the catch of rays by 31.3%. Again, these results are largely due to the devices excluding large individuals: however, TEDs were able to exclude >30% of relatively small ray species including the plain maskray (*Neotrygon annotata*) and the painted maskray (*Neotrygon leylandi*).

2.3.1.2 Bar spacing

The bar spacings of the various TEDs tested by Brewer *et al.* (2006) are likely to have had a significant effect on the exclusion rates of the smaller chondrichthyans. Approximately 80% of vessels operating in the NPF during this study used TEDs with a bar space of 120 mm (Taylor and Day, 2004), the maximum spacing allowed under TED regulations in the NPF. This bar spacing is ~20% wider than the maximum allowed under the United States TED accreditation process. The larger bar spacing used by fishers in the NPF resulted from concerns by fishers regarding the perceived loss of by-product species such as Moreton Bay bugs (*Thenus* spp.) and sand crabs (*Portunus armatus*) with the 102 mm bar space prescribed under the US TED accreditation process. Similar bar space regulations are in place in Queensland, where TEDs have no significant effect on the catch rates of the small chondrichthyans that dominate the catch in the eastern king prawn (EKP: *Melicertus plebejus*) and saucer scallop (*Ylistrum balloti*) fisheries (Courtney *et al.*, 2008; Courtney *et al.*, 2014; Courtney *et al.*, 2006; Kyne *et al.*, 2002). It should be noted, however, that too few individuals were caught to adequately test the effects of TEDs and/or BRDs on the catch rates of the 28 chondrichthyan species caught during these studies.

Noell *et al.* (2018) tested the effects of bar space on the catch of chondrichthyans in the Spencer Gulf penaeid-trawl fishery, targeting the western king prawn (*Melicertus latisulcatus*). These authors compared the catches of chondrichthyans in nets fitted with Nordmøre grids with bar spaces of 35 mm and 45 mm and found no significant difference in either the weight or number of chondrichthyans caught using these grids. The two most common species caught during the study were the Port Jackson shark (*Heterodontus portusjacksoni*) and the ornate wobbegong (*Orectolobus maculatus*), two large species that are unlikely to be impacted by a 10 mm reduction in bar space. In contrast, Garstin and Oxenford (2018) reported a 40% reduction in catches of chondrichthyans, including smooth butterfly rays (*Gymnura micrura*), longnose stingrays (*Hypanus guttatus*) and sharpsnout stingrays (*Fontitrygon geijskesi*), by reducing bar space from 102 to 44.5 mm in the Atlantic seabob (*Xiphopenaeus kroyeri*) fishery in Guyana. These two are the only studies that assess the effects of bar space experimentally in the primary literature. Nalovic (2014) reported that reducing bar space from 100 mm to 50 mm reduced the catch rate of several shark species during at-sea testing in the US Gulf of Mexico penaeid-trawl fishery. Although this study was not published in the primary literature, it is further evidence of increased escape via TEDs when smaller bar spaces are used. Brčić *et al.* (2015) conducted a simulation study that demonstrated reducing bar space from 90 mm to 70 mm significantly reduced the number of blackmouth catsharks (*Galeus melastomus*) caught, whilst maintaining the catch rates of the targeted Norway lobster (*Nephrops norvegicus*).

Provided the TED has prohibited passage of a chondrichthyan into the codend, it is possible to exclude the individual. It is essential that any object large enough to be excluded by a TED be directed to the escape hole and that it exits the trawl as soon as practicable to reduce clogging of the TED and resultant catch loss. Along with bar space, several factors interact to ensure the efficient and prompt exclusion of an animal at a grid including grid angle and grid orientation (top- or bottom-shooter) (Eayrs *et al.*, 1997). Despite the importance of bar spacings, grid angle, grid orientation and their interaction on the exclusion of chondrichthyans, discussion of their interacting effects on chondrichthyan catch rates is largely absent from the primary literature.

2.3.1.3 Grid angle and grid orientation

A grid angle between 45º and 60º is accepted as the most effective for turtle exclusion and catch retention (Eayrs, 2007). At higher angles, TEDs can become clogged with debris (Lucchetti *et al.*, 2016; McGilvray *et al.*, 1999), preventing penaeids from passing through the grid. At low angles, catch loss can result from the escape flap failing to close properly (Eayrs *et al.*, 1997). Generally, grid angles close to 60º can be used on 'clean' grounds, while angles closer to 45º are used in areas where sponges or benthic debris are common. Jaiteh *et al.* (2014) found that a grid installed at 40º excluded a higher proportion of sharks and rays than a grid installed at 70º, but did not elaborate on the effects of grid angle on individual species caught or on the target species. Chosid *et al.* (2012) used two grid angles, 35° and 45°, to exclude spiny dogfish (*Squalus acanthias*) and, despite low sample sizes, catch rates were lowest from the grid installed at 35°.

The interaction between grid angle and grid orientation (i.e., top- or bottom-shooting) has a significant effect on TED performance. For the exclusion of large demersal animals and debris, bottom-shooter TEDs combined with a high grid angle (50–55°) are more effective (Brewer *et al.*, 1998; Taylor and Day, 2004). This takes advantage of an animal's relative mass to aid its exclusion from the trawl. In contrast, top-shooter TEDs are usually installed at lower angles, enabling large objects to move upwards, toward the escape hole. Brewer *et al.* (2006) and Wakefield *et al.* (2016) are the only studies in the primary literature that report the effects of grid orientation on a range of chondrichthyans. Brewer *et al.* (2006) found that top-shooter TEDs reduced the catch of sharks by 20.4% compared to only 8.8% for bottom-shooter TEDs, while Wakefield *et al.* (2016) found top-shooter TEDs excluded 20–30% more benthopelagic sharks than bottom-shooter TEDs. It is difficult, however, to compare results from these two studies as they were conducted in different fisheries.

In the Brewer *et al.* (2006) study, the 20.4% reduction in sharks from top-shooter TEDs was largely due to one Carcharhinid, *C. tilstoni*. The increased exclusion of *C. tilstoni* from top-shooter TEDs was attributed to this species' upward escape response. This supposition is corroborated by Jaiteh *et al.* (2014) who used cameras to observe Carcharhinids attempting to escape a fish trawl by swimming upward. In contrast, Brewer *et al.* (2004) reported that bottom-shooter TEDs excluded a higher proportion of grey carpetsharks (*Chiloscyllium punctatum*) than top-shooter TEDs. Given this species' propensity for benthic camouflage (Dudgeon *et al.*, 2016a), it is reasonable to assume that the escape response of *C. punctatum* is likely to be in a downward direction. In fact, the 4.9% reduction for sharks <1 m in this study is likely the result of the ~25% exclusion of *C. punctatum*. Similarly, the body morphology and preference for benthic habitats of batoids suggests that members of this group are likely to escape in a downward direction.

Although the above postulated behavioural responses to encountering TEDs is a reasonable explanation for these results, the species' position in the trawl likely enhanced escape. Benthic sharks and batoids are likely to position themselves in the lower portion of the trawl net: Main and Sangster (1982) found that skates (Rajidae) and small-spotted catshark (*Scyliorhinus canicula*) were more likely to be caught in the lower level of a fish trawl net divided by a horizontal separator panel. These animals likely encounter the lower half of a TED in a penaeid trawl and, if the TED is a top-shooter, small individuals are likely to pass through the bars of the TED into the codend before they encounter the escape hole. In contrast, if a bottom-shooter TED is used, a high proportion of benthic sharks and batoids are more likely to reach the escape hole quickly and exit the trawl rather than being forced through the bars of TED. About half of the trawl operators in the NPF during the study by Brewer *et al.* (2006) used bottom-shooter TEDs, with a high proportion of these likely to be the U.S.-designed Super–Shooter TED: ~50% of the vessels boarded by observers during the study by Brewer *et al.* (2006) used oval grids (Taylor and Day, 2004). The Super-Shooter is characterised by bent deflector bars at the escape hole which likely enhanced the exclusion of these smaller species.

In addition to bent bars, water flow in the lower half of a net at the TED is at a maximum. Wakeford (2004) analysed the water flow around a Super–Shooter TED in a flume tank and reported that water speed at the bend of the deflector bars in a Super–Shooter is 16% higher than the speed of the trawl. This represents the maximum apparent water flow in the posterior part of a penaeid-trawl net. Increased water flow at this point, combined with the bent bars, would likely aid in the exclusion of benthic sharks and batoids through the lower half of a Super–Shooter TED. As further evidence of the effects of water flow through the lower half of a TED, Wakefield *et al.* (2016) reported that escape was faster through bottom- than top-shooter TEDs for sharks and rays.

These factors are likely responsible for the high exclusion rates of the plain maskray *Neotrygon annotata* in the study by Brewer *et al.* (2006). Exclusion by top- and bottom-shooters was essentially equal for four large batoids: the black-spotted whipray (*Maculabatis toshi*), leopard whipray (*Himantura leoparda*), cowtail stingray (*Pastinachus sephen*) and bottlenose wedgefish (*Rhyncobatus australiae*). Given the relatively large size of these animals, exclusion by TEDs is expected and the grid orientation is unlikely to have influenced exclusion rates if the TEDs were installed correctly (i.e., grid size, grid angle, escape hole dimensions, etc.). However, the bottom-shooter TEDs assessed by Brewer *et al.* (2006) reduced the catch of the much smaller (maximum disc width of 26 cm: Kyne, 2016) *N. annotata* by ~49%, while top-shooters had no effect.

The work of Chosid *et al.* (2012) is the only other published study comparing catch rates from trawls with top- and bottom-shooter TEDs. These authors attempted to assess the effectiveness of top- and bottom-shooter TEDs on the exclusion of spiny dogfish (*Squalus acanthias*) from fish trawls in Massachusetts, USA. However, their results were inconclusive: both grid orientations were shown to reduce the number of dogfish caught but a direct comparison was not possible given the two treatments were not assessed concurrently.

2.3.2 Are BRDs effective?

For the most part, BRDs are installed aft of a TED and are designed to allow small teleosts to escape a penaeid trawl after they have passed through the TED grid into the codend. BRDs rely on fish behaviour and need to be positioned in an area of low water flow to allow fish to orientate towards the escape area and exit the trawl (Broadhurst, 2000). The extended funnel design, also called a radial escape section

(RES), characterised by an internal mesh funnel surrounded by an area of large meshes (Broadhurst, 2000; Eayrs *et al.*, 1997), is one of two BRDs found to affect the catch rates of chondrichthyans. The internal funnel directs catch into the codend, while fish are able to locate the escape area behind the funnel, where water flow is disrupted, and escape through large square or diamond meshes. The RES reduced the catch rates of the shovelnose guitarfish (*Pseudobatos productus*) in trouser trawl experiments conducted in a Mexican penaeid-trawl fishery (Garcıa–Caudillo *et al.*, 2000). The authors installed Super–Shooter TEDs in each leg of the trouser trawl, resulting in the exclusion of all but juvenile *P. productus*. Similarly, Courtney *et al.* (2006) used a TED in conjunction with a modified RES with large diamond meshes around the top half of the net only. These authors found the device was ineffective at excluding chondrichthyans (Kyne *et al.*, 2002) including a confamilial of *P. productus*, the eastern shovelnose ray (*Aptychotrema rostrata*), a common, small (<1.2. TL: Kyne, 2016) batoid. It is difficult to say with certainty that the RES used by Courtney *et al.* (2006) would have reduced the catch rates of *A. rostrata* if the large meshes of the RES extended around the circumference of the net and not just the top half: however, given the likely escape response of batoids, large meshes around the lower half of the trawl would have increased the likelihood of exclusion of *A. rostrata* and other batoids.

Common BRDs such as fisheyes, square mesh panels (SMPs) and composite square mesh panels (CSMPs) are installed aft of a TED in the top of the codend (Eayrs *et al.*, 1997). The area through which animals escape from fisheyes, SMPs and CSMPs is relatively small: mesh sizes in the latter two devices are generally between 50 mm and 100 mm (~2–4 inches) which has been successful at excluding finfish (Broadhurst *et al.*, 1996) but has been shown to be ineffective for chondrichthyans (Brewer *et al.*, 2006; Fennessy and Isaksen, 2007). Fisheyes have been similarly ineffective (Belcher and Jennings, 2011; Brewer *et al.*, 2006; Criales–Hernandez *et al.*, 2006). These devices take advantage of the reduced water flow at the top of the net (Wakeford, 2004), which is necessary for the escape of teleosts (Broadhurst, 2000). The relatively small escape hole size of fisheyes and the relatively small meshes used in the CSMPs, and their position in the net is likely the reason for the poor escape of chondrichthyans via these devices.

In contrast to the fisheye, SMP and CSMP, the square mesh codend (SMC) has an escape area around the circumference of a trawl. Like TEDs, exclusion of chondrichthyans through the meshes of a SMC is a function of the mesh size and the size of the chondrichthyan encountering the device. In penaeid trawls, the mesh size used in the construction of SMCs is generally less than 50 mm (~2 inch) so that catch rates of the target species are maintained. This renders these devices ineffective for chondrichthyans in penaeid trawls (Courtney *et al.*, 2008; Courtney *et al.*, 2014; Ordines *et al.*, 2006). Courtney *et al.* (2008) used an SMC in the Queensland saucer scallop (*Ylistrum balloti*) constructed from ~100 mm (4 inch) mesh. The SMC, in combination with a top-shooter modified Wicks TED, reduced the catch rate of discards by 78%: however, the TED and SMC combination failed to reduce the catch rates of common chondrichthyans, most notably *A. rostrata*. This suggests that the 120 mm bar spacing used in the TED used in the study by Courtney *et al.* (2008) allowed *A. rostrata* to pass through the device and the mesh of the SMC was too small to enable escape.

The Bigeye BRD has been shown to exclude chondrichthyans (Brewer *et al.*, 2006; Robins *et al.*, 1999). The Bigeye was designed by a banana prawn (*Fenneropenaeus merguiensis*) fisher, in Queensland, Australia, and is constructed by making a lateral cut in the panel of a net, before cutting forward from each distal edge of the first cut. The meshes of the forward cut are then sewn down a row of bars forming what is effectively a large fisheye (Robins *et al.*, 1999). The devices can be sewn anywhere in a net and the designer of the Bigeye BRD had upwards of eight devices in each net at any one time. The Queensland banana prawn fishery is a daytime fishery and, consequently, fish bycatch was reduced by between 30% and 40% using the Bigeye (Robins *et al.*, 1999). However, at night or in turbid water, fish bycatch reduction decreased to only 10–15%. Trawl fishers target banana prawns in lower central Queensland during April and migrate to northern ports throughout the austral winter. This facilitated the transfer of this technology from the banana prawn fishers to fishers targeting other species and, by the time that BRDs were made mandatory in the Queensland penaeid-trawl fishery in 2001, most fishers adopted the Bigeye as their primary BRD. Fishers were also convinced that turtles could escape Bigeye BRDs and the devices were legislated as TEDs in 2000. However, the design was tested in the United States by NMFS staff as part of their annual TED testing procedure and failed to facilitate the escape of sea turtles within the testing procedure time limits (Gullett, 2003). The Bigeye BRD was also assessed as part of the NPF's TED testing procedure and failed, with the bigeye catching more large stingrays than an approved TED (Day, 2000).

2.3.3 Other potential gear modifications to reduce chondrichthyan bycatch

Although TEDs have been found to be effective, other methods have been employed to reduce the catch of chondrichthyans in commercial fisheries. Comprehensive discussion of these methods is beyond the scope of the current review: however, some methods warrant mention.

Abrantes *et al.* (2021) assessed the potential of electrical fields in mitigating the incidental catch of sawfish by penaeid trawls. Specifically, Abrantes *et al.* (2021) used tank-based experiments to test the effects of electrical fields on the behaviour of two largetooth sawfish (Pristidae: *Pristis pristis*). The experiments demonstrated that reaction distances were small and would not result in the avoidance of trawls. Electrical pulses are used in shark deterrents (Kempster *et al.*, 2016) by overwhelming an individual's electroreceptors (Huveneers *et al.*, 2018): however, further research is required to determine the effectiveness of similar devices in reducing the incidental catch of chondrichthyans by penaeid trawls.

Lighting has been used to reduce the catch rates of chondrichthyans in gillnet fisheries (Senko *et al.*, 2022). Several studies have assessed the effects of lights on trawl bycatch (Clarke *et al.*, 1986; Gordon *et al.*, 2002; Hannah *et al.*, 2015): however, the effect of lights on the catch of chondrichthyans by penaeid trawls remains unquantified. Similarly, magnets have been used to deter chondrichthyans from entering traps (Richards *et al.*, 2018) and as shark deterrents (Huveneers *et al.*, 2018): however, the large number of magnets required (Rigg *et al.*, 2009) to be effective would likely preclude their use in reducing the catch of chondrichthyans by penaeid trawls.

2.3.4 Summary

In conclusion, acceptable exclusion of chondrichthyans from TEDs is dependent on a number of interacting factors. Firstly, the bar spacing needs to be appropriate for both the chondrichthyans encountered and the target species of the trawl operation. This represents an area of research that requires attention. Secondly, grid angle and grid orientation will be dependent on the predominant species encountered: top-shooter TEDs for sharks and bottom-shooter TEDs for rays. Lastly, grid angle should be altered according to the amount of sponge and large flora and fauna likely to be encountered. These and other factors, such as the size of the grid, the presence/absence of an escape hole cover and the presence/absence of a guiding funnel, are likely to affect the prompt exclusion of chondrichthyans from penaeid trawls. Any legislated changes to TED regulations that improve the escape of chondrichthyans via TEDs should only be implemented only after rigorous testing by researchers and consultation with fishers (Kennelly, 1999; Molina and Cooke, 2012). The benefits of TEDs and BRDs should be emphasised, particularly the improvements in catch quality that can occur when large animals such as chondrichthyans are excluded from the catch (Brčić *et al.*, 2015; Brewer *et al.*, 2006; Chosid *et al.*, 2012; Gorman and Dixon, 2015; Salini *et al.*, 2000). Commercial trawl fishers strive to maximise the retention of target species and any modification to a net that reduces their catch will quickly be rejected (Tucker *et al.*, 1997) unless the potential benefits of the changes are outweighed by the loss of product.

2.4 Post-release survival of chondrichthyans caught by penaeid trawls

Given the relatively poor exclusion rates of chondrichthyans from TEDs and BRDs, it is prudent to assess their survival on release. For the purposes of this review, post-trawl survival (PTS) is the survival of an animal brought to the surface as part of the catch and subsequently discarded (FD in the review by Broadhurst *et al.*, 2006). Although animals escaping trawls may experience injury causing mortalities (Broadhurst *et al.*, 2006; Davis, 2002; Suuronen and Erickson, 2011), this is difficult to quantify and is ignored in the current review.

Discarding occurs for various reasons including minimum legal size or quota restrictions, marketability, value or conservation status (James *et al.*, 2015). Poor survival of discarded catch can have adverse ecosystem and conservation consequences (Kennelly, 1995) and quantifying post-release survival rates is key to understanding these consequences (Braccini *et al.*, 2012; Revill *et al.*, 2005). Despite this, post-release survival is rarely quantified (Davis, 2002), likely due to the cost and logistical constraints of field-based experiments needed to quantify this metric (Benoît *et al.*, 2012; Dapp *et al.*, 2016; Musyl *et al.*, 2011). Estimates of chondrichthyan PTS are under-represented in the primary literature (Dapp *et al.*, 2016; Oliver *et al.*, 2015; Willems *et al.*, 2016): Ellis *et al.* (2017) reviewed 79 studies detailing post-release survival of chondrichthyans, 21 were trawl-related (including beam trawl and scallop dredge) and penaeid trawls were the subject of only two studies (Fennessy, 1994; Stobutzki *et al.*, 2002). This review, therefore, will focus on the PTS of chondrichthyans from all trawl gears given the factors affecting survival are likely common among the various trawl gears. The lack of chondrichthyan PTS data from penaeid trawls represents an avenue of research that requires urgent attention.

In the past, total fishing mortality was synonymous with the portion of the catch that is retained, and discard mortality was regarded as inconsequential (Muoneke and Childress, 1994). However, fishing mortality is now widely recognised as the sum of all sources of mortality resulting from interaction with fishing gears including misreported catch, discard mortality, mortality as a result of escaping or avoiding capture, ghost fishing and habitat degradation (Broadhurst *et al.*, 2006). Of these additional sources of fishing mortality, discard mortality is most commonly quantified in the primary literature. Most studies detail the post-release survival of species that are of commercial or recreational importance that are discarded for regulatory reasons (Broadhurst *et al.*, 2005; Broadhurst *et al.*, 2006; Butcher *et al.*, 2008; Campbell *et al.*, 2014). However, more recent studies focus on those discarded species that are of conservation concern (e.g. Benoît *et al.*, 2013; Epperly *et al.*, 2012; Mandelman and Farrington, 2007a; Mandelman and Farrington, 2007b; Wakefield *et al.*, 2016). Further, the European Union's common fisheries policy dictates that only fish species with a high survival rate can be discarded and, as such, there is a need to assess the PTS for a range of chondrichthyans in the north Atlantic (Ellis *et al.*, 2017).

Where comparisons have been made, PTS of chondrichthyans is generally higher than that of teleosts (see review by Broadhurst *et al.*, 2006). For example, Kaiser and Spencer (1995) assessed the survival of several species caught in a beam trawl in the United Kingdom and found that PTS of small spotted catsharks (*Scyliorhinus canicula*) was >90% after four days of captivity and 59% for *Leucoraja naevus*. These PTS estimates were comparable to those for crustaceans, molluscs and echinoderms that were also part of the catches but higher than those for three teleost species (*Pleuronectes platessa*, *Limanda limanda* and *Callionymus lyra*). Similarly, Benoît *et al.* (2012) estimated that the PTS of skates (*Leucoraja ocellata*, *Amblyraja radiata* and *Malacoraja senta*) caught in fish trawls in Canada was 97%, compared to 32%, 52% and 82% for Atlantic cod (*Gadus morhua*), American plaice (*Hippoglossoides platessoides*) and winter flounder (*Pseudopleuronectes americanus*), respectively. Benoît *et al.* (2013) hypothesised that trawl discard components with a high resistance to stress and hypoxia, resulting from low metabolic rates, are more likely to survive trawl capture and discarding.

Broadhurst *et al.* (2006) suggested that the absence of a swim bladder is at least partially responsible for the higher post-release survival of chondrichthyans. This is consistent with Benoît *et al.* (2013) who reported that the possession of a physoclistous swim bladder decreased time-to-mortality by 42.3%, compared to animals lacking a swim bladder. Time-to-mortality increased for those species possessing a phylostomous swim bladder but was 27.4% lower than animals lacking a swim bladder.

2.4.1 Methods used to quantify post-release survival from trawls

Ellis *et al.* (2017) provide an overview of methods used to assess PTS of chondrichthyans. These authors summarised the at-vessel mortality (AVM) and post-release mortality (PRM) of chondrichthyans as a function of gear type and taxonomic group. For the most part, the methods used to assess PTS in trawled chondrichthyans are similar to those used for teleosts and other fauna including tank-based studies, immediate assessment, tagging and blood chemistry. Each method has been found to include inherent biases which may affect PTS estimates. For example, tank-based studies may exacerbate the effects of trawl capture due to crowding, interaction with other captive animals and other such stressors. For this reason, researchers commonly state that PTS is underestimated (e.g. Mandelman *et al.*, 2013). In order to overcome these deficiencies, the use of animals to control for the effects of confinement are recommended (Broadhurst *et al.*, 2006). In contrast, the absence of predators and scavengers in tank-based studies lead to an overestimation of PTS. Laboratory-based studies offer some insight into the effects of factors such as air exposure but the derived estimates of PTS cannot be extrapolated to the fishery level due to the absence of interacting factors that decrease survival (Frick *et al.*, 2010; Heard *et al.*, 2014).

2.4.1.1 At-vessel mortality (AVM)

At-vessel mortality or within-net survival (Stobutzki *et al.*, 2002) is simply an observation of an animal's vitality on the deck of the fishing vessel immediately on landing. This method of assessing PTS was used by Fennessy (1994) and Stobutzki *et al.* (2002) to assess the survival of chondrichthyans in penaeid-trawl fisheries in South Africa and northern Australia, respectively. However, the PTS estimates from these studies are likely to be overestimated given the delayed effects of capture by trawl gear on PTS (e.g. Laptikhovsky, 2004; Van Beek *et al.*, 1990; Wassenberg and Hill, 1993). For example, Kaiser and Spencer (1995) held small spotted catsharks (*Scyliorhinus canicula*) and cuckoo skates (*Leucoraja naevus*) for up to 6 days after capture by a beam trawl in Wales, UK. These authors reported that PTS of *L. naevus* was 59% after 5 days despite 0% AVM. For the same period of observation, 94% of *S. canicula* survived after AVM was assessed as 3%. The study by Wassenberg and Hill (1993) is often cited by researchers holding animals for a period of time after capture. Wassenberg and Hill (1993) found teleost PTS decreased in the first three days post-capture, with only low levels of mortality occurring after this period. Consequently, most PTS studies aim to hold animals for at least three days (e.g. Enever *et al.*, 2009; Mandelman and Farrington, 2007b). The delayed mortality after trawl capture on individuals is a result of sub-lethal effects resulting from physical damage (Ellis *et al.*, 2017), physiological stress or disease (Davis, 2002). These effects largely invalidate AVM as a reliable proxy for PTS.

2.4.1.2 Vitality assessments

Another method employed by researchers when assessing chondrichthyan PTS, based on an animal's condition on capture, is a qualitative health score (Ellis *et al.*, 2017; Enever *et al.*, 2009) or vitality score (Benoît *et al.*, 2010; Benoît *et al.*, 2012). ICES (2014) describe three techniques used to assess survival at the point of capture, two of which have been used for chondrichthyans: coarse mortality indicators such as time-to-mortality (Benoît *et al.*, 2013) and semi-quantitative vitality assessments (Benoît *et al.*, 2010; Enever *et al.*, 2009). Vitality assessments allow for a large number of animals to be assessed for PTS cheaply and quickly (Benoît *et al.*, 2010; Ellis *et al.*, 2017), without the need to maintain the subjects in aquaria. However, quantifying PTS as a function of health score requires experimental studies to establish the relationship between the two metrics. For example, Benoît *et al.* (2012) found that all skates (Rajidae) caught by fish trawls with a vitality score of "excellent" or "good/fair" survived at least 48 hours post-capture, while PTS decreased to ~66% for those assessed as "poor" or "moribund". A significant limitation with this method is the subjective nature of the scores given by observers (Ellis *et al.*, 2017). To overcome this, Benoît *et al.* (2010) used a mixed effects model to quantify subjectivity among observers and suggested that incorporating this as a random effect can increase precision and accuracy of PTS estimates.

2.4.1.3 On-board holding tanks

Housing animals in on-board tanks supplied with seawater is the most common method used to assess PTS in chondrichthyans (Table 9). However, the containment of animals can introduce factors that increase physiological stress including excessive stocking densities, abrasion against the tank and with other animals and unfavourable environmental factors (Broadhurst *et al.*, 2006; Ellis *et al.*, 2017). This method allows for the observation of captured animals for periods of between 1 hour (Rodríguez–Cabello *et al.*, 2005) and 6 days (Kaiser and Spencer, 1995), with most holding periods between 48 and 72 hours (Table 9).

The duration of containment in PTS studies is problematic. It is difficult to isolate the effects of trawling from the effects of confinement, necessitating the use of controls where possible (Broadhurst *et al.*, 2006; Ellis *et al.*, 2017). However, acquiring control specimens for use in PTS studies is difficult (ICES, 2014). In order to control for the effects of tow duration, Enever *et al.* (2009) conducted shorter tows (0.75–2 h) in an experiment to assess the PTS of skates in the UK. After 66 hours, the PTS of the control animals was 87%, compared to 55% for animals caught in commercial tows (2.7–4.3 h). Beyond the short-term PTS estimates shown in Table 9, delayed effects that are likely to affect longer-term survival include predation (Kaiser and Spencer, 1995), infection and immunosuppression (Mandelman and

Farrington, 2007a). After four days, the appropriate captive duration advocated by Wassenberg and Hill (1993), PTS may also be influenced by the effects of starvation and longer-term studies require feeding of captive animals (e.g. Mandelman and Farrington, 2007b).

The space on vessels used to conduct PTS experiments is often limited, resulting in the use of relatively small holding tanks. This is likely to exacerbate the influence of stressors mentioned above and prohibit the housing of large chondrichthyans. As such, those studies in the primary literature that use on-board tanks to quantify PTS do so for relatively small demersal species (Table 9). Estimating the PTS of larger species is more challenging and derived by using the reciprocal of the AVM estimates only (Fennessy, 1994; Jaiteh *et al.*, 2014; Stobutzki *et al.*, 2002). In an attempt to quantify PTS for a range of tropical reef fish, Brown *et al.* (2010) constructed a large cage with a diameter of 1.9m and a depth of 15m. These cages were designed to allow fish with barotrauma to swim to a depth of 15m to test the effects of a range of relief treatments on captured fish. The cage was designed to hold animals in the lowermost 2.5m, thereby providing an enclosure >28 cubic metres in volume. The depth of the cage also allowed animals to be held in conditions more comparable to those from which they were exposed prior to capture. This apparatus was used to quantify post-release survival in samson fish (*Seriola hippos*) (Rowland, 2009), a large pelagic fish caught in deep water and, as such, may provide a useful method of assessing the PTS for larger chondrichthyans in future.

2.4.1.4 Submerged holding pens

In an effort to hold captive chondrichthyans in the environs to which they were exposed prior to capture, some authors have used pens anchored to the sea floor (Mandelman *et al.*, 2013; Mandelman and Farrington, 2007a; Rulifson, 2007). Rulifson (2007) and Mandelman and Farrington (2007a) quantified the PTS of spiny dogfish (*Squalus acanthias*) by housing trawled animals in square pens. Mandelman and Farrington (2007a) reported that the pens contributed to poor PTS given that previous research had reported high PTS in this species (Mandelman and Farrington, 2007b; Rulifson, 2007). However, Mandelman and Farrington (2007a) conducted their study in summer, while the studies by Mandelman and Farrington (2007b) and Rulifson (2007) were conducted in autumn and spring, respectively. Rulifson (2007) stated that their PTS estimates were higher than those from a previous unpublished report that was conducted in summer and this may explain the differences in PTS between these studies. The pens used by Mandelman and Farrington (2007a) and Rulifson (2007) were identical and trawl duration in both studies was similar, lending further evidence that the pens may not have been the only cause of the lower PTS reported in the former study.

Mandelman *et al.* (2013) used pens anchored to the sea floor to assess the PTS of several species of skate (Rajidae). These authors found that the pens had no effect on the PTS of the species assessed. These authors used round pens, as recommended by Rulifson (2007), and found that trawl duration significantly affected PTS and that those animals with poor health scores on capture may succumb to a range of factors that impact an individual's health. For example, the PTS of thorny skate (*Amblyraja radiata*) landed after a trawl duration of 2 hours with a condition index of 3 (extensive damage) was 60% after 72 hours while 96% of those with a condition index of 1 (minor damage) survived.

2.4.1.5 Land-based tanks

The use of land-based tanks requires fish be caught and transported to land, adding to the stress in captivity. The authors using this method acclimate captured animals for some period before conducting experiments to test the effects of various factors affecting PTS (Heard *et al.*, 2014). Mandelman and Farrington (2007b) assessed the physiological disturbance of *S. acanthias* resulting from multiple stressors: trawl capture, transport and captivity. Of the 34 animals transported to land-based tanks, 32 (~94%) survived 30 days captivity. This represents a higher PTS than a previous study by the same authors (Mandelman and Farrington, 2007a) and was similar to that reported by (Rulifson, 2007). However, in a more recent study, animals were captured in 45-minute trawls, much shorter than commercial trawls (Mandelman *et al.*, 2013) and the trawls undertaken in the two previous studies.

Cicia *et al.* (2012) acclimated little skate (*Leucoraja erinacea*) in land-based tanks in order to assess the effects of various levels of air exposure. Survival was higher in winter with 73% survival after 50 minutes air exposure (water temperature 4°C, air temperature 1°C) compared to 0% in summer (water temperature 18°C, air temperature 27°C). Aerial exposure of <1 min resulted in 100% survival in winter and 63% survival in summer. These authors suggested that the higher temperature gradient experienced in summer led to an increase in the effects of physiological impairment at times of aerial exposure.

2.4.1.6 Trawl simulation

Heard *et al.* (2014) tested the effects of crowding, air exposure and trawl duration on the survival of the sparsely spotted stingaree (*Urolophus paucimaculatus*). Individuals were placed in a trawl codend which was then rotated via an electrical motor in a 19 kL tank. Four treatments were tested: one hour trawl time, three-hour trawl time, one hour trawl time plus ten minutes air exposure and crowding (5 individuals at one time). Overall survival was 85% and there was no significant difference in survival between treatments. Mortalities only occurred after 48 hours. These authors state that the observed PTS in their study likely underestimates those from commercial trawls due to the absence of additional stressors such as temperature change.

Frick *et al.* (2010) conducted a near-identical experiment but assessed the effects of crowding, air exposure and trawl duration on the survival of Port Jackson sharks (*Heterodontus portjacksoni*) and gummy sharks (*Mustelus antarcticus*). These authors tested the effects of five treatments: 30 min trawl duration, 60 min trawl duration, 120-minute trawl duration, 60-minutes trawl duration plus ten-minutes air exposure and 60-minutes trawl duration plus crowding (three sharks in the codend at one time). These authors reported that treatment had no effect on the survival of either species, although PTS was low for *M. antarcticus* after 120-minute trawl duration. As with the study by Heard *et al.* (2014), Frick *et al.* (2010) state that the results of PTS studies conducted in the laboratory cannot be extrapolated to animals caught in the wild due to the absence of additional, important stressors that influence PTS.

2.4.1.7 Tagging

Although tagging has been used to assess PTS in chondrichthyans caught in longline fisheries (e.g. Campana *et al.*, 2009), this method has not been used for trawl-caught chondrichthyans. However, tagging studies have been used to assess the movement of chondrichthyans and have provided some information regarding post-release survival. For example, Walker *et al.* (1997) reported the distribution and movement of thornback (*Raja clavata*), spotted (*R. montagui*) and starry (*Amblyraja radiata*) skates from tagging studies conducted over 17 years in the English Channel and the North Sea. These authors reported a recapture rate of ~27% across all species. However, the highest recapture rate, which was found for *R. clavata*, corroborates the relatively high post-release survival quantified in short-term survival studies (Enever *et al.*, 2009; Saygu and Deval, 2014). As such, assessing the longer-term survival of chondrichthyans is possible by using mark-recapture studies in conjunction with short-term survival studies. Ideally, such studies should also include control animals to determine the effects of tagging-only on survival (ICES, 2014).

Biotelemetry studies have also been used to assess post-release survival in large pelagic sharks. For example, Campana *et al.* (2016) used pop-up satellite archival tags (PSATs) to estimate post-release survival of blue sharks (*Prionace glauca*), shortfin mako (*Isurus oxyrinchus*) and porbeagle (*Lamna nasus*) in the Canadian pelagic longline fishery. This study showed that *I. oxyrinchus* and *L. nasus* experience lower post-release survival than *P. glauca*, with most of the post-release mortality occurring within 2 days of release. Further, Dudgeon *et al.* (2013) used passive acoustic telemetry to assess the site fidelity of the zebra shark (*Stegostoma fasciatum*) in south east Queensland and this represents an avenue of research yet to be explored in quantifying post-release survival for chondrichthyans.

These tags have the advantage of providing high resolution spatial information for animals after release in natural conditions (ICES, 2014). However, they may not be appropriate for small species of chondrichthyan, while the use of acoustic tags requires invasive surgery and may reduce PTS.

2.4.2 Factors affecting the PTS of chondrichthyans

Again, given the lack of scientific studies in the primary literature assessing factors affecting PTS of chondrichthyans caught in penaeid trawls, a review of factors affecting PTS from all trawl gears is presented below. The review by Davis (2002) discusses the factors that affect post-release survival and is well-cited in PTS studies. This author discusses classes of interacting stressors including capture stressors (e.g. crushing and wounding), fishing conditions (e.g. towing time, time-on-deck) and biological attributes (e.g. size and species). Davis (2002) reviewed the effect of temperature, anoxia, sea conditions, air exposure and fish size and concluded that a combination of laboratory and field experiments is necessary in determining the effect of these and other factors on post-release survival.

Broadhurst *et al.* (2006) discusses the factors that may influence PTS and cites morphological characteristics (e.g. the presence of a shell), gender, swim bladder characteristics, catch volume, time-on-deck and temperature gradient as being likely to affect the post-release survival of discards. Generally, PTS is likely to decrease with increasing catch volumes, increasing time-on-deck and increasing air temperature. Similar results were reported in the review Ellis *et al.* (2017). Suuronen and Erickson (2011) reviewed the factors affecting mortality of discarded fish and reported that time-on-deck and air temperature are important predictors of survival, along with temperature gradient, tow duration and catch size in fish trawls. Similarly, Andrew and Pepperell (1992) found that survival was dependent on both biological and operational factors, such as tow duration and time-on-deck.

2.4.2.1 Tow duration

Where measured, increased tow duration has resulted in lower PTS for chondrichthyans. For example, Saygu and Deval (2014) found that increasing tow duration had a negative effect on the survival of two skates (*Raja clavata* and *R. miraletus*) in a fish trawl fishery in Turkey. Similarly, Enever *et al.* (2009) assessed PTS in four skates (*Leucoraja naevus*, *Raja microocellata*, *R. brachyura* and *R. clavata*) and found that mean survival from shorter (~48 min) experimental trawls was 87%, compared to 55% for commercial trawls with a mean duration of 216 minutes. Further, Fennessy (1994) reported that shorter tows resulted in increased PTS of backwater butterfly rays (*Gymnura natalensis*) in a South African penaeid-trawl fishery. Mandelman *et al.* (2013) found that the survival of three species of skate (*Amblyraja radiata*, *Leucoraja ocellata* and *L. erinacea*) was higher from short control trawls than from longer commercial tows; however, this result was confounded by the effect of catch weight with survival higher from trawls where catch biomass was relatively low, suggesting that catch duration and catch size were correlated. These authors also state that assessing the effect of tow duration is limited by the inability to determine the time an animal is caught in the net.

2.4.2.2 Catch weight

As with tow duration, catch weight is a common factor affecting the PTS of chondrichthyans. Enever *et al.* (2010) reported improved health scores for skates (*Leucoraja naevus*, *Raja microocellata*, *R. brachyura* and *R. clavata*) from shorter experimental trawls conducted in a United Kingdom fish trawl fishery. The use of effective square mesh codend BRDs reduced bycatch and resulted in lower catch weights. Although not stated in this study, square mesh codends result in increased water flow at the aft end of a trawl net (Courtney *et al.*, 2008) which would have likely alleviated possible suffocation and facilitated improved survival of skates within the net. Similarly, Benoît *et al.* (2010) reported that increasing catch weight led to poorer health scores for skates in a Canadian fish trawl fishery.

Catch weight and AVM were correlated for backwater butterfly ray (*G. natalensis*) caught in penaeid trawls in South Africa (Fennessy, 1994). Catch weights in excess of 300 kg were found to significantly decrease the survival of *G. natalensis*. Fennessy (1994) discussed the fact that chondrichthyans are less susceptible to the effects of barotrauma during capture but are exposed to crushing in the codend during retrieval. Mandelman and Farrington (2007a) reported catch weights in excess of 200 kg resulted in lower PTS for spiny dogfish (*Squalus acanthias*) from fish trawls in Massachusetts, U.S.A. Similarly, Saygu and Deval (2014) reported that the PTS of brown skate (*Raja miraletus*) increased when catch weight was <100 kg in a Turkish fish trawl fishery. Mandelman *et al.* (2013) found that shorter tows resulted in higher PTS for three species of skate (*Amblyraja radiata*, *L. ocellata* and *L. erinacea*) but also found that catch weights <318kg increased PTS for *L. erinacea* and *A. radiata*, while a higher injury occurrence in *A. radiata* was associated with increased catch weights. The study by Mandelman *et al.* (2013) demonstrates that tow duration and catch weight are likely to be correlated: increased catch weights result in poorer health outcomes from crushing within the net and interaction with other animals. Conversely, the shorter control tows undertaken by Mandelman *et al.* (2013), Saygu and Deval (2014) and Enever *et al.* (2009) would have resulted in relatively low catch weights and isolating the effect of tow duration and catch weight for the respective species is, therefore, difficult.

2.4.2.3 Air exposure

Air exposure is a factor often confounded by tow duration and catch weight. Logically, if long tows result in higher catch weights, fishing crews are more likely to take longer to process the catch, prolonging air exposure for discards. For the most part, crews that process catches are more likely to remove the target species from a catch first, leaving the unwanted portion in the processing area before discarding back to the sea. To isolate the effects of air exposure, several studies have used laboratory-based experiments. For example, Cicia *et al.* (2012) acclimated trawl-caught little skates (*L. erinacea*) in land-based tanks for ten days before subjecting individuals to various levels of air exposure. Survival decreased from 100% for <1 min air exposure to 73% after 50 minutes exposure in an experiment conducted in the boreal winter. In summer, however, survival after <1 minute air exposure decreased to 63% and no animals survived 50 minutes air exposure. These authors give a detailed explanation of the effect of aerial exposure, describing that the collapse of the lamellae inhibits gas exchange leading to a hypercapnic state and blood acidosis.

Frick *et al.* (2010) found that 10 minutes air exposure exacerbated physiological stress after simulated trawling for gummy sharks (*M. antarcticus*). Although sample size precluded robust analyses, the authors state that larger sample sizes would likely reveal increased physiological disturbance when animals are exposed to air for extended periods after trawling. In a very similar experiment, Heard *et al.* (2014) assessed the effect of air exposure on sparsely spotted stingarees (*U. paucimaculatus*) and found that exposure to air after trawling was the primary source of stress for stingarees caught in trawling operations. In both of these studies, however, the authors state that PTS would be lower from commercial trawl operations due to the absence in the laboratory setting of confounding factors which influence survival. As such, these results cannot be extrapolated to commercial fisheries (Frick *et al.*, 2010). Interestingly, Frick *et al.* (2010) found that no Port Jackson sharks (*H. portjacksoni*) died as a result of simulated trawling and no physiological disturbance was measured after various levels of trawling and air exposure, highlighting the species-specific nature of PTS.

Few field-based studies have assessed the effect of air exposure on the PTS of chondrichthyans. Benoît *et al.* (2010) reported that air exposure was negatively correlated to health score for skates caught in Canadian fish trawls, while Benoît *et al.* (2012) found longer deck times, confounded by larger catch sizes, contributed to poorer vitality in discarded skates from the same fishery.

2.4.2.4 Fish size and sex

The effects of size and sex on the PTS of chondrichthyans are often confounded due to females growing larger than males in a high proportion of species (Stobutzki *et al.*, 2002). The effect of size has been assessed primarily for skates (Rajidae) in the primary literature, with larger individuals more likely to survive trawl capture. Depestele *et al.* (2014) studied the PTS of skates (no species identified) in a beam trawl fishery in the North Sea and determined that survival and length were positively correlated. Similarly, Saygu and Deval (2014) reported that survival increased with size for skates (*R. miraletus* and *R. clavata*) caught by fish trawls in Turkey and Mandelman *et al.* (2013) found that PTS increased with size for larger thorny skates (*A. radiata*) caught in Massachusetts, U.S.A., fish trawls. Benoît *et al.* (2013) used mass-specific respiration demand (MSRD) as a proxy for size and found that MSRD and survival were positively correlated for three species of skate (*Leucoraja ocellata*, *Malacoraja senta* and *Amblyraja radiata*) caught in Canadian fish trawls. These authors state that smaller individuals are more prone to crushing in the net and are likely more susceptible to hypoxia due to their higher mass-specific metabolic rate and higher energy cost for breathing.

In contrast, Fennessy (1994) found that backwater butterfly rays (*Gymnura natalensis*) between 50 cm and 1 m disc width had lower survival than animals < 50 cm disc width: however, these individuals were assessed for survival on capture (i.e., at-vessel or within net survival) and were not held in tanks for any period. As such, caution is required when assessing the results and making comparisons between studies. Stobutzki *et al.* (2002) also assessed AVM of chondrichthyans caught in an Australian penaeid-trawl fishery and found that larger individuals were more likely to survive than smaller individuals: however, they found size was confounded by gender, with females being larger and more likely to survive. This is an example of the difficulty in isolating individual factors that influence PTS. Studies by Dapp *et al.* (2017) and Braccini *et al.* (2012), assessing post-release survival of chondrichthyans caught by gillnets in Australia, both reported that post-release survival was affected by size, while gender had no effect.

Enever *et al.* (2009) tested the effect of both sex and size on the survival of skates (*Leucoraja naevus*, *Raja microocellata*, *R. brachyura* and *R. clavata*) and found that males were more likely to die as a result of capture, but these authors explicitly state that this was not a function of size. Studies that have found sex-specific influence on PTS in chondrichthyans invariably report that female survival is higher (Enever *et al.*, 2009; Enever *et al.*, 2010; Laptikhovsky, 2004; Mandelman *et al.*, 2013; Stobutzki *et al.*, 2002). 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. Mandelman *et al.* (2013) hypothesise that the presence of claspers may lead to injuries for males.

2.4.2.5 Morphology, physiology and biology

Two studies have highlighted species-specific differences in PTS. In their study, Enever *et al.* (2009) reported that the survival of the thorny skate (*Raja clavata*) was higher than three confamilials (*Leucoraja naevus*, *R. microocellata* and *R. brachyura*) due to the physical protection afforded by the species' spinulose skin. Similarly, Saygu and Deval (2014) reported that the PTS of *R. clavata* was higher than for the brown skate (*R. miraletus*), attributing the difference to morphology.

In a meta-analysis, Dapp *et al.* (2016) reported that PTS for obligate ram-ventilating chondrichthyans was 15.8% compared to 58.1% for stationary-respiring species. This difference was attributed to obligate ram-ventilating chondrichthyans' reduced ability to recover from capture: on release, the physiological disruption caused by capture prevents obligate ram-ventilating species from swimming and, therefore, respiring. In contrast, stationary-respiring animals are able to resume normal breathing as soon as they return to the water. Dapp *et al.* (2016) cited a study by Stobutzki *et al.* (2002), who assessed AVM for a range of species caught in penaeid trawls in northern Australia and found that AVM was 40% (i.e., 60% survival) for batoids (*Neotrygon leylandi, Maculabatis toshi, Gymnura australis* and *Rhyncobatus australiae*), compared to 61% (39% survival) for sharks (*Carcharhinus dussumieri*, *Carcharhinus sorrah*, *Carcharhinus tilstoni*, *Rhizoprionodon acutus* and *Hemigaleus australiensis*). In contrast, Fennessy (1994) reported that AVM was not species-specific, finding little difference among a range of sharks and rays caught in penaeid trawls in South Africa, although sample size was low for half of the species assessed. As discussed previously, both Fennessy (1994) and Stobutzki *et al.* (2002) assess AVM only and, therefore, likely underestimate PTS of the respective species.

Benoît *et al.* (2013) found that sedentary species, including skates (*Leucoraja ocellata*, *Malacoraja senta* and *Amblyraja radiata*), had higher times-to-mortality following trawl capture than mobile species. These authors used a binomial factor (sedentary or not) in a survival function and found that fast-swimming species such as mackerel succumb to hypoxia much faster than sedentary species. Generally, sedentary species are more resistant to hypoxia making them more resilient to capture and release than species with a high oxygen demand. This appears to be the case for batoids although, as stated previously, very few studies have assessed PTS across a range of species.

2.4.2.6 Depth

The PTS of skates decreased with increasing depth in a squid trawl fishery in the Falkland Islands (Laptikhovsky, 2004). However, this result is confounded by the species assessed: shallow water shelf species (*Psammobatis* sp., *Bathyraja brachiurops* and *B. magellanica*) had higher survival than species occurring in deeper water (*B. albomaculata*, *B. griseocauda* and *Bathyraja* sp.). Laptikhovsky (2004) attributes this to the shallow water species' resilience to environmental change. This author used observed survival and does not quantitatively assess factors likely to affect PTS. For example, *Bathyraja* sp. was the largest species caught (21–74 cm DW) and exhibited 75% survival, contradicting the earlier assertion that fewer deep water species survived. Similarly, PTS of *B. albomaculata* was 71.4% despite it being a deep-water species. As such, it is difficult to draw conclusions regarding the effect of depth in this study.

Fennessy (1994) found that depth had no effect on the AVM of the backwater butterfly ray (*G. natalensis*) in a South African penaeid-trawl fishery. These authors categorised trawl depth as shallow (20–33 m) and deep (33–45 m) and found no difference in PTS which is not unexpected given the lack of contrast between depth classes. Several other studies have also reported no effect of depth on PTS (Benoît *et al.*, 2010; Depestele *et al.*, 2014; Enever *et al.*, 2009; Rodríguez–Cabello *et al.*, 2005).

Benoît *et al.* (2013) found that the time-to-mortality of winter skate (*Leucoraja ocellata*) caught by Canadian fish trawls decreased with increasing depth. In contrast, depth and time-to-mortality were correlated for thorny skate (*Amblyraja radiata*), while depth had no effect on the time-to-mortality for smooth skate (*Malacoraja senta*). However, these authors offer no discussion about the size of the depth related size structure of the respective species and it is unclear whether the size of the animals varied across depths. For example, the size of *A. radiata* has been shown to be positively correlated to PTS (Mandelman *et al.*, 2013) and larger animals sampled in deeper water would likely confound the effect of depth.

2.4.2.7 Temperature gradient

Davis (2002) stated that exposure to increases in temperature during capture results in additional stress and lower survival. This author attributed this to an increase in body core temperature, with smaller fish affected more. Further, Broadhurst *et al.* (2006) reported a positive correlation between the discard mortality of most fish and the temperature of the air. In their review, Suuronen and Erickson (2011) found that temperature gradient, the difference in temperature between the surface and the bottom, was negatively correlated to survival post-capture.

In accord with these authors, temperature gradient had a significant effect on the survival of little skates (*L. erinacea*) caught in fish trawls in New Hampshire, U.S.A (Cicia *et al.*, 2012). These authors found that air temperatures in winter were 3°C lower than the water temperature and survival ranged between 100% for <1 min air exposure and 73% after 50 minutes exposure. In contrast, air temperature in summer was 9°C higher than the water temperature resulting in significant decreases in survival to 63% after <1 minute air exposure and 0% after 50 minutes air exposure. Mandelman *et al.* (2013) assessed the effect of temperature change on the PTS of *L. erinacea*, *A. radiata* and *Malacoraja senta* but found that survival was lowest when temperature gradient was lowest. These authors could not explain the difference between their study and the results reported by Cicia *et al.* (2012), concluding that additional factors not assessed elevated the number of fatalities in the colder months.

In the study by Benoît *et al.* (2013), *L. ocellata* had a longer time-to-mortality, compared to *Amblyraja radiata*. Given the results reported by Cicia *et al.* (2012), this result may have been a function of temperature gradient between the surface and the depths at which the respective species were caught. The temperature of the water at 22–55 m where *L. ocellata* were caught was 5–19.9°C and the ambient air temperature for this area in September is approximately 14–15°C. In contrast, *Amblyraja radiata* were caught at depths between 49–300 m where temperature ranged between 1°C and 5.5°C, representing a significant temperature gradient between ambient air and the sea floor. Further, the fact that increasing depth decreased time-to-mortality of *L. ocellata* may have been a result of the increasing temperature gradient between air and water as depth increased.

2.4.3 Summary

The PTS of chondrichthyans is poorly understood despite its importance when assessing ecological risk. This is largely due to the cost and logistical constraints of conducting field-based experiments needed to quantify this metric. The majority of studies in the primary literature assessing the PTS of chondrichthyans have been conducted in northern hemisphere fish trawls and demonstrate that PTS varies between species. Factors found to have affected PTS include catch weight, air exposure, tow duration, fish size, temperature, and sex. These factors are likely to interact, necessitating the use of controls to isolate the effect of individual factors.

2.5 Conclusion

For the most part, TEDs have been shown to reduce the capture of turtles in penaeid trawls. A flow-on benefit of these devices has been a reduction in the capture of large chondrichthyans which also escape penaeid trawls via the TED escape hole. However, current TED regulations allow for the capture of smaller chondrichthyans, some of which are threatened with an elevated risk of extinction. Despite this, there are very few studies in the primary literature that assess the various design aspects that influence the exclusion of chondrichthyans such as TED bar spacing, grid angle, grid orientation and escape hole size. Extensive research has determined the factors that facilitate turtle escape and the retention of target species: however, very little research has been undertaken to determine the factors that increase the escape of chondrichthyans from trawls. In order to evaluate the risk posed to chondrichthyans impacted by penaeid trawling, it is necessary to quantify their post-trawl survival. Reliable survival estimates, however, are absent from the primary literature. As with estimates of escape, post-trawl survival is dependent on several interacting factors which are difficult to isolate during field experiments.

Estimating both escape and post-trawl survival is difficult and costly. Field research, involving the use of vessels operating under commercial conditions, is necessary in order to derive reliable estimates. Low sample sizes represent a significant impediment to obtaining robust estimates and testing the effects of more than one or two factors that affect either metric is difficult without introducing biases from uncontrolled influences. Additionally, post-trawl survival is species-specific and, as such, it is difficult to infer results across genera or families. Given these limitations, quantifying both escape and PTS remain a subject of scientific research that requires attention, particularly for those species that are threatened by overfishing. However, In the absence of reliable estimates of escape and PTS, ecological risk assessments will generally utilise more conservative estimates, which may artificially inflate the level of risk for some species. The cost of at-sea experiments to quantify escape and PTS could be reduced by assisting penaeid-trawl fishers to conduct experiments in collaboration with researchers. Such experiments not only result in improved estimates, but also provides relevant stakeholders, conservationists and the broader community with the confidence that risks to threatened species are being addressed.

For those species where escape and PTS are quantified but low, steps can be taken to improve these estimates. Escape can be improved through reductions in bar spacing and PTS can be increased by reducing trawl duration and the amount of time each animal is exposed to air. Improvement in TED design is relatively easy to implement and regulate: however, voluntary measures such as reducing trawl duration are more difficult to enforce and likely to have operational costs to fishers. Therefore, mitigation of risk for chondrichthyans will generally be facilitated through improvements in TED and BRD design.

3. Life-history characteristics of the eastern shovelnose ray, *Aptychotrema rostrata* (Shaw, 1794), from southern Queensland, Australia

Plate 3: First shot of the night, offshore Surfer's Paradise on the FV *C–Rainger*, November 2016.

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3.1 Introduction

The Queensland east coast otter trawl fishery (QECOTF) is the largest penaeid-trawl fishery in Australia. This fishery targets shrimps (Penaeidae: *Melicertus* spp., *Penaeus* spp., *Metapenaeus* spp.), sea scallops (Pectinidae: *Ylistrum balloti*), bugs (Scyllaridae: *Thenus* spp. and *Ibacus* spp.) and squid (Teuthoidea) with demersal otter trawl gear. In 2019, logbook data indicated that 299 vessels fished 35,950 days and landed ~5,986 t of product for sale at both domestic and international markets. Further, two vessels target stout whiting (*Sillago robusta*), using Danish seine and fish trawl gear, in southern Queensland and are subject to an annual total allowable catch (TAC) of ~1100 t.

It has been estimated that 55% of the global catch from penaeid trawls is discarded (Gilman *et al.*, 2020). The discard rate from the QECOTF is higher at 70% resulting in >25,000 t discarded annually (Wang *et al.*, 2020), representing 28.5% of Australia's total annual discards (Kennelly, 2020). Consequently, quantifying and mitigating discards have been the subjects of significant research efforts in Queensland since the mid-1990s (e.g. Robins–Troeger, 1994; Robins and McGilvray, 1999). Hundreds of species comprise the discarded portion of the QECTOF catch (Courtney *et al.*, 2008; Courtney *et al.*, 2006), some of which are of conservation concern such as sea turtles (McGilvray *et al.*, 1999).

Elasmobranchs (i.e., sharks and rays) are one component of penaeid-trawl discards that have received increasing attention in the last two decades (Dulvy *et al.*, 2017). Elasmobranch life history strategies, including late maturity, few offspring, long life spans and slow growth (Dulvy *et al.*, 2008), make this group vulnerable to overexploitation (Stevens *et al.*, 2000). Twenty-five percent of elasmobranchs have an elevated risk of extinction due to capture in fisheries (Dulvy *et al.*, 2017; Simpfendorfer and Dulvy, 2017) and the Rhinopristiformes (wedgefishes and guitarfishes) are of particular concern (Kyne *et al.*, 2020). The introduction of turtle excluder devices (TEDs) has gone some way to reduce this risk in penaeid-trawl fisheries, particularly for larger species: however, TEDs remain ineffective for smaller elasmobranchs (Campbell *et al.*, 2020).

The TEDs used in the QECOTF have no effect on the catch rate of the eastern shovelnose ray (Trygonorrhinidae: *Aptychotrema rostrata,* Shaw 1794) (Courtney *et al.*, 2008). This is the most common elasmobranch in the discarded portion of the penaeid-trawl (Kyne *et al.*, 2002) and *S. robusta* (Rowsell and Davies, 2012) catches in southern Queensland (>22°S). *Aptychotrema rostrata* is endemic to the east coast of Australia between Halifax Bay in north Queensland (18°30'S) and Merimbula in southern New South Wales (36°53'S). The species rarely exceeds 1 m total length (TL), generally in depths <100 m (Last and Stevens, 2009), and feeds on crustaceans, teleost fish and squid (Kyne and Bennett, 2002a). In southern Queensland, parturition occurs in November and December after a gestation period of three to five months with litter sizes of 4–18 pups (Last *et al.*, 2016).

In Queensland, the incidental capture of *A. rostrata* in the QECOTF is the main source of fishing mortality, although post-release survival is high (Campbell *et al.*, 2018). In 2010, the two vessels targeting *S. robusta* caught 3075 *A. rostrata*, of which ~22% were released alive (Rowsell and Davies, 2012). Recreational anglers land *A. rostrata* (Kyne and Stevens, 2015); however, the catch is negligible in Queensland (James Webley, Fisheries Queensland, pers. comm.).

Despite its frequent occurrence in trawl catches, age and growth studies on *A. rostrata* are lacking. Diet (Kyne and Bennett, 2002a), dentition (Gutteridge and Bennett, 2014), sensory characteristics (Hart *et al.*, 2004; Wueringer *et al.*, 2009) and post-trawl survival (Campbell *et al.*, 2018) have been the subject of recent research. Although reproductive strategies were described in two studies (Kyne and Bennett, 2002b; Kyne *et al.*, 2016), no previous study has quantified growth and age-at-maturity.

The lack of these data and the absence of information regarding the number of *A. rostrata* caught annually are the main impediments for the assessment of population status. Currently, the IUCN Red List of Threatened Species categorise *A. rostrata* as "Least Concern" (Kyne and Stevens, 2015). In Australia, all fisheries are subject to environmental assessment, whereby jurisdictions are required to demonstrate the impacts on individual species, both target and non-target, are sustainable in the long term. Failure to do so can result in the revocation of export privileges, prohibiting access to lucrative international markets.

Previous qualitative ecological risk assessments (ERAs) indicate that trawling in Queensland (Jacobsen *et al.*, 2018; Pears *et al.*, 2012) and New South Wales (Astles *et al.*, 2009) poses a high ecological risk to *A. rostrata* in the respective jurisdictions. These ERAs rely on qualitative assessments of a species' exposure and resilience to trawling, rather than empirical data, to assess risk. Generally, qualitative ERAs overestimate the ecological risk posed by fishing when compared to results from formal stock assessments, more so than quantitative ERAs (Zhou *et al.*, 2016). As such, life history data are fundamental to assessing stock status and form the basis of quantitative ERAs, an improved method for assessing data poor species such as *A. rostrata*. The aim of the present study, therefore, was to estimate the growth parameters and age-at-maturity of *A. rostrata*.

3.2 Materials and methods

Specimens of *A. rostrata* were primarily obtained from the operator of a Danish seine vessel, the FV *San Antone II*, targeting *S. robusta* in southern Queensland on an *ad-hoc* basis in the period between April 2016 and November 2017. The *San Antone II* is a 17 m steel twin-hulled vessel powered by two 148 kW diesel engines. The Danish seine gear consisted of two 2500 m sweeps separated by a single net with a headline length of 34.75 m with a mesh size of 85 mm in the wings and 55 mm in the codend. Samples were collected in southern Queensland waters between Sandy Cape (24°42.043' S, 153°16.027' E) and Coolangatta (28°09.844' S, 153°32.942' E) in depths between 35 and 50 m. During commercial operations, *A. rostrata* were removed from the catch and stored whole in the vessel's freezer for processing in the laboratory.

Sample collection on the *San Antone II* was supplemented by specimens obtained during the post-trawl survival (PTS) experiments conducted by Campbell *et al.* (2018).

3.2.1 Laboratory processing

All *A. rostrata* were thawed, sexed, weighed (±0.01 g) and measured (total length, TL, ±0.1 cm). In accordance with Pierce and Bennett (2009), a segment of four or five vertebrae, located at the posterior of the abdominal cavity, was excised. Each segment was cleaned following Goldman *et al.* (2004) and air dried. The neural and haemal arches were removed, along with any remnant connective tissue. After drying, each segment was embedded in polyester resin and sectioned with a Buehler IsoMet Low Speed cutting saw (www.buehler.com/isoMet-low-speed-cutter.php), at a width of ~200 µm, and mounted on a microscope slide. The vertebral sections were examined with a Leica M60 stereo microscope (www.leica-microsystems.com/products/stereo-microscopes-macroscopes/p/leica-m80/), under reflected light on a matt black background, and photographed with a Leica IC90 E digital camera (www.leica-microsystems.com/products/microscope-cameras/p/leica-ic90-e/).

The maturity of each individual was assessed according to Kyne *et al.* (2016). Maturity in males depended on the calcification of the claspers, categorised as immature (possessing short, flexible, uncalcified claspers) or mature (rigid, calcified and elongated claspers). Mature female *A. rostrata* possessed one or more of the following: developed ovaries with yellow vitellogenic follicles ≥5 mm diameter; fully developed oviducal glands and uteri; uterine eggs; and embryos *in situ*. Immature females were categorised by undifferentiated ovaries, undeveloped oviducal glands and thin uteri.

3.2.2 Ageing

Nominal age was estimated by two readers based on the number of band pairs. A band pair was defined following Figure 1c from Rolim *et al.* (2020) as one (narrow) translucent band and one wide (opaque) band, combined. Initially, the birth mark was defined as an angle change along the corpus calcareum (White *et al.*, 2014), associated with the first distinct opaque band after the focus (called the 'birth mark', Campana, 2014). However, preliminary investigation revealed that the birth mark and the change of angle was absent or difficult to identify in a high proportion of individuals. As such, the first growth band (i.e., 1 year of age) was identified using a method described by Campana (2014). The mean distance between the waist and distal edge of the first growth band was calculated by measuring this distance for those one-year-old animals (<~25 cm TL) where the birthmark was visible. This distance was measured with the Leica Application Suite software associated with the camera used to view centrum images. A line of this length was superimposed on the image of each sectioned centrum to determine the expected location of the first complete opaque band after the birth mark.

Counts were made without knowledge of the size or sex of the animal and the readability of each section was qualitatively assessed in accord with Officer *et al.* (1996). Where counts differed between readers, the count by the experienced reader was accepted. Three measures of precision were calculated to assess consistency between readers: 1) percent agreement (PA); 2) average percent error (APE, Beamish and Fournier, 1981); and 3) average coefficient of variation (ACV, Chang, 1982). Further, Bowker's test of symmetry was used to assess bias between readers.

3.2.3 Marginal increment ratio (MIR)

To determine the periodicity of band formation, monthly MIR was calculated following Natanson *et al.* (1995), who defined MIR as $MIR = (CR–CR_n)/(CR_n–CR_{n–1})$, where CR is the centrum radius, CR_n is the radius of the final complete band pair and $CR_{n–1}$ is the radius of the next to last complete band pair. Given this method, MIR was calculated only for animals aged ≥2 years. Following (Simpfendorfer *et al.*, 2000), MIR was compared between months using the Kruskal–Wallis one way analysis of variance on ranks.

Edge type was qualitatively assessed to provide further evidence of band formation periodicity (Cailliet *et al.*, 2006) and was classified into three levels: 'new', 'intermediate' and 'wide'. A 'new' edge was one where an opaque zone occurred at the distal edge of the centrum irrespective of the width of the opaque band. An edge of a centrum with any translucence visible beyond the last complete band pair was categorised as 'intermediate' and an edge was classified as 'wide' if the width of the translucent band beyond the last complete band pair was ≥2/3 the width of the previous translucent band. A chi–squared test was used to compare the observed frequency of each edge type, as a function of month, to the expected frequencies. In this case, the null hypothesis of the test was that the frequency of edge type was not dependent on month of capture.

3.2.4 Growth

Band pair counts (i.e., nominal age) were adjusted for growth beyond the last complete band pair based on edge type (Pierce and Bennett, 2009). Nominal age was increased by 0.33 year for intermediate edges and 0.66 year for wide edges.

Initial analysis indicated that younger individuals were under sampled. As such, back calculation techniques were used to increase the sample size of smaller size classes. The Linear-modified Dahl–Lea method (Francis, 1990) was used to estimate the total length (L_a) of each individual at age a as follows:

$$L_a = L_c \times \left(\frac{b + mCR_a}{b + mCR_c}\right)$$

where L_c is the length at capture; CR_a is the centrum radius at age a; CR_c is the centrum radius at capture; and b and m are the coefficients of the linear regression between CR_c and L_c. This method was preferred to the Dahl–Lea direct proportions method as the CR_c–L_c relationship did not pass through the origin (Goldman, 2005). Following Goldman (2005), the quadratic-modified Dahl–Lea method (Francis, 1990) was used for comparison to the linear-modified Dahl–Lea method to determine the most appropriate approach for estimating L_a as a function of CR_a. Francis (1990) defined the quadratic-modified Dahl–Lea equation as:

$$L_a = L_c \times \left(\frac{d + eCR_a + fCR_a^{\,2}}{d + eCR_c + fCR_c^{\,2}}\right)$$

where *d*, *e* and *f* are the quadratic regression estimates. The mean observed lengths and the mean back calculated lengths, as a function of age, were compared using two-sample *t*-Tests where sample size permitted. In this case, the observed lengths were restricted to those animals where new or intermediate edges occurred at the distal edge of the centrum.

In accord with Smart *et al.* (2016), three growth functions were used to estimate mean length-at-age: von Bertalanffy growth function (VBGF), logistic function and Gompertz function (Table 1). In all instances, the biologically relevant length-at-birth (L_0) was estimated, rather than the age when length is zero (i.e., t_0), as recommended for elasmobranchs by Cailliet *et al.* (2006). Relevant parameters were estimated via non-linear least squares regression: however, the under sampling of larger individuals resulted in an underestimate of L_∞. As such, a Bayesian approach using Markov chain Monte Carlo (MCMC) was used to estimate biologically appropriate growth parameters (Emmons *et al.*, 2021).

Bayesian models were fit using the 'BayesGrowth' package (Smart, 2020, accessed 18 February 2021) using R statistical software (Version 3.6.1, R Foundation for Statistical Computing, Vienna, Austria, see https://www.R-project.org/, accessed 18 February 2021), in accord with methods described by Smart and Grammer (2021) and Emmons *et al.* (2021). The 'BayesGrowth' package uses the 'Stan' computer program (Carpenter *et al.*, 2017), via the 'Rstan' package (Stan Development Team, 2020) to perform MCMC using No U-Turn Sampling (NUTS). Four MCMC chains with 10,000 simulations, with a burn in period of 5,000 simulations, were used to determine parameter posterior distributions. Model convergence was assessed using the Gelman–Rubin test and diagnostic plots generated using the 'Bayesplot' package (Gabry, 2020, accessed 18 February 2021) in R.

The models were fit with a normal residual error structure (σ). Prior distributions for the L_0 and L_∞ estimates were informed by data published by Last *et al.* (2016). These authors report the maximum size of *A. rostrata* as 1200 mm TL and a length-at-birth (L_0) of 130–150 mm TL. Given this information, priors were set at $L_\infty \sim N(1200, 50)$ and $L_0 \sim N(140, 10)$. A non-informative prior was used for σ and a common non-informative prior was used for the growth coefficients of candidate models (k, g_1 and g_2, Table 1). An upper bound was nominated for the uniform distributions of σ and k of 100 and 0.3 year^{-1}, respectively.

The common non-informative prior for the growth coefficients allowed for comparison of the three candidate growth functions, each with identical priors. Leave-one-out-information-criterion weights (LOOICw), calculated within the 'BayesGrowth' package using the 'loo' R package (Vehtari *et al.*, 2020), were used to determine the most appropriate candidate model. As with the Akaike weights in the frequentist approach, the candidate model with the highest LOOICw was considered the most appropriate.

Table 1: Equations of the three candidate growth functions used to assess the growth of 212 *Aptychotrema rostrata* caught in southeast Queensland, Australia, between April 2016 and November 2017. L_t is the length at age t; L_∞ is the asymptotic length; L_0 is the length at $t = 0$; and k, g_1 and g_2 are coefficients of the respective growth functions to be estimated.

Growth Function	Equation
Von Bertalanffy	$L_t = L_0 + (L_\infty - L_0)(1 - e^{-kt})$
Gompertz	$L_t = L_0 \times e^{\left(Ln\left(\frac{L_\infty}{L_0}\right)(1-e^{-g_1 t})\right)}$
Logistic	$L_t = \frac{L_\infty \times L_0\left(e^{(g_2 t)}\right)}{L_\infty \times L_0(e^{(g_2 t-1)})}$

3.2.5 Maturity

To overcome the under sampling of larger, older animals, Beverton–Holt life-history invariants (BH–LHI) were used to estimate age-at-maturity (t_{50}) and length-at-maturity (L_{50}). Life-history ratios described by Jensen (1996) and Frisk *et al.* (2001) were used to estimate t_{50} and L_{50} using natural mortality (M) and the previously defined k *and* L_∞ ($L_{50}/L_\infty = 0.66$, $\ln(M) = 0.42 \times \ln(k) - 0.83$ and $M \times t_{50} = 1.65$).

3.3 Results

Overall, 214 *A. rostrata* were collected to assess growth: 132 were collected by the crew of the *San Antone II* and 72 were collected as part of the PTS experiments conducted by Campbell *et al.* (2018). The animals caught during the PTS experiments had significantly smaller TL than those caught on the *San Antone II* ($t = -4.180$, d.f. = 166.7, $P < 0.001$). Two animals were excluded from the analysis, as age could not be determined from the respective vertebral centra. Of the 212 animals assessed for growth, 102 were female with a mean TL of 403 mm (S.E. = 11.00, range = 192 − 753) and 110 were male with a mean TL of 413 mm (S.E. = 11.16, range = 193 − 671). No significant difference in size was detected between sexes ($t = -0.670$, d.f. = 209.9, $P = 0.504$) (Figure 1).

3.3.1 Ageing

Generally, ageing between readers was consistent (PA = 82.67%, ACV = 4.21, APE = 2.97) with the age bias plot revealing little variation from the 1:1 line of equivalence (Figure 12). Further, Bowker's test of symmetry showed no between-reader bias (χ^2 = 13.93, d.f. = 12, P = 0.305). The nominal age (i.e., the number of complete band pairs) range of males and females was 0–15 years and 0–17 years, respectively. The oldest female was 750 mm TL and the two males assigned the nominal age of 15 years were 624 and 648 mm TL.

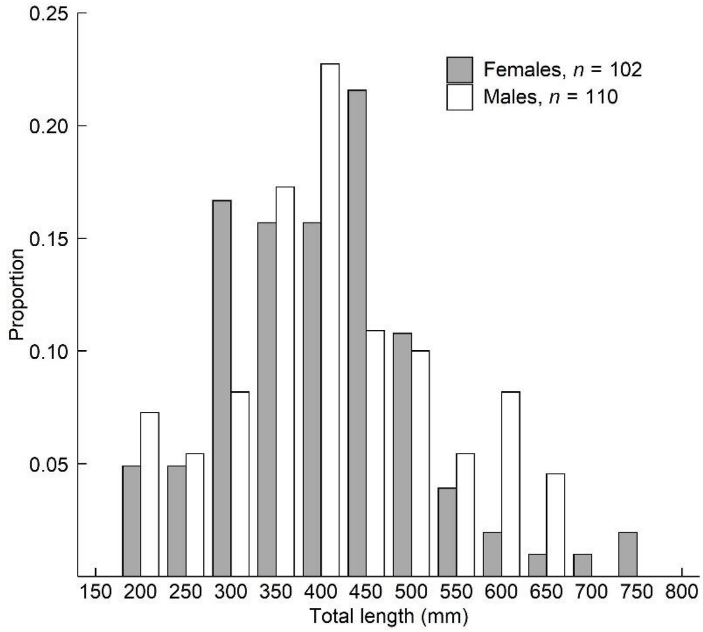

Figure 1: Length–frequency (TL, cm) distribution for 212 *Aptychotrema rostrata*, caught in southeast Queensland, Australia, between April 2016 and November 2017, as a function of sex.

3.3.2 Marginal increment ratio

Marginal increment ratio was lowest during August and September (Figure 13). New edges were also most likely during these months. The Kruskal–Wallis test on ranks indicated MIR varied significantly between months (χ^2 = 23.927, d.f. = 4, P < 0.001). Mean MIR decreased from March through to August, before increasing in September. The highest mean MIR occurred in November, at the end of the austral spring.

Wide edges were also most likely to occur in November; however, the frequency of each edge type was not dependent on month (χ^2 = 12.67, d.f. = 8, P = 0.124).

3.3.3 Growth

The relationship between TL and CR was best described by the quadratic-modified Dahl–Lea method ($TL = -0.038CR^2 + 12.312CR - 17.01$, R^2 = 0.964). Back calculated and observed lengths-at-age were not significantly different (Table 10). As such, the observed and back calculated data were combined, resulting in a dataset containing 1112 measures of length-at-age.

The VBGF was found to best fit the length-at-age data (Table 2, LOOICw = 1). There was no support for either the Gompertz (LOOICw = 0) or the Logistic (LOOICw = 0) growth functions. With the sexes combined, the estimated VBGF parameters were $L_\infty = 923$ mm TL, $L_0 = 193$ mm TL and $k = 0.08$ year^{-1} (Table 2, Figure 2, Figure 14). Estimates of L_∞ and L_0 were higher for females (1141 and 193 mm, respectively), compared to males (813 and 187 mm, respectively) (Figure 3). The growth coefficient for females ($k = 0.05$ year^{-1}) was half that of males ($k = 0.10$ year^{-1}).

Table 2: Relative performance and mean parameter estimates for the three candidate growth functions used to assess the growth of 212 *Aptychotrema rostrata* caught in southeast Queensland, Australia, between April 2016 and November 2017. After back calculation, a total of 1112 length-at-age measures were assessed. The parameter estimates shown are the mean values of the posterior distributions of the respective parameters generated by the 'BayesGrowth' package via R statistical software. Note: LOOIC is the leave-one-out-information-criterion; LOOICw is the LOOIC weights; L_∞ is the asymptotic length; L_0 is the length at $t = 0$; k and g are the growth coefficients of the von Bertalanffy, Gompertz and Logistic functions (see Table 1); and σ is the estimated residual error. Numbers in parentheses are the 95% credible interval of the respective parameters from their posterior distributions.

Function	LOOIC	LOOICw	L_∞ (mm)	L_0 (mm)	k/g (year^{-1})	σ
von Bertalanffy						
All	11428.5	1	923 (843–953)	193 (185– 200)	0.08 (0.06–0.09)	40.8 (39.2–42.6)
Female			1141 (1047–1175)	190 (183–197)	0.05 (0.05–0.06)	40.0 (37.7–42.6)
Male			813 (724–934)	187 (175–199)	0.10 (0.07−0.12)	40.6 (38.2–42.2)
Gompertz						
All	11442.9	0	726 (691–766)	202 (195–208)	0.17 (0.16−0.19)	41.0 (39.3–42.7)
Female			985 (895–1083)	210 (203–217)	0.11 (0.10−0.13)	40.9 (38.5–43.5)
Male			652 (622–687)	192 (183–201)	0.22 (0.19–0.24)	40.2 (37.9–42.6)
Logistic						
All	11452.3	0	666 (643–692)	210 (204–215)	0.27 (0.25–0.28)	41.4 (39.8–43.2)
Female			853 (786–931)	218 (211–226)	0.20 (0.18−0.22)	41.4 (39.0–44.1
Male			631 (618–648)	207 (202–212)	0.29 (0.28–0.30)	40.6 (38.3–41.3)

3.3.4 Maturity

Of the 212 *A. rostrata* used to assess growth, only nine females and nine males were sexually mature. The oldest immature animals (female and male) were >10 years in age, while the youngest mature animals were 6 years. Using the BH–LHI described by Jensen (1996) and Frisk *et al.* (2001), age-at-maturity for both sexes combined was 10.9 years, and 13.3 and 10.0 years for females and males, respectively. Further, length-at-maturity was 609 mm TL for both sexes combined, and 753 and 555 mm TL for females and males, respectively.

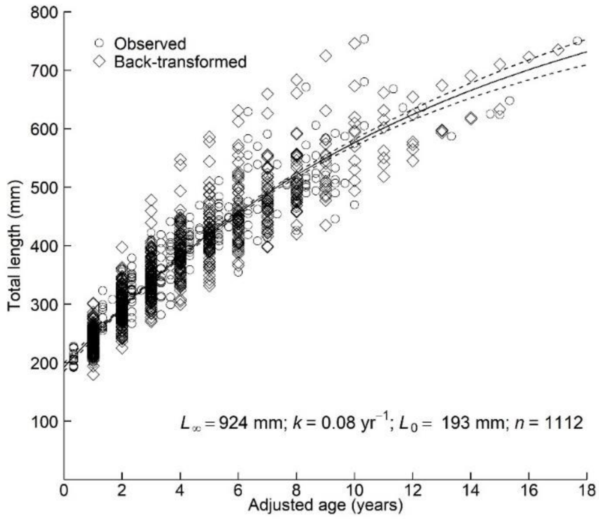

Figure 2: von Bertalanffy growth curve for 212 *Aptychotrema rostrata* caught in southeast Queensland, Australia, between April 2016 and November 2017. Shown are both the observed and back calculated lengths-at-age which resulted in 1112 measures of length-at-age. Priors were set at $L_\infty \sim N(1200, 50)$ and $L_0 \sim N(140, 10)$. Dashed lines represent 95% credible intervals.

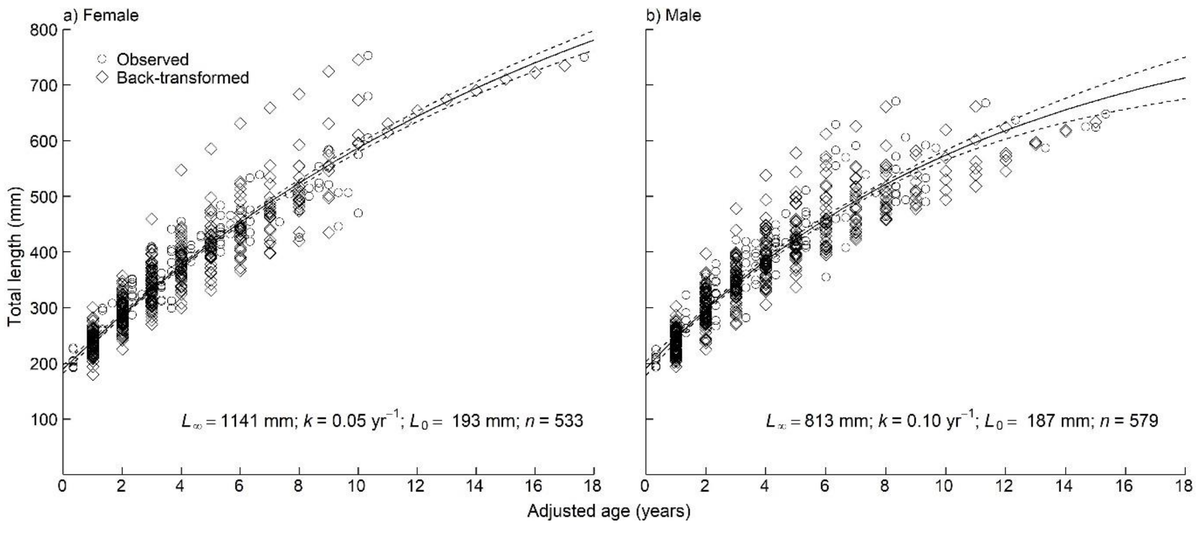

Figure 3: von Bertalanffy growth curve for a) female and b) male *Aptychotrema rostrata* caught in southeast Queensland, Australia, between April 2016 and November 2017. Shown are both the observed

and back calculated lengths-at-age which resulted in 533 and 579 measures of length-at-age for females and males, respectively. Priors were set at $L_\infty \sim N(1200, 50)$ and $L_0 \sim N(140, 10)$ for both sexes. Dashed lines represent 95% credible intervals.

3.4 Discussion

The results from the current study represent the first estimates of growth and age-at-maturity published in the primary literature for *A. rostrata.* Slow growth and late maturity is common among elasmobranchs (Dulvy *et al.*, 2008) making this group vulnerable to overexploitation (Stevens *et al.*, 2000). These characteristics, combined with intense fishing pressure, have resulted in increasing concern for Rhinopristiformes, many of which are at extremely high risk of extinction (Kyne *et al.*, 2020). The landings and catch rates of Rhinopristiformes have declined by up to 80% throughout most of their ranges (D'Alberto *et al.*, 2019); however, a combination of reduced fishing pressure, prohibiting the retention of shark products and networks of marine protected areas have been shown to mitigate risk for this group (Kyne *et al.*, 2020). This is especially the case for *A. rostrata*, which is considered abundant due to its diverse habitat use and the extent of refuges across its range (Kyne and Stevens, 2015). Significant reduction in shrimp trawl effort since 2000 (Wang *et al.*, 2020) is also likely to have had a positive effect on the species' abundance in Queensland.

These factors ensure the continued high levels of abundance in Queensland despite this species' low productivity. Delayed maturity and small maximum size implies a low maximum intrinsic population growth rate (r_{max}) in Rhinopristiformes (D'Alberto *et al.*, 2019). These authors evaluated population productivity in nine Rhinopristiformes and concluded the trygonorrhinids exhibit low r_{max} values compared to larger species such as the giant shovelnose ray (*Glaucostegus typus*) and bottlenose wedgefish (*Rhynchobatus australiae*), both of which co-occur with *A. rostrata*. This is due to the ability of these larger species to produce numerous and large offspring. Age-at-maturity was also found to be negatively correlated with productivity and the t_{50} derived for *A. rostrata* using the BH–LHI is higher than the estimates for all nine species assessed by D'Alberto *et al.* (2019).

Maximum intrinsic population growth rate was calculated for only nine species of Rhinopristiformes due to the lack of reliable life history information. Only three of the eight species that comprise Trygonorrhinidae have published growth information: southern fiddler ray (*Trygonorrhina dumerilii*), shortnose guitarfish (*Zapteryx brevirostris*) and banded guitarfish (*Zapteryx exasperata*). The VBGF growth coefficient derived in the current study, $k = 0.08$ year^{-1}, is lower than any of those published for Trygonorrhinidae. Values of k have been published for *T. dumerilii* and *Z. brevirostris* at 0.13 year^{-1} (Izzo and Gillanders, 2008) and 0.12 year^{-1} (Carmo *et al.*, 2018), respectively. Cervantes–Gutiérrez *et al.* (2018) reported a higher growth coefficient for male and female *Z. exasperata* of $k = 0.174$ year^{-1} and $k = 0.144$ year^{-1}, respectively. Additionally, Caltabellotta *et al.* (2019) reported faster growth in the smaller *Z. brevirostris* of $k = 0.24$ year^{-1}.

Only *T. dumerilii* has a higher published estimate of L_∞ than that presented here. Izzo and Gillanders (2008) reported an L_∞ for *T. dumerilii* of 1129 mm TL for females and males combined. The L_∞ (1157 mm TL) for female *T. dumerilii* is similar to that derived in the current study for female *A. rostrata* (1141 mm TL), despite *T. dumerilii* reaching a higher maximum size (1460 mm TL, Last *et al.*, 2016). Cervantes–Gutiérrez *et al.* (2018) reported estimates of L_∞ for female and male *Z. exasperata* of 1007 mm and 898 mm, respectively. In contrast, Caltabellotta *et al.* (2019) reported an L_∞ of 624 mm and 602 mm for female and male *Z. brevirostris*, while Carmo *et al.* (2018) derived smaller values of 56.0 cm and 50.4 cm, respectively. In accord with other elasmobranchs, the published estimates of L_∞ for the trygonorrhinids was higher for females: Cortés (2000) found that the maximum size of males was on average approximately 10% smaller than females in 164 shark species.

Kyne *et al.* (2016) reported an L_{50} for male *A. rostrata* of 597·3 mm. This is comparable to the present study: however, their L_{50} for females (639·5 mm) is lower than the L_{50} reported here. This difference in L_{50} may be a result of using the BH–LHIs calculated using the estimates of k and L_∞ derived here. The under sampling of large, mature females resulted in biased estimates of female L_{50}, necessitating the use of the BH–LHI. This under sampling may have been due to the selectivity of the sampling gears used in the respective studies. Kyne *et al.* (2016) conducted sampling using shrimp (*Melicertus plebejus*) trawls in water depths to 100 m. In contrast, present study samples were predominantly (62%) collected on the *San Antone II*, which deploys Danish seine gear to target *S. robusta* in water <50 m. The Danish seine used in this fishery is characterised by slow haul speeds and short haul times (Rowsell and Davies, 2012) which may allow larger *A. rostrata* to escape capture.

The difference in water depth is unlikely to be the cause of the under sampling of large animals. Kyne and Bennett (2002b) collected *A. rostrata* from Moreton Bay, adjacent to the grounds in the current study, and reported 41 of 48 (~85%) females sampled were mature. These authors used rod-and-reel in water depths of 3–10 m and reported a similar female L_{50} to Kyne *et al.* (2016). The number of mature females was higher than from the current study: only nine of the 102 females caught aboard the *San Antone II* were mature and no mature animals were collected during the PTS experiments conducted by Campbell *et al.* (2018).

Sexual bimaturism is a common life history strategy among viviparous elasmobranchs (Colonello *et al.*, 2020) due to less investment by females in growth to compensate for attaining a larger size to support pups (Cortés, 2000). The higher L_{50} and t_{50} derived here for female *A. rostrata* is consistent with Kyne *et al.* (2016), who estimated a higher L_{50} for female *A. rostrata*. Similarly, Jones *et al.* (2010) reported a higher L_{50} for females for the congeneric *A. vincentiana* caught in southern Western Australia. Delayed female maturity has been reported for the confamilial banded guitarfish (*Zapteryx exasperate*) caught in Mexico (Cervantes–Gutiérrez *et al.*, 2018).

The MIR analysis suggests that band pair formation occurs annually. Ideally, sampling should have occurred throughout the year to ensure a complete analysis of the periodicity of band pair formation; however, the seasonal nature of fisheries that catch *A. rostrata* as bycatch resulted in irregular access to samples during the study period. Similarly, Cervantes–Gutiérrez *et al.* (2018) reported that fishery closures hampered year-round sampling of *Z. exasperata* in Mexico and published incomplete measures of marginal increments; however, these authors assumed annual band pair formation when quantifying growth. Caltabellotta *et al.* (2019) suggested annual band formation for *Z. brevirostris* using MIR and, like previous studies on the growth of trygonorrhinids (Carmo *et al.*, 2018; Izzo and Gillanders, 2008), this assumption is also reasonable for *A. rostrata.* The minimum MIR occurred in August indicating a period of slow somatic growth coinciding with minimum monthly mean sea surface temperature (Meynecke and Lee, 2011) and high reproductive activity (Kyne *et al.*, 2016). The chi–squared test conducted on the edge frequency data somewhat contradicts the results from the MIR analysis and, as such, further sampling should be undertaken throughout the year to confirm band pair formation occurs annually.

The back calculated lengths-at-age were not significantly different to the observed values. However, the mean back calculated lengths were higher than the observed mean lengths for ages 0 to 6 years due to the inclusion of those vertebral centra where intermediate edges were observed. The low number of vertebral centra with new edges at each age necessitated the inclusion of the centra with intermediate edges for robust comparison between back calculated and observed lengths-at-age.

The under sampling of older animals resulted in biased growth parameter estimates. However, estimating the VBGF parameters in a Bayesian framework allowed for the use of informed priors to estimate L_∞, overcoming the lack of larger animals sampled. Similarly, back calculation increases the number of measures of length-at-age for smaller size classes resulting in improved growth parameter estimates for elasmobranchs where age data are sparse for smaller individuals (e.g. Carmo *et al.*, 2018; D'Alberto *et al.*, 2017; Smart *et al.*, 2013). These two techniques allowed for the estimation of reasonable growth parameters for use in assessing the population status of *A. rostrata*.

Various methods correlate k and L_∞ to natural mortality (M, e.g. Frisk *et al.*, 2001; Pauly, 1980; Then *et al.*, 2015). As such, biased estimates of growth result in biased estimates of M (D'Alberto *et al.*, 2019), leading to inaccurate assessments of stock status (Pardo *et al.*, 2013). Estimating growth parameters in a Bayesian framework overcomes this bias (Smart and Grammer, 2021). Campbell *et al.* (2017) quantified the ecological risk posed to *A. rostrata* by the ECOTF using the sustainability assessment for fishing effects (SAFE) quantitative ERA developed by (Zhou *et al.*, 2009). In this instance, risk was quantified via comparison between the level of fishing mortality (F) and the maximum sustainable fishing mortality (F_{msm}), where $F_{msm} = 0.41M$ (Zhou *et al.*, 2012). Hence, the growth parameter estimates derived here allow for the calculation of unbiased estimates *M* and F_{msm}, which enable the accurate assessment of the population status.

Campbell *et al.* (2017) assessed the risk posed to 47 elasmobranchs, only 18 of which had published growth estimates. Of these 18, growth was quantified for seven species based on samples collected within the study area. This reinforces the need for basic life-history data to inform fishery impacts in batoids (Kyne, 2016) and elasmobranchs in general. In the absence of life history information, previous studies (Zhou *et al.*, 2015; Zhou *et al.*, 2013) have used the "Life History Tool" on the Fishbase website (www.fishbase.se) to determine values for *M*. There is a need, therefore, to increase knowledge of life-history information to ensure the accurate assessment of fishery impacts on elasmobranchs with sparse catch data.

In conclusion, the current study contributes to the scientific knowledge of *A. rostrata* and Rhinopristiformes more broadly. Consistent with other elasmobranchs, *A. rostrata* exhibit slow growth, late maturity and a long lifespan. Despite this, the species is abundant in Queensland due to its diverse habitat use and the extent of refuges throughout its range. This result contrasts with other Rhinopristiformes, many of which are at high risk of extinction. The life history characteristics derived from this research can be used in future studies to determine population status and inform management decisions.

4. Factors affecting elasmobranch escape from turtle excluder devices (TEDs) in a tropical penaeid-trawl fishery

Plate 4: Wicks TED on the FV *Markina* in the Gulf of Carpentaria, September 1998

Available online 12 December 2019 as: Campbell, M.J., Tonks, M.L., Miller, M., Brewer, D.T., Courtney, A.J., and Simpfendorfer, C.A. (2020) Factors affecting elasmobranch escape from turtle excluder devices (TEDs) in a tropical penaeid-trawl fishery. Fisheries Research 224, 105456. https://doi.org/10.1016/j.fishres.2019.105456

4.1 Introduction

Tropical penaeid trawling is recognised as a poorly selective form of fishing (Griffiths *et al.*, 2006). Penaeids cohabit in demersal environments with various species that are susceptible to capture by trawls (Andrew and Pepperell, 1992) resulting in the highest discard rates of 25 gear types assessed by Perez Roda *et al.* (2019) at 54.9%. The discarded portion of penaeid-trawl catches comprises hundreds of species (Courtney *et al.*, 2006; Stobutzki *et al.*, 2001b) some of which have substantial conservation interest such as sea turtles and sawfish (Brewer *et al.*, 1998; Robins–Troeger *et al.*, 1995; Watson and Seidel, 1980). Concerns regarding the impacts of discarding unwanted animals on ecosystems are also recognised globally (Broadhurst, 2000; James *et al.*, 2015). Consequently, quantifying and mitigating discards have been the subjects of significant research efforts since the early 1990s (Broadhurst *et al.*, 2006; Kelleher, 2005).

The introduction of turtle excluder devices (TEDs) in tropical penaeid-trawl fisheries has led to beneficial flow-on effects (Jordan *et al.*, 2013) including significant reductions in the capture of large elasmobranchs (e.g. Brewer *et al.*, 1998; Robins–Troeger *et al.*, 1995; Willems *et al.*, 2016). Elasmobranchs (i.e., i.e., sharks and rays) are one component of penaeid-trawl discards that have received increasing attention in the last two decades (Dulvy *et al.*, 2017). A major driver for this work is that elasmobranch life histories include late maturity, few offspring, long life spans and slow growth (Dulvy *et al.*, 2008; James *et al.*, 2015) making them vulnerable to overexploitation (Ellis *et al.*, 2008).

It has been estimated that 25% of elasmobranchs are threatened with an elevated risk of extinction due to, for the most part, capture in marine fisheries, either as target species or by fishing gears targeting other species (Dulvy *et al.*, 2014; Simpfendorfer and Dulvy, 2017). Research has shown that elasmobranchs caught by penaeid trawls are predominantly batoids and small demersal sharks (Courtney *et al.*, 2006; Ellis *et al.*, 2017; Robins and McGilvray, 1999; Shepherd and Myers, 2005; Stobutzki *et al.*, 2002), although larger pelagic sharks (e.g. carcharhinids) are caught by larger, and/or fast moving penaeid and fish trawls (e.g. Brewer *et al.*, 2006; Jaiteh *et al.*, 2014; Raborn *et al.*, 2012; Wakefield *et al.*, 2016).

There are relatively few studies detailing the effects of TEDs and other bycatch reduction devices (BRDs) on the catch of elasmobranchs in the primary literature (some examples are: Brewer *et al.*, 2006; Brewer *et al.*, 1998; Fennessy and Isaksen, 2007; Jaiteh *et al.*, 2014; Noell *et al.*, 2018; Wakefield *et al.*, 2016). As these devices were adopted in penaeid-trawl fisheries, their effects on target catch and discards were a focus of research (see review by Broadhurst, 2000). Numerous studies from the 1990s reported the effects of TEDs and BRDs on penaeid and discard catches (e.g. Broadhurst *et al.*, 1997; Isaksen *et al.*, 1992), while others also confirmed the exclusion of turtles (Brewer *et al.*, 1998; McGilvray *et al.*, 1999; Robins–Troeger, 1994; Robins–Troeger *et al.*, 1995). Most studies during the 1990s were conducted on known trawl grounds in an effort to replicate commercial conditions

(Broadhurst *et al.*, 1997; Robins–Troeger, 1994; Robins and McGilvray, 1999), resulting in sufficient quantities of both target species and bycatch to enable robust analyses from a relatively small number of trawls. However, given interactions with penaeid trawls are relatively rare for most species caught (Kyne *et al.*, 2002; Wakefield *et al.*, 2016), analyses regarding the effect of TEDs and BRDs on elasmobranchs were largely absent.

The lack of detailed information in the primary literature describing the effects of TEDs on elasmobranchs warrants attention. Although the composition of elasmobranch bycatch caught by penaeid trawlers is poorly understood (e.g. Molina and Cooke, 2012), previous research has shown it includes species groups of conservation value. Hammerhead sharks (Sphyrnidae: Raborn *et al.*, 2012; Wakefield *et al.*, 2016), sawfish (Pristidae: Brewer *et al.*, 2006; Wakefield *et al.*, 2016), guitarfish (Glaucostegidae: Garcıa–Caudillo *et al.*, 2000; Robins–Troeger, 1994), wedgefish (Rhinidae: Brewer *et al.*, 2006; Fennessy, 1994; Robins and McGilvray, 1999) and skates (Rajidae: Kyne *et al.*, 2002) have all been shown to occur in penaeid-trawl bycatch.

The objective of the current study was to quantify the impact of TEDs on the catches of various elasmobranchs caught off northern Australia using data collected during a previous study (Brewer *et al.*, 2006). The effect of fish size and various aspects of TED design such as grid orientation, grid angle and bar space were quantified to determine their effect on the escape of elasmobranchs from penaeid trawls. For the purposes of the current study, a TED was considered to be a barrier installed in a trawl designed to exclude any component of the discarded portion of a catch. Further, all care has been taken to provide updated species names when discussing previous studies.

4.2 Materials and methods

In the current study, data collected by Brewer *et al.* (2006) were re-analysed to determine factors affecting the escape of elasmobranchs from penaeid trawls. These data were recorded by scientific observers with the objectives of providing information regarding the impact of TEDs and BRDs on target and non-target catches within the NPF. Five observers collected data on board 23 vessels while fishing commercially during the tiger (*Penaeus semisulcatus* and *P. esculentus*) prawn season (August to November) of 2001. At this time, fishers primarily targeted tiger and endeavour (*Metapenaeus endeavouri* and *M. ensis*) prawns at night using one Florida Flyer (>~18 m or 10 fathoms headline length) net towed from each side of the vessel. Fishers were required to have a TED and one of seven prescribed BRDs installed in each net. Of the seven prescribed BRDs, the bigeye and square-mesh panels (see Brewer *et al.*, 2006 for illustrations) were the most popular during the sampling period. Observers spent approximately two weeks on board a vessel before moving to the next vessel (hereafter referred to as a ‘trip’). The two-week period was chosen as it approximated the time between visits to a refuelling barge: the barges anchored in calm inshore waters which facilitated the easy transfer of observers between vessels and negated the need to perform potentially dangerous transfers on the fishing grounds.

4.2.1 Sampling protocol

Once an observer boarded a vessel, the gear was left unaltered for one night to quantify between-net variation in catch, termed a 'calibration night'. On the second night, the TED and BRD were removed from one net, chosen by the master of the vessel, resulting in a 'control' net and a 'treatment' net being towed simultaneously. After seven nights, the BRD in the treatment net was either removed or made ineffective by sewing trawl mesh over the escape opening, thereby providing information on the effects of the TED only. On the last night of sampling (typically night 14), a second calibration night was undertaken to ensure any between-net variation detected on the first night was consistent throughout the sampling period.

Various design aspects of the TEDs used were recorded by the observer at the start of each trip. Important information including grid size, orientation (top-shooter or bottom-shooter), angle, bar spacing and dimensions of the escape hole were documented in order to determine their effects, if any, on catches. The BRDs tested had no effects on the catch rates of elasmobranchs and, as such, we focus only on the effects of TEDs, used either in combination with a BRD, or individually.

During the sampling period, vessels completed up to four trawls per night. Each trawl was 3–4 h in duration depending on the amount of bycatch present. At the end of each trawl, the two codends were spilled onto the sorting tray ensuring the catches from each net were separated. All large animals (attaining >30 cm in length) such as sea turtles, elasmobranchs, sponges and sea snakes, were removed from the catch and, where possible, identified to species, weighed, measured and released alive. The crew then commenced sorting, by removing all commercial penaeids and byproduct, including squid (Teuthoidea), Moreton Bay bugs (Scyllaridae: *Thenus australiensis* and *T. parindicus*) and scallops (Pectinidae: *Amusium pleuronectes*), for immediate processing and storage in on-board freezers. Generally, *P. semisulcatus* and *P. esculentus* ≥26 mm carapace length (CL) were retained, while *T. australiensis* and *T. parindicus* had a minimum legal size of 60 mm carapace width. All remaining bycatch was sorted into lug baskets and weighed by the observer.

4.2.2 Statistical analyses

For the purposes of the present study, a subset of the data obtained by Brewer *et al.* (2006), containing only those trawls where an elasmobranch was caught in either the control net or treatment net, was used. In accord with Brewer *et al.* (2006), the effect of TEDs on the elasmobranch catch was initially assessed using the exact binomial test. That is, the probability of an elasmobranch being caught in the treatment net is p and $1 - p$ for capture in a control net with a null hypothesis of $p = 0.5$. Retention of the null hypothesis implies that the TED failed to exclude elasmobranchs. The exact binomial test was performed using R statistical software (Version 3.3.3, R Foundation for Statistical Computing, Vienna, Austria, see https://www.R-project.org/, accessed 19 April 2018) via the "binom.test" function from the "stats" package.

In order to provide information on the factors affecting the escape of elasmobranchs via TEDs, the data were analysed using a logistic regression model of length data which relates the probability of capture in the treatment net to the size of the animal (Brewer *et al.*, 2006). Assuming each net was equally likely to catch an elasmobranch, the expected number caught in a control net is n and $n(1 - e)$ in the treatment net, where e is the rate of escape due to the presence of the TED. Therefore, the probability of capture in the treatment net, t, is $t = n(1 - e)/(n + n(1 - e))$.

Where sample size permitted, t was estimated for each species, family and order. This probability was estimated via generalised linear mixed modelling using R statistical software via the 'glmer' function within the 'lme4' package (Bates *et al.*, 2015). The probability of capture in the treatment net of each species or species group was modelled separately and datasets were restricted to cases where all relevant data were present. A vessel identifier was added as a random term while grid orientation (top-shooter or bottom-shooter), grid shape (circular, elliptical, tombstone or rectangular), the presence of a BRD (0, 1), the presence of bent deflector bars (0, 1) and the presence of an escape-hole cover (0, 1) were added as categorical fixed terms. Additionally, grid angle, bar space, area, escape hole area and fish size (TL for all sharks and Rhinopristiformes, DW for all other rays) were added as covariates. Given their importance in the results reported by Brewer *et al.* (2006), fish size and grid orientation were added to all models. All other categorical terms and covariates were tested for significance and retained in the model only if their addition improved the Akaike Information Criteria (AIC). Relevant two-way interactions were also tested and excluded if their addition had no significant effect on the probability of capture in the treatment net. The 'bootMER' function within the 'lme4' package was used to calculate 95% confidence intervals around the estimated probabilities.

Following Brewer *et al.* (2006), the probability of capture in the treatment net, t, was then converted to escape, e, with the equation $e = 1 - t(1 - t)$. A simple function in R converted the vectors of estimated probabilities and the associated confidence intervals to escape rates.

Preliminary analysis revealed that treatment nets caught more smaller elasmobranchs than control nets. This resulted in negative values of escape (i.e., $e < 0$). As such, the size at which escape and retention were equal (i.e., the size at which escape was zero, S_0) was calculated for each taxonomic order. This metric provided additional information on the effects of TED design on escape. The size at which 50% escape (S_{50}) occurred was also calculated.

To ensure the nets were fishing similarly before and after each sampling period, the number of elasmobranchs caught during the calibration nights were analysed using generalised linear mixed modelling. For this analysis, a vessel identifier was added as a random term while trawl number, calibration period (0 = before sampling, 1 = after sampling) and vessel side were added as fixed effects. The number of elasmobranchs in each net was the variable of interest which was modelled as a Poisson distribution using R.

4.3 Results

During the sampling period, 720 trawls were undertaken on 22 vessels where a treatment net and a control net were towed simultaneously (i.e., 1440 net trawls). Results from one vessel were excluded due to the limited number of trawls conducted with treatment and control nets present. Various TED designs were used during the sampling period (Table 11). Most devices were deployed as bottom-shooters: 430 trawls were undertaken with bottom-shooter TEDs; and 290 as top-shooters. Only two devices were tested in both top- and bottom-shooting configurations. TEDs were generally tombstone-shaped, rectangular or elliptical with a grid angle ranging between 40 and 72° from the horizontal. Bar space ranged between 95 and 120 mm, with 32% of trawls completed using TEDs with the maximum permitted bar space of 120 mm. The majority of trawls were conducted with guiding panels installed (~79%) and no deflector bars (~90%). A BRD was installed during 427 (~59%) trawls.

Generalised linear mixed modelling revealed there was no significant between-net variation in the catches of elasmobranchs (all species) at each location during the calibration phase of each sampling trip ($P = 0.817$). This indicated that, for all vessels, any variability in catch between nets could not be attributed to the nets themselves.

From the 1440 net trawls, a total of 6204 elasmobranchs were identified representing 34 species from 27 genera, 15 families and four orders (Table 12). The most common species caught was the whitecheek shark (*Carcharhinus coatesi*, $n = 1218$), while Australian blacktip sharks (*Carcharhinus tilstoni*, $n = 634$), brown whiprays (*Maculabatis toshi*, $n = 634$), painted maskrays (*Neotrygon leylandi*, $n = 627$), Australian butterfly rays (*Gymnura australis*, $n = 641$) and bottlenose wedgefish (*Rhynchobatus australiae*, $n = 571$) occurred frequently in catches. In contrast, ten or fewer individuals were caught for 15 of the remaining 28 (~54%) species identified during sampling (Table 12).

Generally, the most abundant species were small (Table 12). Median TL of the most common sharks (*C. coatesi*, *C. tilstoni*, *Rhizoprionodon acutus* and *Chiloscyllium punctatum*) was ≤81 cm, while the median DW of the most common rays (*Maculabatis toshi*, *Neotrygon annotata*, *N. leylandi* and *Gymnura australis*) was ≤44 cm. Further, the median TL of the most common Rhinopristiform, *R. australiae*, was 65 cm. However, small numbers of large (≥3.0 m TL) *S. lewini*, tiger sharks (*Galeocerdo cuvier*) and *R. australiae* were caught in control nets.

4.3.1 Factors affecting escape

It should be noted that, for the most part, the low number of elasmobranchs encountered during the sampling resulted in a lack of power to isolate the effects of the various factors tested on the catches of elasmobranchs. The observational nature of the data, combined with the lack of control over the design factors of the TEDs tested (Table 11), caused some issues when analysing these data. For example, only 20 trawls were undertaken with a circular TED compared to 234 and 188 trawls with elliptical and rectangular grids, respectively.

Table 3: Beta parameters from the generalised linear mixed models testing the effects of fish size (cm, total length for all except the Myliobatiformes which are measured by disc width), TED grid orientation (bottom-shooter TED is the reference level) and bar spacing on the probability of capture in treatment nets undertaken for each species or species group. Numbers in parentheses represent the standard error around the beta parameter estimate. *** $P < 0.001$; ** $P < 0.01$; * $P < 0.05$; and ns $P > 0.05$.

Species or species group	Size	Grid orientation	Bar spacing
Carcharhiniformes	–0.013 (0.002)***	–0.214 (0.101)*	ns
Carcharhinidae	–0.014 (0.003)***	–0.259 (0.099)**	ns
Carcharhinus tilstoni	–0.031 (0.007)***	–0.360 (0.174)*	0.022 (0.010)*
Sphyrnidae	–0.028 (0.012)*	–2.281 (1.038)*	ns
Orectolobiformes	–0.011 (0.002)***	ns	ns
Myliobatiformes	–0.018 (0.002)***	0.216 (0.105)*	ns
Dasyatidae	–0.024 (0.002)***	0.237 (0.117)*	ns
Maculabatis toshi	–0.019 (0.006)**	ns	ns
Myliobatidae	–0.032 (0.013)*	ns	ns
Rhinopristiformes	–0.012 (0.002)***	ns	ns
Rhynchobatus australiae	–0.014 (0.003)***	ns	ns

The GLMMs indicated fish size significantly affected the probability of capture in treatment nets and, therefore, escape for only three species (Table 3): *C. tilstoni* (β = –0.031, S.E. = 0.007: P < 0.001), *M. toshi* (β = –0.019, S.E. = 0.006: P < 0.001) and *R. australiae* (β = –0.014, S.E. = 0.003: $P < 0.001$). In all instances, increasing size was found to reduce the probability of capture in treatment nets. Further, top-shooter TEDs (β = –0.360, S.E. = 0.174: $P < 0.05$) reduced the probability of capture of *C. tilstoni*, while increasing bar space (β = 0.022, S.E. = 0.010: $P < 0.05$) had the opposite effect. For the remaining species, the respective GLMMs failed to attribute differences in the probability of capture in treatment nets to any factor or covariate tested.

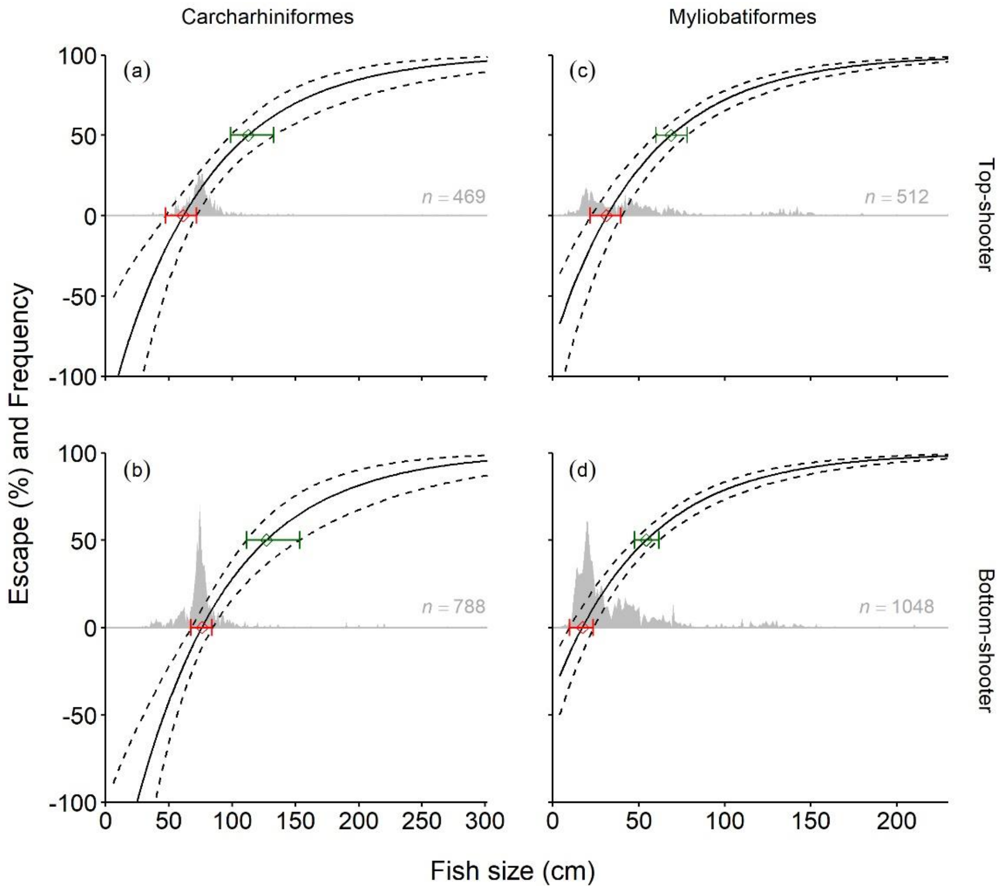

Figure 4: Escape of Carcharhiniformes (a, b) and Myliobatiformes (c, d) as a function of fish size (total length for Carcharhiniformes and disc width for Myliobatiformes) and turtle excluder device (TED) grid orientation. Dashed lines represent 95% confidence intervals. Also shown are the length frequencies of the respective species groups, as a function of grid orientation, caught in control nets (i.e., no TEDs) only. Sample sizes represent the number of individuals assessed in the respective reduced generalised linear mixed models. The red points show the sizes at which escape and retention were equal (i.e., S_0) and the green points represent the size at which 50% escape (i.e., S_{50}) occurred.

At the family level, increasing fish size significantly reduced the probability of capture in treatment nets (Table 3) for carcharhinids (β = –0.014, S.E. = 0.003: $P < 0.001$), sphyrnids (β = –0.028, S.E. = 0.012: $P < 0.05$), dasyatids (β = –0.024, S.E. = 0.002: $P < 0.001$) and myliobatids (β = –0.032, S.E. = 0.013: $P < 0.05$). Top-shooter TEDs reduced the probability of capture in treatment nets for both carcharhinids (β = –0.259, S.E. = 0.099: $P < 0.01$) and sphyrnids (β = –2.281, S.E. = 1.038: $P < 0.05$). In contrast, the probability of capture in treatment nets was significantly greater (β = 0.237, S.E. = 0.117: $P < 0.05$) for dasyatids when top-shooter TEDs were used.

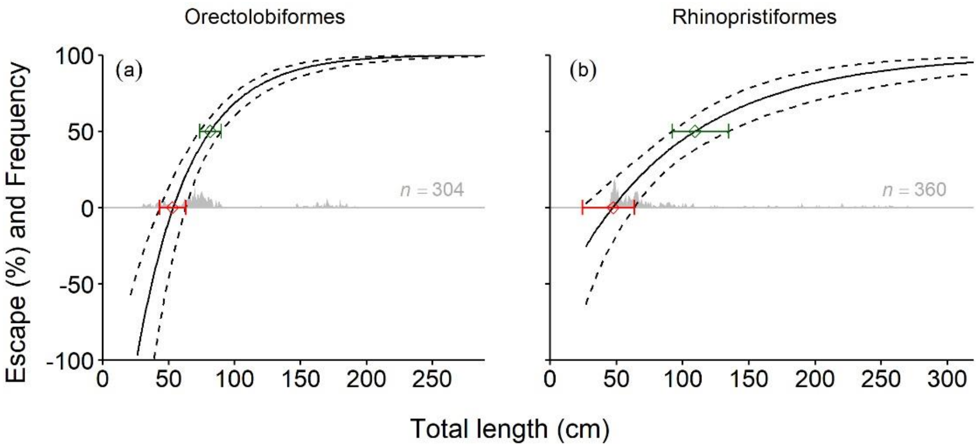

Figure 5: Escape of Orectolobiformes (a) and Rhinopristiformes (b) as a function of total length, in centimetres. Dashed lines represent 95% confidence intervals. Also shown are the length frequencies of the respective species groups caught in control nets (i.e., no TEDs) only. Sample sizes represent the number of individuals assessed in the respective reduced generalised linear mixed models. The red points show the sizes at which escape and retention were equal (i.e., S_0) and the green points represent the size at which 50% escape (i.e., S_{50}) occurred.

At the order level, increasing fish size significantly reduced the probability of capture ($P < 0.05$, Table 3). Top-shooter TEDs caught fewer Carcharhiniformes ($\beta = -0.214$, S.E. = 0.101: $P < 0.05$) and more Myliobatiformes ($\beta = 0.216$, S.E. = 0.105: $P < 0.05$). Grid orientation had no effect on the probability of capture in treatment nets for both Orectolobiformes and Rhinopristiformes ($P > 0.05$).

Escape from TEDs was greatest for large animals (Figure 4 and Figure 5). However, the GLMMs indicated that escape from treatment nets was negative for animals in smaller size classes across all species groups. It was prudent, therefore, to quantify the size at which retention and escape were equal (i.e., S_0): animals below S_0 experienced higher retention by treatment than control nets, while escape occurred for some proportion of those animals larger in size than S_0. The S_0 occurred at 17 and 31 cm DW for Myliobatiformes caught in treatment nets containing bottom- and top-shooter TEDs, respectively (Table 13). In contrast, the estimate of S_0 was lower for Carcharhiniformes caught in top-shooter TEDs (S_0= 61 cm TL) than those caught in bottom-shooter TEDs (S_0 = 76 cm TL). The S_0 was similar for the Orectolobiformes and Rhinopristiformes.

4.4 Discussion

The TEDs used throughout the sampling period facilitated the escape of a high proportion of species of conservation interest. Importantly, the TEDs significantly reduced the number of Endangered *S. lewini* (Baum *et al.*, 2009) and *S. fasciatum* (Dudgeon *et al.*, 2016b). This is the first study to demonstrate that TEDs reduce the catch of these species in penaeid-trawl fisheries. In contrast, the TEDs used throughout the sampling period had no effect on catches of the Endangered (D'Anastasi *et al.*, 2013) narrow sawfish

(*Anoxypristis cuspidata*), although sample size was low (n = 16). Similar to observations by Wakefield (2016), TEDs failed to exclude four narrow sawfish due to entanglement of the rostrum forward of the TED.

Because the bar spacing of a TED dictates what can physically pass through to the codend, it is an important factor influencing the escape of elasmobranchs. In the current study, reducing bar space resulted in significantly fewer *C. tilstoni* caught in treatment nets (Table 3). This was the only species where sufficient individuals were caught at appropriate sizes (37–159 cm TL) to isolate the effects of bar space on escape. This result is consistent with Noell *et al.* (2018), who found reducing bar space from 45 to 35 mm resulted in significantly lower numbers and weights of elasmobranchs with no loss of the targeted western king prawns (*Melicertus latisulcatus*). Similarly, Garstin and Oxenford (2018) reported a 40% reduction in catches of elasmobranchs using a modified TED (4.45 cm bar space) compared to standard TEDs (102 mm bar space) used in the Atlantic seabob (*Xiphopenaeus kroyeri*) fishery in Guyana. These authors reported significant reductions for various batoids including the smooth butterfly ray (*Gymnura micrura*), longnose stingray (*Hypanus guttatus*) and sharpsnout stingray (*Fontitrygon geijskesi*). Further, in a simulation study, Brčić *et al.* (2015) suggested that reducing the bar spacing of a TED from 90 to 70 mm would significantly reduce the number of blackmouth catsharks (*Galeus melastomus*) whilst maintaining the catch rates of the targeted Norway lobster (*Nephrops norvegicus*).

While the use of narrower bar spaces is a logical modification to improve the escape of elasmobranchs from penaeid trawls, the size of the target and other commercially important species determines the appropriate bar space. Acceptable bar spaces have been shown to range between 19 mm, for targeting Pandalus sp. (Hannah *et al.*, 2011; Isaksen *et al.*, 1992), and 150–200 mm for targeting fish in Western Australia (Jaiteh *et al.*, 2014; Wakefield *et al.*, 2016). Assessing the loss of target species is important when quantifying the effects of TEDs in penaeid-trawl fisheries because fishers are likely to resist any modification to a net that reduces their catch (Gullett, 2003). However, fishers may accept small catch losses if this is offset by improved quality (Eayrs, 2007; Noell *et al.*, 2018; Salini *et al.*, 2000). For example, Salini *et al.* (2000) estimated the reduction in damage to *P. semisulcatus* and *P. esculentus*, due to the introduction of TEDs and BRDs in the NPF, would result in increased revenue to the fleet of ~$AU1 million per annum.

In addition to fish size, other morphological characteristics of individual species influence escape. For example, significantly fewer brown whiprays (*Maculabatis toshi*) were caught in treatment nets, while no significant reductions were detected for Australian butterfly rays (*Gymnura australis*), despite broadly similar sizes (Table 12). This result is consistent with Willems *et al.* (2016) who reported greater escape for longnose stingrays (*Hypanus guttata*) than smooth butterfly rays (*Gymnura micrura*). These authors attributed this result to the contrasting morphology of the two species: while *H. guttata* possess a thick, rigid disc, *G. micrura* has a flexible, smooth disc which allows for easy passage through the TED bars and into the codend. Similarly, *M. toshi* is much thicker through the trunk than *G. australis*, which is extremely flattened (Last and Stevens, 2009), making the latter more likely to pass through the bars of a TED and into the codend at similar disc widths.

Grid orientation was the only other factor tested that was found to affect the escape of elasmobranchs in the current study (Table 3 and Figure 4). Top-shooter TEDs facilitated greater escape of Carcharhiniformes while bottom-shooter TEDs improved the escape of Myliobatiformes. These results are likely a function of the escape response, and the resultant position in the trawl, of the respective species groups. Wakefield *et al.* (2016) and Jaiteh *et al.* (2014) observed carcharhinids attempting to exit a fish trawl in an upward direction during field trials using underwater video equipment. Considering these results, the use of top-shooter TEDs in the Raborn *et al.* (2012) study may have resulted in greater escape of *R. terraenovae* in the Gulf of Mexico penaeid-trawl fishery.

Top-shooter TEDs were less effective for Myliobatiformes. This outcome was particularly the case for smaller (S_0 = 31 cm DW) individuals, with 36% of animals in control nets of a size where escape did not occur (Figure 4 and Table 13). This contradicts the previous study by Brewer *et al.* (2006), who used exact binomial tests to demonstrate grid orientation had no effect on the escape of Myliobatiformes and Rhinopristiformes combined (what they refer to as "rays"). However, these species groups were analysed separately here.

The greater escape of small Myliobatiformes from bottom-shooter TEDs may be a result of the location of these animals in the net. Their morphology suggests that the majority of these animals live on the sea floor and are, therefore, more likely to escape in a downward direction. Main and Sangster (1982) found that skates (Rajidae) and spotted dogfish (*Scyliorhinus canicula*) were more likely to be caught in the lower level of a fish trawl net divided by a horizontal separator panel. Escape holes placed in the bottom of the net may allow more animals to escape before passing through the bars of a bottom-shooter TED.

Grid orientation is fishery specific (Eayrs, 2007). In areas where sedentary organisms (e.g. sponges) or slow moving heavy animals (e.g. turtles or rays) are present, bottom-shooter TEDs are more appropriate (Mitchell *et al.*, 1995). In 'cleaner' areas, top-shooter TEDs can be more suitable (Eayrs, 2007). Where large animals are absent from catches and the escape-hole cover remains closed throughout the trawl, top-shooter TEDs are able to maintain penaeid catch compared to control nets (Courtney *et al.*, 2014).

There is scant information in the primary literature regarding the effect of grid orientation on the catches of elasmobranchs in penaeid-trawl fisheries. Two studies (Chosid *et al.*, 2012; Wakefield *et al.*, 2016) discussed the effects of grid orientation on the escape of elasmobranchs in fish trawls and provide some information on this important factor. Chosid *et al.* (2012) attempted to assess the effectiveness of top- and bottom-shooter TEDs on the exclusion of spiny dogfish (*Squalus acanthias*) from fish trawls in

Massachusetts, USA. Their results were inconclusive due to the low number of trawls undertaken. However, retention rates were lowest in nets containing bottom-shooter TEDs.

Wakefield *et al.* (2016) tested the effects of a TED on various Endangered, Threatened and Protected species (ETP) species, including elasmobranchs, in a fish-trawl fishery in Western Australia. These authors assessed behaviour at a TED using underwater cameras. In accordance with the current study, Wakefield *et al.* (2016) reported that a top-shooter TED allowed a significantly greater number of carcharhinids to escape trawls compared to a bottom-shooter TED. These authors also reported that significantly fewer Rhinopristiformes were caught in trawls with top-shooter TEDs, while grid orientation had no effect on Myliobatiformes, Rajidae, Scyliorhinidae or Orectolobiformes.

Similarly, grid orientation had no effect on the escape of Orectolobiformes in the current study. Fewer individuals of the dominant species, *C. punctatum* and *S. fasciatum*, were caught in treatment nets. Given the size of *S. fasciatum*, the only effect grid orientation is likely to have is to increase the speed at which the animals escape the trawl. Wakefield *et al.* (2016) reported that the escape times were lower when bottom-shooter TEDs were used, which was likely to reduce blockages at the grid and any resultant catch loss (McGilvray *et al.*, 1999).

Since the 2001 fishing season, advancements in TED design have occurred to minimise catch loss. The effect of these advancements on the escape of elasmobranchs remains unquantified. For example, double escape-hole covers were developed in the early 2000s (Mitchell, 2006) to allow the escape of leatherback turtles (*Dermochelys coriacea*). The covers are designed to close quickly, due to the extra material used, which prevents the loss of target catch. Further research is required to quantify the effect of this and other modifications on the escape of elasmobranchs.

In the original study, Brewer *et al.* (2006) found that nets with a BRD only reduced the capture of *C. tilstoni* by 23.8% compared to control nets. In contrast, the current study indicates that the BRDs used throughout the sampling period were ineffective for *C. tilstoni*, and all other elasmobranchs, when used in combination with a TED. Since 2001, the bigeye BRD has been removed from the list of approved devices with several others added after at-sea testing revealed their efficacy in reducing bycatch. For example, Raudzens (2007) reported that a modified fisheye BRD reduced bycatch whilst simultaneously maintaining penaeid catches, compared to a control net. Importantly, the device reduced the number of elasmobranchs caught in nets containing the device.

In conclusion, this research has shown that TEDs facilitate the escape of large elasmobranchs including several species of conservation interest. Bar space and orientation are important TED design factors affecting the escape of elasmobranchs. Top-shooter TEDs enable more Carcharhiniformes to escape penaeid trawls while bottom-shooter TEDs increase the escape of Myliobatiformes. However, the bar space that facilitates maximum escape of elasmobranchs, while maintaining the catches of target species, is more difficult to quantify given the relatively low catch rates of elasmobranchs in penaeid trawls. Experiments using TEDs with reduced bar spacing, such as those conducted by Noell *et al.* (2018), should be undertaken to quantify the effect on penaeid loss. This is especially the case for the NPF where the maximum regulated bar space remains at 120 mm. Any loss in target catch is likely to be offset by improved quality resulting from less damage in the codend. Further, the mitigation of the ecological risk posed to elasmobranchs by penaeid trawling, via less bycatch, is a beneficial result of reduced bar space.

5. Post-release survival of two elasmobranchs, the eastern shovelnose ray (*Aptychotrema rostrata*) and the common stingaree (*Trygonoptera testacea*), discarded from a prawn trawl fishery in southern Queensland, Australia

Plate 5: Common stingaree (*Trygonoptera testacea*) at Southport, Queensland, October 2016.

Available online 16 January 2018 as: Campbell, M.J., McLennan, M.F., Courtney, A.J., and Simpfendorfer, C.A. (2018) Post-release survival of two elasmobranchs, the eastern shovelnose ray (*Aptychotrema rostrata*) and the common stingaree (*Trygonoptera testacea*), discarded from a prawn trawl fishery in southern Queensland, Australia. Marine and Freshwater Research 69(4), 551–561. https://doi.org/10.1071/MF17161

5.1 Introduction

Tropical prawn (or shrimp) trawling is recognised as a non-selective form of fishing (Griffiths *et al.*, 2006) and accounts for 27.3% of the world's fisheries discards (Kelleher, 2005). The discarded portion of prawn trawl catches is known to comprise hundreds of species (Courtney *et al.*, 2006; Stobutzki *et al.*, 2001b; Tonks *et al.*, 2008) and includes species of conservation interest such as sea turtles (Brewer *et al.*, 1998; Robins–Troeger *et al.*, 1995; Wallace *et al.*, 2010; Watson and Seidel, 1980). This has led to the introduction of gear modifications to mitigate the interaction of these animals in many prawn trawl fisheries worldwide: turtle excluder devices (TEDs) and bycatch reduction devices (BRDs) have been used to reduce discards in many countries since the 1980s and numerous studies have reported their effects on catch rates of target and bycatch species (for a review, see Broadhurst, 2000).

The introduction of TEDs has likely resulted in a reduction in the number of elasmobranchs retained by prawn trawl gear (Brewer *et al.*, 2006). This component of prawn trawl bycatch 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). Although the introduction of TEDs has gone some way to decreasing the ecological risk posed to large elasmobranchs by prawn trawling (Brewer *et al.*, 1998; Fennessy, 1994; Gorman and Dixon, 2015; Kendall, 1990), numerous studies have shown that the catch rates of smaller species (total length (TL) or disc width (DW) <1 m) remain unaffected (Brewer *et al.*, 2006; Courtney *et al.*, 2008; Griffiths *et al.*, 2006; Raborn *et al.*, 2012).

The introduction of TEDs in the eastern king prawn (EKP: *Melicertus plebejus*) fishery in southern Queensland, Australia in 2001 has had little effect on the catch rates of small elasmobranchs (Kyne *et al.*, 2002). The EKP fishery operates between the Swains Reefs (22° 10' 12"S, 152° 41' 45"E) in central Queensland and the New South Wales border (28° 09' 52" S, 153° 32' 51" E), with annual landings of ~3000 t, valued at approximately A$40 million. For management purposes, the fishery is partitioned into two separate components, the shallow water EKP fishery and the deep water EKP fishery, based on the 50 fathom (~91 m) bathymetric contour. Total discard rates from the shallow water fishery are much higher at 9.56 kg ha^{-1} (Courtney *et al.*, 2006) than from the deep water fishery (1.11 kg ha^{-1}) (Courtney *et al.*, 2014). 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 of 12 cm. Kyne *et al.* (2002) reported that two of the most common elasmobranchs found in the discarded portion of the shallow water EKP catch were the common stingaree (Urolophidae: *Trygonoptera testacea*) and the eastern shovelnose ray (Trygonorrhinidae: *Aptychotrema rostrata*).

Trygonoptera testacea and *A. rostrata* are small (<1.2 m TL) batoids endemic to Australia's east coast (Kyne, 2016). Both species are known to occur to depths of 90–100m, feeding on benthic crustaceans

(Kyne and Bennett, 2002a; 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 (PTS) estimates for these and other species represent the greatest source of uncertainty in assessing the effect of prawn trawling on elasmobranchs within, and adjacent to, the GBRMP.

The PTS of elasmobranchs is poorly understood (Braccini *et al.*, 2012; Dapp *et al.*, 2016; Oliver *et al.*, 2015; 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 most studies in the primary literature were conducted in pelagic longline fisheries, whereas 21 were trawl related (including beam trawl and scallop dredge) and only two studies were of prawn trawls (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 (Benoît *et al.*, 2012; Benoît *et al.*, 2013; Dapp *et al.*, 2016; Musyl *et al.*, 2011). Most of the trawl-based field studies assessing the PTS of elasmobranchs have been conducted in northern hemisphere fish trawls (e.g. Mandelman *et al.*, 2013; Mandelman and Farrington, 2007a; Revill *et al.*, 2005; Rodríguez–Cabello *et al.*, 2005) and have shown that survival is highly variable between species (Ellis *et al.*, 2017). For example, in the Falkland Islands squid trawl fishery, Laptikhovsky (2004) found that the PTS of skates (Rajidae) ranged between 0% for *Bathyraja griseocauda* and *Bathyraja macloviana* to 71.4% for *B. albomaculata*. Several factors have been shown to affect the PTS of elasmobranchs including: catch weight (Benoît *et al.*, 2010; Enever *et al.*, 2010; Mandelman and Farrington, 2007a; Saygu and Deval, 2014), air exposure (Benoît *et al.*, 2010; Cicia *et al.*, 2012; Frick *et al.*, 2010), tow duration (Enever *et al.*, 2009; Fennessy, 1994; Saygu and Deval, 2014), fish size (Benoît *et al.*, 2013; Depestele *et al.*, 2014; Enever *et al.*, 2010; Mandelman *et al.*, 2013; Saygu and Deval, 2014; Stobutzki *et al.*, 2002), temperature (Cicia *et al.*, 2012) and sex (Enever *et al.*, 2009; Laptikhovsky, 2004; Mandelman *et al.*, 2013; Stobutzki *et al.*, 2002). These and other factors have been shown to interact (Davis, 2002), necessitating the use of controls, where possible, to isolate the effects of individual factors (Enever *et al.*, 2010; Mandelman *et al.*, 2013).

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. The fate of these discards is unknown. The aims of the present study were to assess the short-term (~3 days) PTS and to examine factors affecting the survival of the two most common elasmobranchs found in the catch, namely *T. testacea* and *A. rostrata*.

5.2 Materials and methods

This work was undertaken in accordance with General Fisheries Permit 186281, Marine Parks Permit QS2015/MAN322, Queensland Department of Fisheries and Agriculture Animal Ethics Approval Number CA2015/06/867 and James Cook University Ethics Approval Number A2236.

The PTS of *T. testacea* and *A. rostrata* was assessed during a dedicated experiment conducted off Southport (Figure 6), southern Queensland (27°49.2' S; 153°30.0 E). This area was chosen for the survival experiment as past research (Courtney *et al.*, 2007) indicated that both species occur in this area. Southport is a popular port, supporting at least 12 trawlers targeting EKPs year-round, although at least 200 vessels operate in the EKP fishery.

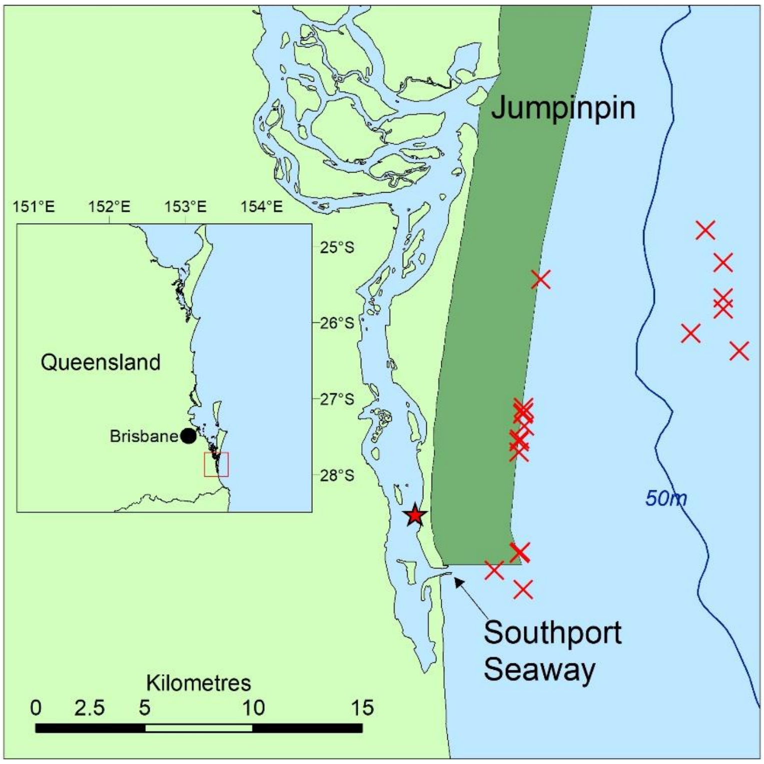

Figure 6: Location of the post-release survival experiments conducted during 2016. The red crosses represent the starting point of each trawl undertaken by the *C–Rainger* and the star represents the anchorage used by the *Tom Marshall* whilst the animals were housed in the vessel's on-board tanks. The dark green area represents grounds closed to trawling.

FV *C–Rainger*, a 15.6 m steel prawn trawler, was engaged to undertake trawls on known prawn grounds. *C–Rainger* used triple gear (Broadhurst *et al.*, 2013), consisting of three 12.8 m headline Florida flyer nets, spread by louvre-style otter boards. The body of each net was constructed from ~50 mm (2 inch) #36 ply polyethylene trawl mesh, whereas the codends were constructed from ~45 mm (1.75 inch) #60 ply polyethylene mesh. All nets were fitted with a top-shooting, single-grid hard TED with a bar space of 120 mm and a bigeye BRD as required by legislation. FRV *Tom Marshall* was chartered to undertake trawls to catch animals to control for a range of factors including trawl duration and time-on-deck. The *Tom Marshall* is a 14.5 m aluminium catamaran that used a single beam trawl towed from the stern. The beam net was a 6.5 m Florida flyer equipped with a top-shooter Wicks TED with a bar space of 120 mm and a fisheye BRD. The body of the net and the codend were constructed from the same materials used on the *C–Rainger*.

5.2.1 Experimental procedure

The PTS of *A. rostrata* and *T. testacea* was assessed during three separate sampling trips. The sampling trips were conducted over 5 days from 11 March 2016, over 5 days from 28 October 2016 and over 4 days from 15 December 2016. On the first night of each trip, 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 (PST13S; Hallprint, Adelaide, SA, Australia) and moved to an ~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) : 1, dead or nearly dead, no body movement, slight movement of spiracles; 2, limp wing movement, some spiracle movement; and 3, vigorous wing or body movement, rapid spiracle movement. The weight of the discards was quantified as the number of baskets of discards multiplied by 40 kg, the mean weight of a basket on the first night of trawling. Trawl duration, defined as the time between the end of winch away and the start of haul back, trawl 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 *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 6). Flow-through seawater was supplied to each container at a rate of ~36 L min–1. The tag number of any dead animals was recorded before the dead 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 sampling night, the *Tom Marshall* undertook 20 individual trawls using a 5 m beam. The objective of these trawls was to collect control animals of both species: short (10–20 min) trawls were conducted in relatively shallow (~25 m) water, minimising factors known to affect PTS, such as total catch weight and trawl duration (Enever *et al.*, 2009; Fennessy, 1994). All individuals were tagged with a streamer tag before being placed in a 150 L insulated fish box supplied with flow-through seawater. As for the *C–Rainger*, time-on-deck was recorded for each individual to assess its effect on survival. 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 animals were transferred randomly to one of the two 1400 L on-board holding tanks.

For all three sampling trips, the animals were monitored and, in accordance with previous studies (Enever *et al.*, 2009; Mandelman *et al.*, 2013; Mandelman and Farrington, 2007a) and results reported by (Wassenberg and Hill, 1993), survival was assessed after three days (72 h). During that time, the tanks were inspected every 2 h and dead animals were removed and stored in the vessel's on-board freezer. For each animal, the TL for *A. rostrata* or DW for *T. testacea* (in mm) was 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.

5.2.2 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 TL or DW, 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 (Version 3.3.3, R Foundation for Statistical Computing, Vienna, Austria, see https://www.R-project.org/, accessed 19 November 2017). 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 the 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).

5.3 Results

The *C–Rainger* completed 18 trawls during the three sampling trips (Table 4), with trawl duration ranging from 64 to 217 min. 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; however, six maskrays (*Neotrygon* spp.) and three coffin rays (*Hypnos monopterygius*) were caught in total and discarded alive immediately on capture.

The *Tom Marshall* completed 36 control trawls, ranging in duration from 10 to 20 min (Table 4). All trawls were conducted adjacent to the Southport Seaway (Figure 6) in depths between 19 and 47 m (mean (± s.e.m.) 27.2 ± 1.0 m). The only other elasmobranch caught by *Tom Marshall* was the Kapala stingaree (*Urolophus kapalensis*): two individuals were caught during the second and third sampling trips (113 and 210 mm DW respectively), the larger of which aborted two pups (78 and 79 mm DW) during the sorting process. The two larger animals were tagged and retained, whereas the pups died soon after capture.

Table 4: Number of individual *Trygonoptera testacea* and *Aptychotrema rostrata* caught by each vessel during the three post-release survival experiments conducted in 2016. Unless indicated otherwise, data are given as the mean ± s.e.m.

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	0	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

5.3.1 Trygonoptera testacea

In all, 187 *T. testacea* were assessed for PTS during the three experiments (Table 4), ranging in size between 75 and 245mm DW (Figure 15a). Animals caught on the *C–Rainger* were larger than those caught on the *Tom Marshall* (Figure 15a; t = 6.804, P < 0.001). Females were more prevalent than males, with 109 and 73 caught respectively, and females were significantly larger than males (Figure 15b; t = 2.8776, P < 0.0 1). 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*. *T. testacea* were more abundant on inshore grounds because none were captured by the *C–Rainger* on deeper (~50m) trawl grounds during the first experiment (Figure 6). For *T. testacea*, time on deck ranged between 1 and 28 min (mean (± s.d.) 14.5 ± 7.1 min). Of the 187 *T. testacea* caught, 11 were given a condition index of 1, whereas 71 were given a score of 3 (Table 5). Generally, individuals were in good condition when placed in the holding tanks. Approximately half the animals contained throughout the experiment suffered abrasions to the ventral surface due to contact with the plastic holding tanks. Further, infection at the tag site was obvious in ~25% of the animals. However, survival was consistent across animals with and without abrasion injury or infection at the tag site.

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 and, as such, data from the two trips were pooled. Survival of the control animals during the two trips did not differ significantly (P = 0.603) from that of animals caught on the *C–Rainger*, and trawl duration had no effect on PTS ($P = 0.154$). The natural logarithm of DW was the best predictor of survival ($P < 0.001$), with survival increasing with size (Figure 7; $\beta = 8.598$, s.e. = 1.454). Both sex ($P = 0.051$) and time on deck ($P = 0.054$) had marginally significant effects on survival. The survival of females was higher than that of males ($\beta = 0.895$, s.e. = 0.461), whereas increasing time on deck was found to result in lower survival ($\beta = -0.109$, s.e. = 0.057). Given the confounding effect of females having a larger mean DW than males (b), a term representing the interaction of sex 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 ($P < 0.001$) on PTS (Figure 7). Mean (±s.e.m.) overall PTS for female and male *T. testacea* was 33.5 ± 6.0 and 17.3 ± 5.5% respectively. The condition index did not affect PTS despite a trend for higher survival with increasing levels of condition (Table 5).

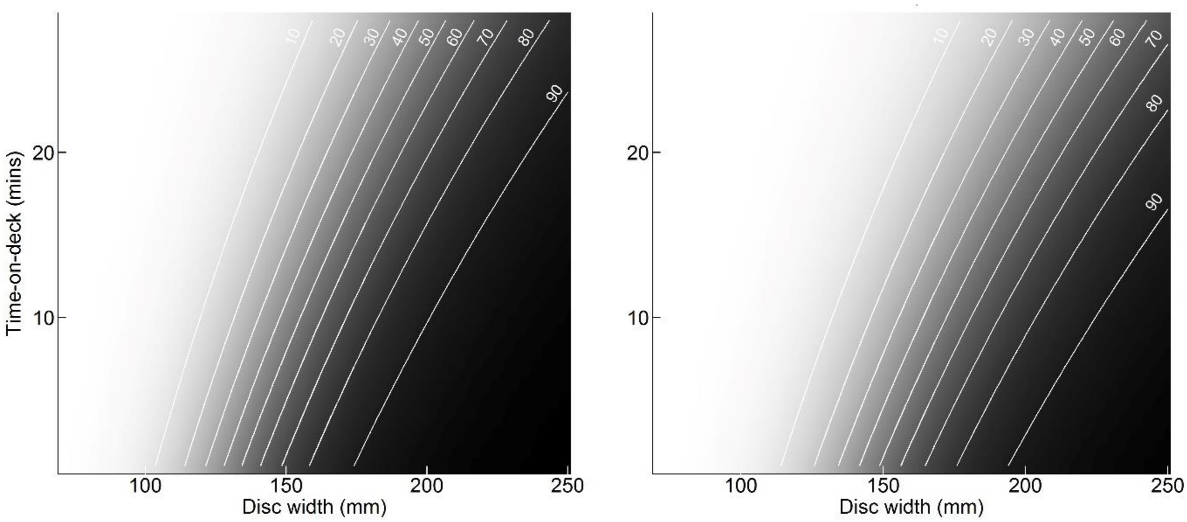

Figure 7: Post-trawl survival of common stingarees (*Trygonoptera testacea*) as a function of disc width and time on deck for (a) females and (b) males caught during two sampling trips conducted in southern Queensland, Australia. The background shading represents increasing survival from 0% (white) to 100% (black) and the white contour lines indicate survival in 10% increments.

5.3.2 Aptychotrema rostrata

Of the 155 *A. rostrata* assessed for PTS during the three experiments, 118 (~76%) were caught aboard the *C–Rainger* (Table 4). Of these, 24 (~20.3%) died, whereas only one of the 37 (2.7%) control animals caught on the *Tom Marshall* died. *A. rostrata* ranged in size between 166 and 555 mm TL (Figure 16a) and animals caught on the *C–Rainger* were larger than those caught on the *Tom Marshall* ($t = 2.702$, $P = 0.009$). Females and males were equally represented in catches ($n = 78$ and $n = 77$ respectively) and their size did not differ significantly (Figure 16b; $t = -0.321$, $P = 0.749$). Time on deck ranged between 1 and 63 min (mean ± s.d., 18.4 ± 12.0 min). Of the 155 *A. rostrata* caught, 12 were given a condition index of 1, 101 were given a score of 2 and 42 were given a score of 3 (Table 5). Very little physical damage was observed on the animals before they were placed into the holding tanks and, in contrast with *T. testacea*, infection at the tag site was not obvious on any *A. rostrata* held during the three sampling trips.

Table 5: Post-trawl survival (PTS) of *Trygonoptera testacea* and *Aptychotrema rostrata* as a function of Condition Index, as described by (Enever *et al.*, 2009). Unless otherwise indicated, data are given as the mean ± s.e.m. The condition index was graded as follows: 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

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 TL was the best predictor of survival ($P = 0.011$), with survival increasing with TL ($\beta = 3.227$, s.e. = 1.272). Both the natural logarithm of trawl duration ($\beta = -1.206$, s.e. = 0.587) and time on deck ($\beta = -0.050$, s.e. = 0.022) were found to have a significant negative effect on survival ($P = 0.040$ and $P = 0.026$ respectively; Figure 8). Mean (±s.e.m.) overall PTS for *A. rostrata* was 86.8 ± 3.2%. The 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 5).

5.4 Discussion

These results represent the first short-term 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 present 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), whereas the survival of *T. testacea* was relatively low, especially for males (17.3%).

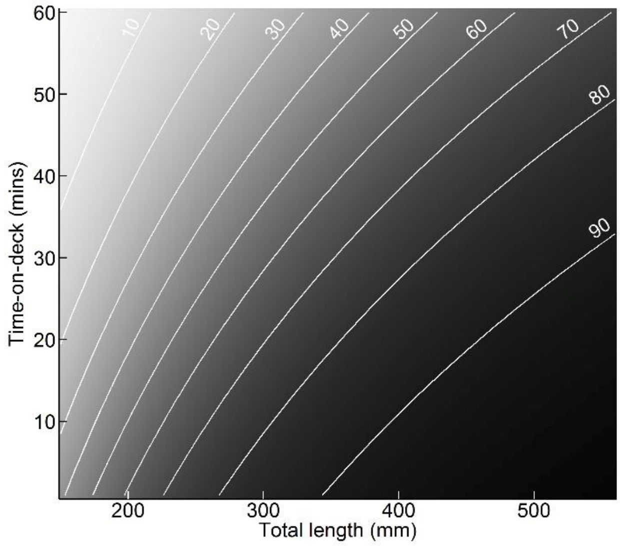

Figure 8: Post-release survival of eastern shovelnose rays (*Aptychotrema rostrata*) as a function of total length and time on deck at the mean trawl duration of 146 min caught during three sampling trips conducted in southern Queensland, Australia. The background shading represents increasing survival from 0% (white) to 100% (black) and the white contour lines indicate survival in 10% increments.

Species-specific differences are evident in previous studies assessing PTS of elasmobranchs from trawl gear. For example, the PTS of thornback skate (*Raja clavata*) was 80.8% in the Turkish bottom trawl fishery, compared with 20.6% for the brown skate (*Raja miraletus*) (Saygu and Deval, 2014). Similarly, Enever *et al.* (2009) reported a PTS of 59% for *R. clavata*, compared with 33% for the cuckoo skate (*Leucoraja naevus*). In both studies, the higher PTS for *R. clavata* was attributed to its accentuated spines, which provide improved physical protection compared with other species. Differences in morphology were obvious in the two species assessed in the present study: *A. rostrata* are covered in fine denticles (Kyne, 2016), whereas *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 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, whereas no *A. rostrata* appeared affected. The addition of an anti-biotic and anti-fungal ointments to the tag wounds, such as those used 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; Ellis *et al.*, 2017; Mandelman and Farrington, 2007a). Ellis *et al.* (2017) suggest that captive stress, stocking densities and environmental conditions may affect post-release survival 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. Benoît *et al.*, 2010; Cicia *et al.*, 2012; Depestele *et al.*, 2014; Enever *et al.*, 2009; Enever *et al.*, 2010; Kaiser and Spencer, 1995; Revill *et al.*, 2005; Rodríguez–Cabello *et al.*, 2005; 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 *et al.*, 2013; Mandelman and Farrington, 2007a; Rulifson, 2007), land-based tanks (Cicia *et al.*, 2012; Mandelman and Farrington, 2007b), blood physiology (Frick *et al.*, 2010; Mandelman and Farrington, 2007a; Mandelman and Farrington, 2007b) and trawl simulation studies (Frick *et al.*, 2010; Heard *et al.*, 2014).

However, each of these methods 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 (i.e., *Heterodontus portusjacksoni*, *Mustelus antarcticus* and *Urolophus paucimaculatus*) in a laboratory and concluded that the PTS 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 (Kaiser and Spencer, 1995; Van Beek *et al.*, 1990; Wassenberg and Hill, 1993) this method would have likely yielded underestimates of PTS for *A. rostrata* and *T. testacea*. Passive acoustic telemetry, used by Dudgeon *et al.* (2013) to assess the site fidelity of the zebra shark (*Stegostoma fasciatum*) in south east Queensland, was deemed unsuitable for the present study given the time required to carefully perform invasive surgery at night on small animals (~10 cm DW) during commercial trawl conditions. Similarly, pop-up satellite archival tags (PSATs), such as those used by Campana *et al.* (2016) to estimate post-release survival of pelagic elasmobranchs, 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 5) on capture in the present study, 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 5). Only 38% of *T. testacea* with a condition index of 3 survived, reinforcing the inadequacy of at-vessel mortality 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 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 present study, the survival of animals presenting with a condition index of 1 had the most variable PTS (Table 5) 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 present study, bull sharks (*Carcharhinus leucas*) were observed feeding on the discards from the *C–Rainger* while the catch was being sorted. Further, blue swimmer crabs (*Portunus armatus*) and three-spot crabs (*Portunus 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 7 and Figure 8). This is consistent with previous studies for elasmobranchs (Depestele *et al.*, 2014; Enever *et al.*, 2010; Saygu and Deval, 2014; Stobutzki *et al.*, 2002). 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 UK, resulting in higher PTS. The size-related difference in survival has been attributed to reduced resilience of smaller animals to fatigue and injury (Benoît *et al.*, 2013; Davis, 2002). 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 (Broadhurst *et al.*, 2006; Davis, 2002; ICES, 2014) and is a function of the time required to process the catch (Davis, 2002). In the present study, time-on-deck affected the PTS of both *T. testacea* (Figure 7) and *A. rostrata* (Figure 8). Time-on-deck reflected 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 (~0.75 m^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 EKP 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), showed 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 present study, prohibiting analysis of this metric.

Female *T. testacea* were more likely to survive than males. Studies that have found sex-specific PTS in elasmobranchs invariably report that the survival of females is higher (Enever *et al.*, 2009; Laptikhovsky, 2004; Mandelman *et al.*, 2013; Stobutzki *et al.*, 2002). Enever *et al.* (2009) and Mandelman *et al.* (2013) suggest higher survival in females is a result of the thicker skin that 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 present study, female *T. testacea* were larger (Figure 15), confounding the effect of sex, given the 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 sex-specific, nor were there significant size differences between the sexes of this species. These results suggest that sex-specific differences in PTS in other studies (e.g. Stobutzki *et al.*, 2002) may have been due to the larger size of the females rather than any morphological differences between sexes.

Tow duration had a negative effect on the PTS of *A. rostrata*. Where measured, increased tow duration has resulted in lower PTS for elasmobranchs (Enever *et al.*, 2010; Fennessy, 1994; Mandelman *et al.*, 2013; Mandelman and Farrington, 2007a). 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 present 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.

In conclusion, TEDs are effective at reducing the catch of turtles from prawn trawls: however, current TED regulations regarding bar spacing have resulted in the retention of a large number of smaller elasmobranchs in prawn trawl fisheries worldwide. It is, therefore, prudent to assess PTS of these animals when determining the ecological risk posed by prawn trawling. Field-based experiments conducted as part of the present study have shown that *A. rostrata* are more resilient to trawl capture and release than *T. testacea*. To ensure maximum PTS for both species, fishers should limit the duration of trawls where these species are present and fishers should prioritise removing elasmobranchs from the catch and return them to the sea as quickly as possible.

In excess of 45 species of elasmobranch have the potential to interact with the EKP fishery in southern Queensland (Last and Stevens, 2009). Of these, the scalloped hammerhead (*Sphyrna lewini*) is considered to be 'endangered' according to the IUCN Red List (Baum *et al.*, 2009), whereas the Endeavour skate (*Dentiraja endeavouri*) and the bluegrey carpetshark (*Brachaelurus colcloughi*) are classified as 'Vulnerable' (Kyne, 2011; Kyne *et al.*, 2015). 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.

6. Applying a quantitative method to assess the ecological risk posed to sharks, rays, and chimaeras by a sub-tropical trawl fishery

Plate 6: Sorting the daylight shot, offshore Bribie Island, 19 May 2021.

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6.1 Introduction

It has been estimated that 55% of the catch from penaeid-trawl fisheries is discarded (Gilman *et al.*, 2020). This high discard rate is a result of the nets used to target penaeids, characterised by small mesh sizes (<50 mm), which are towed on the seafloor. As such, the discarded portion of penaeid-trawl catches is comprised of hundreds of species (Courtney *et al.*, 2008; Fennessy and Groeneveld, 1997; Velip and Rivonker, 2015; Ye *et al.*, 2000), including species of conservation concern, such as sea turtles (Brewer *et al.*, 2006) and sea snakes (Milton, 2001). Chondrichthyans (sharks, rays and chimaeras) are also incidentally caught by penaeid trawls and have been the subject of increased conservation efforts in the past two decades (Dulvy *et al.*, 2017). This group is vulnerable to overexploitation as a result of life history characteristics including late maturity, long life spans and slow growth (Dulvy *et al.*, 2008). Dulvy *et al.* (2021) reported that about one-third of all chondrichthyans are threatened by overfishing, classified as Vulnerable, Endangered or Critically Endangered by the International Union for Conservation of Nature's Red List of Threatened Species (IUCN Redlist), mainly due to capture in fisheries targeting other species. Understanding the impact of incidental capture can help identify management needs.

Chondrichthyans are caught incidentally in Australia's largest penaeid-trawl fishery, the Queensland east coast otter trawl fishery (QECOTF, Kyne *et al.*, 2002). The QECOTF is a multi-sector fishery that operates between 11°S and 28.3°S to depths of ~250 m in north-eastern Australia (see Wang *et al.*, 2020 for details). Fishers target penaeid prawns (*Penaeus* spp., *Melicertus* spp., *Metapenaeus* spp. and *Fenneropenaeus* spp.), saucer scallops (*Ylistrum balloti*), scyllarid lobsters (Moreton Bay bugs, *Thenus* spp.) and squid (Loliginidae) using benthic otter trawls, but they can also retain limited amounts of permitted (byproduct) species including portunid crabs (*P. armatus* and *P. sanguinolentus*), Balmain bugs (*Ibacus* spp), and cuttlefish (*Sepia* spp.). Further, two vessels share an annual 1100 t total allowable catch of stout whiting (*Sillago robusta*) using Danish seine and fish trawling gear. All other species, including the chondrichthyans, are returned to the sea as soon as practicable: approximately 21,000 t of catch is discarded annually in the QECOTF, representing 70% of the total catch (Wang *et al.*, 2020), and accounting for >25% of Australia's total annual discards from all fisheries combined (Kennelly, 2020).

In Australia, all export fisheries are subject to assessment to ensure that management arrangements are ecologically sustainable for both target and discarded species. Failure to do so can result in the revocation of export permits, prohibiting access to lucrative international markets. Generally, stock assessments are undertaken for target species due to the availability of relevant information on life history and catch-and-effort data (e.g. Helidoniotis *et al.*, 2020; Hutton *et al.*, 2018; Noell and Hooper, 2015; Wortmann, 2021). Conversely, demonstrating the sustainability of discarded species is more difficult due to the paucity of these data (Baje *et al.*, 2021; Braccini *et al.*, 2006). The most common method of assessing the impact of fisheries on data-poor catch components, particularly the chondrichthyans, is through ecological risk assessment (ERA, e.g. Baje *et al.*, 2021; Braccini *et al.*, 2006). Generally, qualitative ERAs compare the likelihood of capture for each species to its resilience to fishing impacts (Astles *et al.*, 2009; Baje *et al.*, 2021; Hobday *et al.*, 2011) and rely on experts to provide relevant information on each species.

A quantitative approach to assess the risk posed to chondrichthyans by penaeid-trawl fisheries, known as Sustainability Assessment for Fishing Effects or SAFE, was developed to provide a scientifically robust measure of fishing impacts (Zhou *et al.*, 2015; Zhou and Griffiths, 2008; Zhou *et al.*, 2009; Zhou *et al.*, 2011). The SAFE method has been used to assess the risk to species discarded in major fisheries managed by the Australian Commonwealth Government and inshore fisheries in New Zealand. 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.

The objective of the current study was to use SAFE to determine the ecological risk to chondrichthyans caught in the southern portion of the QECOTF (>24.5°S) in the 2019 fishing year (1 November 2018 to 20 September 2019, Helidoniotis *et al.*, 2020). In this case, ecological risk refers to the risk that a population is subjected to levels of fishing mortality that are unsustainable in the long term. Pears *et al.* (2012) used qualitative methods to assess the risk posed by the QECOTF to a range of catch components, including chondrichthyans, and marine habitats within the Great Barrier Reef Marine Park (GBRMP), leaving the risk posed to chondrichthyans south of the GBRMP unquantified. This area (Figure 9) has been trawled since the 1950s, primarily targeting the eastern king prawn (EKP, *Melicertus plebejus*). In the 2019 fishing year, logbook data indicate that 1435 t of EKP were landed, along with ~108 t of Moreton Bay bugs (*Thenus* spp.), 105 t of saucer scallops and ~1063 t of *S. robusta*. It should be noted that the area within Moreton Bay (Figure 9) is managed as a separate fishery and was excluded from the analysis.

6.2 Materials and Methods

Data from previous research (Courtney *et al.*, 2007; Dodt, 2005; Kyne *et al.*, 2002), the Queensland Fisheries' Fishery Observer Program (FOP) and the Queensland Museum indicate that at least 48 species of chondrichthyan occur south of the GBRMP (see Table 15). The ecological risk posed to these 48 species by the trawl fisheries in the area south of the GBRMP to the Queensland/New South Wales border, to a depth of 250 m (hereafter referred to as 'the study area', Figure 9), in the 2019 fishing year was quantified using the SAFE method described by Zhou *et al.* (2014). The general approach of the SAFE method is to compare the instantaneous fishing mortality of each species in a given year (F_y) to the reference point that corresponds to the maximum sustainable fishing mortality (Fmsm, Zhou *et al.*, 2015; Zhou *et al.*, 2011). The derived estimates of F_y are also compared to other reference points to assess the risk posed to each species.

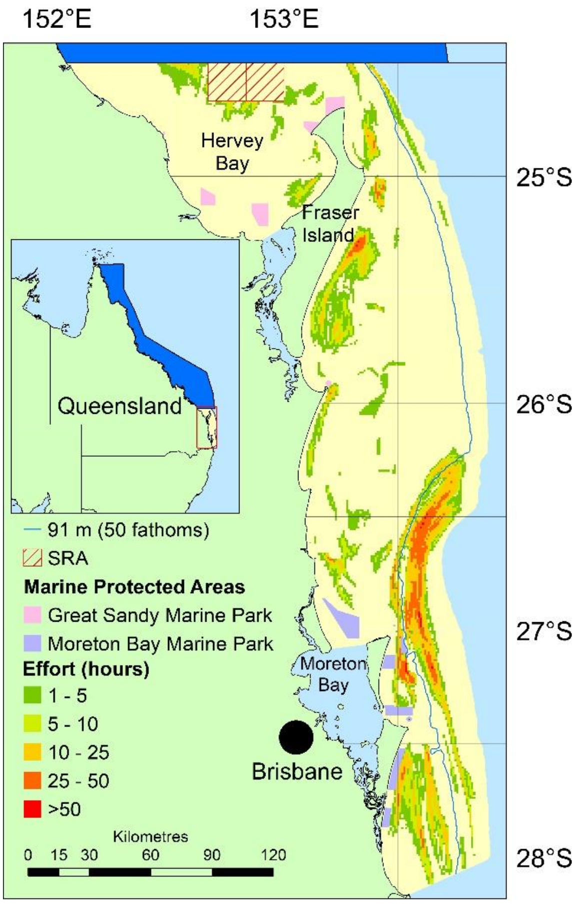

Figure 9: Spatial extent of the study area, in pale yellow, on the Queensland (Australia) east coast. Trawl effort is the number of hours trawled in each one square kilometre in the 2019 fishing year (1 November 2018 to 30 September 2019) calculated using TrackMapper software (Peel and Good, 2011). Also shown are the 30-minute by 30-minute grids used for reporting purposes, and the 91 m (50 fathom) bathymetric contour which separates the shallow water and deep water eastern king prawn (EKP, *Melicertus plebejus*) fisheries. The red hatched area is the Hervey Bay Scallop Replenishment Area (SRA), which has been closed to fishing since 2016. Also shown are the areas closed to all fishing as part of the Great Sandy Marine Park and the Moreton Bay Marine Park. Inset: The extent of the Great Barrier Reef Marine Park and the study area are coloured dark blue and pale yellow, respectively. The extent of the main window is shown as a red rectangle.

6.2.1 Fishing mortality

In accord with Zhou *et al.* (2015) and Zhou *et al.* (2011), the instantaneous fishing mortality for species i in fishing year y, F_{iy}, is:

$$F_{iy} = \frac{C_{iy}}{N_{iy}} = \frac{Q_i(1 - E_i)(1 - S_i)\, A_{iy}}{A_i}$$

where: C_{iy} is number of individuals of the species i dying as a result of interaction with trawl gear in fishing year *y*; N_{iy} is the population size of the species i in the same year; Q_i is the efficiency of the fishing gear expressed as the probability of catching species i; E_i is the escape rate resulting from the presence of a turtle excluder device (TED); S_i is the discard survival rate; A_{iy} is the area trawled within the distribution of the species in the study area in year *y*; and A_i is the species distribution range within the study area. In the current study, the impacts of four gear types were assessed: penaeid trawl, scallop trawl, fish trawl and Danish seine. To avoid confusion, all deployments of any gear type will be hereafter referred to as a "trawl shot".

6.2.1.1 Species distribution, A

The area over which each species occurred in the study area was quantified using published data, primarily from Last *et al.* (2016) and Last and Stevens (2009). These resources provided information regarding the latitudinal extent of each species' distribution, along with the depth range within which each species occurs. Where these resources provided inadequate information, the Atlas of Living Australia (ALA, https://www.ala.org.au/) and the respective IUCN Red List assessment pages were used to determine species distribution. In all cases, a distribution shapefile of each species was generated in ArcMap (ESRI, 2011) and the area over which each species occurred in the study area was calculated using the same software.

6.2.1.2 Area trawled, A_y

Within each species' distribution in the study area, the area trawled by penaeid and scallop trawls was quantified using vessel monitoring system (VMS) data (Peel and Good, 2011). The VMS data for the period 1 November 2018 to 20 September 2019 south of 24.5°S were imported into ArcMap and the 'Clip' function was used to isolate the trawl tracks that were undertaken within the bounds of each species' distribution. These data provided the length of each trawl shot, in kilometres, undertaken by vessel *v* using gear type *g* (penaeid or scallop) on day *d* using a TED with escape-hole orientation *o* ($l_{v,g,d,o}$) within each species' distribution. The width of each trawl was obtained from gear information supplied by the operator of vessel *v* on day *d*, which described relevant gear attributes including the number of nets deployed, the TED escape-hole orientation, *o*, and the combined headrope length ($h_{v,g,d,o}$), in metres. Finally, the area swept by vessel *v* on day *d* using gear type *g* with a TED with escape-hole orientation *o*, in km^2, was defined as $SA_{v,g,d,o} = l_{v,g,d,o} f_{v,g} \left(h_{v,g,d,o}/1000 \right)$ where $f_{v,g}$ is a trawl net spread factor from Sterling (2000), as a function of the number nets deployed (Wang *et al.*, 2020). Areas swept were then summed with respect to gear type *g* (penaeid, scallop) and TED escape-hole orientation o.

The area of each trawl shot conducted by the fish trawl vessel and the Danish seine vessel were calculated using methods described by Wang *et al.* (2020). Only bottom-shooter TEDs were used by the fish trawl vessel and, as such, the areas swept within each species distribution were summed with respect to gear type *g* (fish trawl and Danish seine).

6.2.1.3 Catch efficiency, Q

The catch efficiency Q of each gear type *g* on species i was quantified following Zhou *et al.* (2014). The study area was divided into grids for reporting purposes, based on 30-minute latitude and longitude bands (Figure 9), that are comparable to the equal-sized cells identified by Zhou *et al.* (2014). The number of individuals for a particular species in cell *c*, N_c, is assumed to follow a Poisson distribution with a mean λ such that $N_c \sim pois(\lambda)$. An individual in cell *c* that encounters gear type *g* is either caught or not caught, hence, the number of fish caught at sampling time j (year x month) follows a binomial distribution $C_{c,j,g} \sim bin(Q_g, n_{c,j,g})$, where n_{cjg} is the number of individuals at sampling time *j* within the area swept by gear type *g*. The $n_{c,j,g}$ is the product of the area swept by each gear type *g* (a_g) and the density of individuals within cell *c*, and assumes a Poisson distribution such that $n_{c,j,g} \sim pois(a_g \times D_c)$, where D_c is the density of fish within cell *c*, calculated by dividing N_c by the area of cell *c*.

The number and density of chondrichthyans in the study area was estimated from data collected during previous research (Courtney *et al.*, 2007; Dodt, 2005; Kyne *et al.*, 2002) and the FOP (see a summary by Rowsell and Davies, 2012). These data provided the number of chondrichthyans caught by the four gear types, identified to species. Relevant data describing the characteristics of each shot undertaken during this sampling, such as location, depth, duration, and net size, were also recorded. Again, the area swept for each trawl shot was calculated in accord with Wang *et al.* (2020).

The catch and abundance of each species was modelled in a Bayesian framework using WinBUGS (Spiegelhalter *et al.*, 2007). Bayesian models were fit in WinBUGS using R statistical software (Version 3.6.1, R Foundation for Statistical Computing, Vienna, Austria, see https://www.R-project.org/, accessed 18 February 2021) via the 'R2WinBUGS' package (Sturtz *et al.*, 2005). Priors for both Q and λ were uninformative: $Q \sim beta(1,1)$ and $\lambda \sim lognorm(1, 0.1)$. Three Markov Chain Monte Carlo (MCMC) chains with 100,000 iterations, after a burn in period of 70,000, were used to determine parameter posterior distributions. Model convergence was assessed using the Gelman–Rubin test statistic and by visual examination of chain trajectories. In accord with Campbell *et al.* (2020), the low number of chondrichthyans caught during sampling necessitated the grouping of species by genus, family, or order to ensure adequate numbers for model fitting. Further, cases were restricted to those cells where the number of individuals caught during the sampling was >30 to ensure the samples were collected from areas where the species or species groups are known to occur, and to avoid issues of zero inflation and overdispersion.

6.2.1.4 Escape, E

Since TEDs are not required in the Danish seine gear, $E = 0$ for this gear type. Escape from TEDs used by penaeid, scallop and fish trawls, as a function of size, was estimated from models developed by Campbell *et al.* (2020). These authors estimated the escape of chondrichthyans, as a function of fish length, for species classified within four taxonomic orders: Carcharhiniformes, Orectolobiformes, Myliobatiformes and Rhinopristiformes. Size data (total length, TL, for all species except the Myliobatiformes which are measured by disc width, DW), obtained from various studies, were used to inform the median size of each species caught by otter trawls targeting penaeid prawns and saucer scallops in eastern Australia (Brewer *et al.*, 2006; Courtney *et al.*, 2007; Kyne *et al.*, 2002; Rigby *et al.*, 2016a; Rigby *et al.*, 2016b). The median size was then included in the prediction data frame used to estimate the mean (± S.D.) escape rates for each species or species group from the relevant models (Campbell *et al.*, 2020). Escape from the TED used by the fish trawl vessel was assumed to be equal to the escape from bottom-shooter TEDs used in penaeid and scallop trawls.

Escape was set to $E = 0$ for those species that: 1) were not classified within the four taxonomic orders, 2) were classified within the four taxonomic orders but lacked reliable size data with which to determine median size, or 3) had no other published information regarding escape from TEDs comparable to those used in the QECOTF.

6.2.1.5 Survival, S

Only two of the species assessed (*Aptychotrema rostrata* and *Trygonoptera testacea*) have published post-release survival rates based on dedicated experiments assessing this metric (Campbell *et al.*, 2018). Stobutzki *et al.* (2002) published the at-vessel mortality estimates of five species of chondrichthyan caught incidentally in the Northern Prawn Fishery (*Gymnura australis*, *Hemigaleus australiensis*, *Maculabatis toshi*, *Rhizoprionodon acutus* and *Rhynchobatus australiae*) and these have been used as estimates of survival for these species in the current study. Further, data collected as part of research conducted by Courtney *et al.* (2007) were used to estimate the at-vessel survival of four species assessed in the current study (*Asymbolus analis*, *Asymbolus rubiginosus*, *Dentiraja endeavouri* and *Figaro boardmani*). For the remainder of the species assessed, survival was set to $S = 0$. Survival was assumed to be constant for all gear types, despite the presence of hoppers (i.e., water-filled tanks used to separate catch components prior to sorting) on the fish trawl and Danish seine vessels, likely resulting in increased survival.

6.2.1.6 Cumulative effects

The four gear types (penaeid trawl, scallop trawl, fish trawl and Danish seine) access the same population and the impacts of each gear type are, therefore, cumulative. The cumulative fishing mortality for species *i* by all gear types *g* in the 2019 fishing year is given by:

$$F_i{}^C = \sum_g F_{i,g}$$

Where available, variances in the estimation of E, S and Q were incorporated when evaluating F_i^C. These variances were obtained from the respective models used to estimate the parameters. The variance in the estimates of S reported by Stobutzki *et al.* (2002) was assumed to be 10% of the published point estimates. Using the built-in 'rnorm' function in R statistical software (ver. 4.0.2, R Foundation for Statistical Computing, Vienna, Austria, see https://www.R-project.org/, accessed 4 November 2021), a vector of 10,000 estimates of F was generated. The mean (+ 90% confidence interval) of these estimates was considered the point estimate of F.

6.2.2 Maximum sustainable fishing mortality

In accord with Zhou *et al.* (2011), three biological reference points, based on a simple surplus production model, were used to assess the risk to each species of chondrichthyan caught in the fishery. The first reference point is the instantaneous fishing mortality rate corresponding to the number of fish in the population that can be killed by fishing in the long term, designated as maximum sustainable fishing mortality (F_{msm}). This level of fishing mortality, corresponding to the fishing effort that produces the maximum sustainable yield (MSY) for a target species, results in a biomass that supports maximum sustainable mortality (B_{msm}). The second reference point, F_{lim}, is the instantaneous fishing mortality rate that corresponds to the biomass B_{lim} which represents $0.5B_{msm}$. Finally, F_{crash} is the 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 $\omega = 0.41$ (S.D. = 0.09) for chondrichthyans based on empirical data (Zhou *et al.*, 2012) and M is the instantaneous rate of natural mortality. Estimates of M were derived using eight empirical relationships (Table 6). Further, $F_{lim} = 1.5\omega M$ and $F_{crash} = 2\omega M$ (Zhou *et al.*, 2011).

Where possible, relevant life history characteristics were obtained from the primary literature; however, many species had no such data (Table 15). In these cases, a single estimate of M was possible based on L_∞, derived from the maximum length (Froese and Binohlan, 2000), and the average annual water temperature (http://www.fishbase.org, Table 6). Where incomplete growth information was available, the following equations were used to derive relevant parameters for use in the derivation of M: $\log L_\infty = 0.044 + 0.9841 \log L_{max}$ (Froese and Binohlan, 2000); $t_{mat} = 7.2 \ln t_{max} - 12.68$ (Frisk *et al.*, 2001); $t_{max} = \ln(2/k) \times 7$ (Smart *et al.*, 2018); and, for Rajidae only, $k = -0.17ln(L_{max}) + 0.97$ (Frisk *et al.*, 2001).

Temperature data required to estimate M were sourced from the eReefs website (https://ereefs.org.au/ereefs). Mean water temperature for the 2019 fishing year was calculated for the mid-point of the published depth range of each species. The mean of the derived estimates of F_{msm}, F_{lim} and F_{crash} were considered the respective point estimates, and the variance of each reference point was the range between their minima (i.e., min[F_{msm}], min[F_{lim}] and min[F_{crash}]) and maxima (i.e., max[F_{msm}], max[F_{lim}] and max[F_{crash}]).

Table 6: Eight methods used to estimate the instantaneous rate of natural mortality (M, year^{-1}) for use in the derivation of biological reference points F_{msm}, F_{lim} and F_{crash}. L_{∞}, k and t_0 are the von Bertalanffy growth parameters, T is the average daily water temperature during the 2019 fishing year, t_{mat} is the average age-at-maturity, and 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
$M = 1.65/t_{\mathrm{mat}}$	Jensen (1996)
$M = 4.118k^{0.73}L_{\infty}{}^{-0.33}$	Then *et al.* (2015)
$M = 4.899t_{\max}{}^{-0.916}$	Then *et al.* (2015)
$\ln(M) = 0.42\ln(k) - 0.83$	Frisk *et al.* (2001)
$1/M = 0.44t_{\mathrm{mat}} + 1.87$	Frisk *et al.* (2001)

6.2.3 Assessing risk posed by trawling

The risk of overfishing, defined as the population level that is unable to support its maximum sustainable fishing mortality, was considered as the primary concern for fishery managers. This risk is quantified by comparing the fishing mortality (F_{2019}) to the derived reference points such that:

1. Low risk: $F_{2019} < F_{msm}$ (overfishing is not occurring);
2. Medium risk: $F_{msm} \leq F_{2019} < F_{lim}$ (overfishing is occurring, but the population is sustainable);
3. Precautionary medium risk: $F_{2019} \geq$ min[F_{msm}] or F_{2019} + 90% CI $\geq F_{msm}$;
4. High risk: $F_{lim} \leq F_{2019} < F_{crash}$ (population may be driven to very low levels in the longer term);
5. Precautionary high risk: $F_{2019} \geq$ min[F_{lim}] or F_{2019} + 90% CI $\geq F_{lim}$;
6. Extreme high risk: $F_{2019} \geq F_{crash}$ (risk of local population extinction in the long term); and
7. Precautionary extreme high risk: $F_{2019} \geq$ min[F_{crash}] or F_{2019} + 90% CI $\geq F_{crash}$.

6.3 Results

The spatial extent of the trawl fishery south of the GBRMP encompasses 27,915 km^2 (Figure 9). Three species, *Gymnura australis*, *Sphyrna lewini* and *Tetronarce nobiliana*, occurred throughout the entire fishery (Table 16). The distribution of *Hypnos monopterygius*, *Orectolobus maculatus* and *Hemigaleus australiensis*, exceeded 25,000 km^2 in area. In contrast, *Squalus montalbani*, *Urolophus bucculentus* and *U. viridis* were distributed over less than 2,500 km^2.

In total, the trawled area within the fishery was 15,185 km^2 in the 2019 fishing year (Table 16). The area trawled by penaeid trawls with bottom- and top-shooter TEDs installed was 10,742 and 3,381km^2, respectively. Trawl effort targeting saucer scallops was much lower at 533 and 58 km^2 for vessels with bottom- and top-shooter TED-equipped nets, respectively. The Danish seine vessel swept an area of 330 km^2 and the fish trawl swept a total of 141 km^2.

The data from 1,175 net trawls conducted during previous research and from the Queensland Government's FOP were examined to determine the catch efficiency (Q) of each gear type used. The 1,175 net trawls comprised 160 trawls from the Danish seine vessel, 597 penaeid trawls, 261 fish trawls and 157 scallop trawls. A total of 6,566 individuals from 24 families and ten taxonomic orders were caught and identified to species (Table 16). The most common species caught were *Aptychotrema rostrata* (n = 3,933) and *Trygonoptera testacea* (n = 1,270). More than 100 *Hemigaleus australiensis*, *Urolophus kapalensis*, *D. endeavouri* and *Neotrygon trigonoides* were caught during the sampling; however, <10 individuals were sampled from each of 23 species assessed in the current study.

Table 7: Catch efficiency, Q, for each gear type. Sample size prevented the estimation of Q at the species level for all but the two most common species, *A. rostrata* and *T. testacea*. Species were grouped to estimate Q for the respective taxonomic orders. Carcharhiniformes, Orectolobiformes, Heterodontiformes, Squaliformes and Squatiniformes were grouped to "All sharks". All species were grouped to estimate the overall catch efficiency of each gear type.

Species/species group	Danish seine	Penaeid trawl	Fish trawl	Scallop
Aptychotrema rostrata	0.231 (0.037)	0.142 (0.005)	0.052 (0.005)	0.106 (0.033)
Trygonoptera testacea	0.237 (0.036)	0.138 (0.004)	0.054 (0.006)	-
Myliobatiformes	0.129 (0.006)	0.118 (0.020)	0.022 (0.004)	0.020 (0.004)
Rhinopristiformes	0.191 (0.032)	0.051 (0.010)	0.054 (0.011)	0.013 (0.003)
All sharks	0.009 (0.003)	0.100 (0.019)	0.110 (0.019)	-
All species	0.067 (0.007)	0.159 (0.015)	0.026 (0.003)	0.024 (0.007)

Catch efficiency (Q) was estimable for only two species, *Aptychotrema rostrata* and *Trygonoptera testacea* (Table 7). Generally, catch sampling was inadequate to quantify Q at the species level and, as a result, species were grouped at increasingly higher classification levels until the models were able to produce satisfactory estimates of catch, compared to the observed values. This was particularly the case for sharks, which were relatively uncommon. As such, sample size restricted the estimation of Q to Myliobatiformes, Rhinopristiformes and "All sharks", which included individuals from the following taxonomic orders: Carcharhiniformes, Heterodontiformes, Orectolobiformes, Squaliformes and

Squatiniformes. Finally, a gear-specific single Q was estimated by combining samples for all species. Too few sharks were caught in scallop trawls to estimate Q for this species group and, similarly, Q was inestimable for *T. testacea* in scallop trawls. In these cases, the Q derived for penaeid trawls was used to calculate F for scallop trawls, given the similarity of the two gear types.

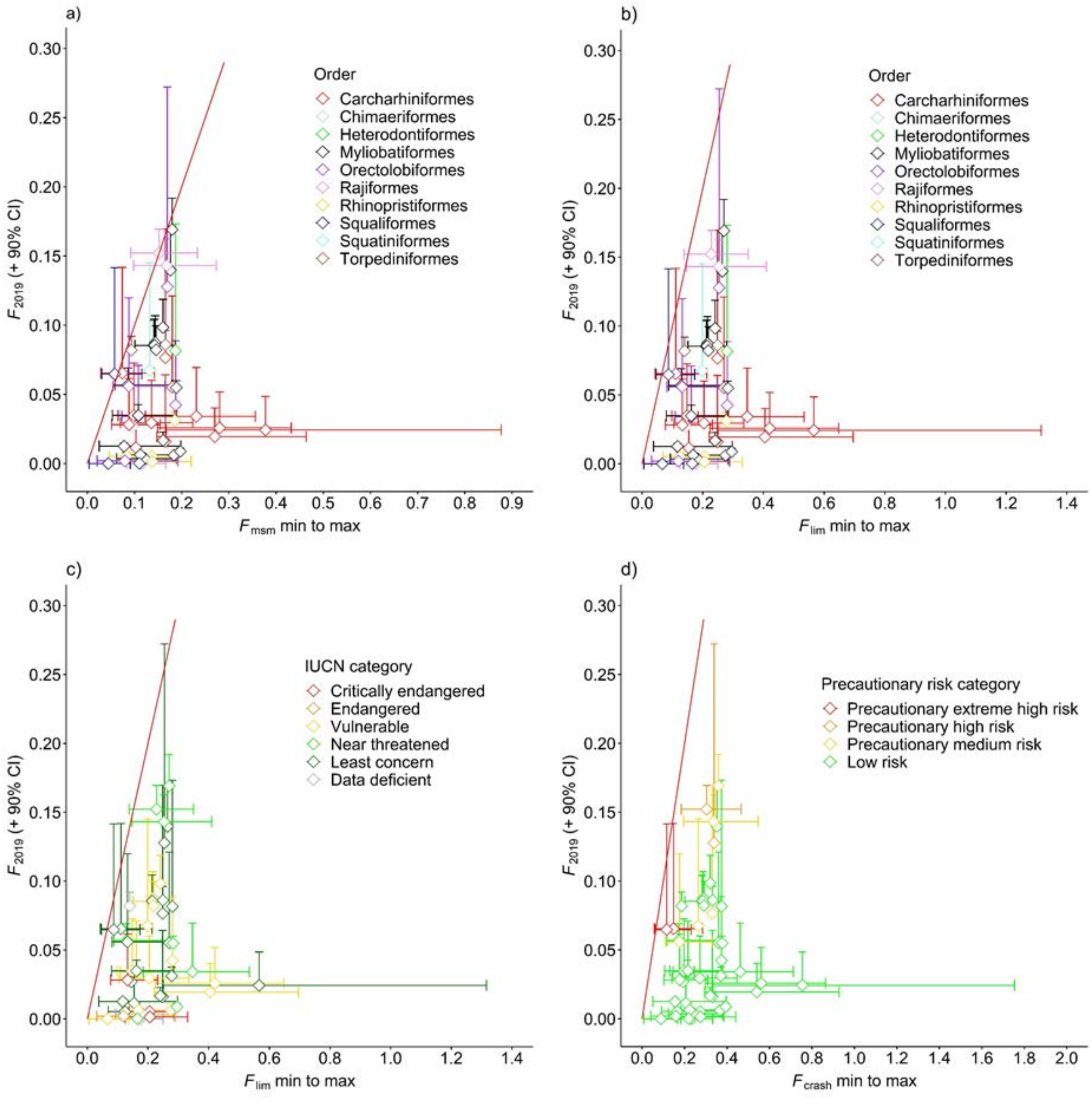

Figure 10: Comparison of the fishing mortality (F_{2019}) of 48 chondrichthyans occurring in southern Queensland to: a) F_{msm} as a function of taxonomic order, b) F_{lim} as a function of taxonomic order, c) F_{lim} as a function of IUCN Redlist category, and d) unsustainable fishing mortality (F_{crash}) as a function of precautionary risk category. The red line represents the 1:1 line of equivalence.

Estimates of escape were only possible for 20 species (Table 16). Escape ranged from 93.5% for *Stegostoma tigrinum* to –62.8% for *Hemigaleus australiensis*. Nets equipped with TEDs were found to reduce the catch of only ten of the 20 species, compared to standard nets. The catch of *Neotrygon* *trigonoides* decreased by 16.9% in the presence of bottom-shooter TEDs reduced but increased by 3.2% where top-shooter TEDs were used. In contrast, catches of *Carcharhinus coatesi* decreased by 15% when top-shooter TEDs were used and increased by 5.3% when bottom-shooter TEDs were used. The catch of the remaining eight species increased in the presence of TEDs.

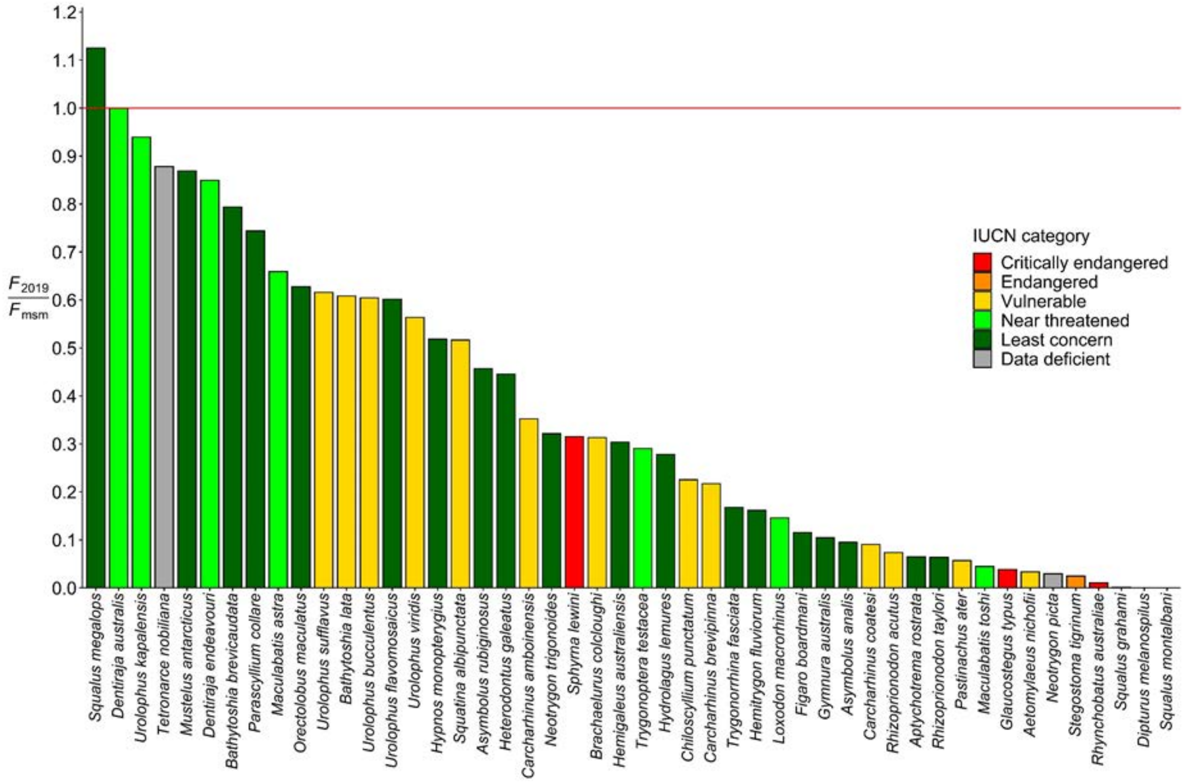

Figure 11: Fishing mortality in the 2019 fishing year (F_{2019}) as a proportion of mean maximum sustainable fishing mortality (F_{msm}) for the 48 chondrichthyans interacting with the QECOTF in southeast Queensland. Also shown are the IUCN Redlist categories of each species. The red horizontal line represents $F_{2019}/F_{msm} = 1$.

Estimates of post-trawl survival (*S*) were available for only 12 species (Table 16). The post-release survival rate estimates of *T. testacea* and *A. rostrata* reported by Campbell *et al.* (2018) were the only experimentally derived estimates used herein. The remaining estimates are at-vessel estimates published by Stobutzki *et al.* (2002) and from unpublished data collected during previous research (Courtney *et al.*, 2007).

Of the 48 species assessed, only 25 had published life history parameters (Table 15). A single estimate of F_{msm} was possible for 23 species (Figure 10). Values of F_{msm} were lowest for two squalids, *Squalus montalbani* (F_{msm} = 0.044) and *S. megalops* (F_{msm} = 0.058), and highest (F_{msm} > 0.2) for the carcharhinids *Rhizoprionodon taylori*, *Carcharhinus coatesi*, *Rhizoprionodon acutus* and *Loxodon macrorhinus* (Table 14).

Fishing mortality ranged between $F_{2019} = 0$ and $F_{2019} = 0.169$ year^{-1} (Table 14). Due to the lack of trawl effort that occurred within the distributions of *S. montalbani*, *S. grahami* and *Dipturus melanospilus* in the 2019 fishing year, $F_{2019} = 0$ for these species (Table 16). Fishing mortality was low for *A. rostrata*, *S. tigrinum*, *Rhyncobatus australiae* and *Glaucostegus typus* due to high *S* and/or *E*. Fishing mortality was highest for *Urolophus kapalensis* ($F_{2019} = 0.169$ year^{-1}), *Dentiraja australis* ($F_{2019} = 0.152$ year^{-1}) and *D. endeavouri* ($F_{2019} = 0.143$ year^{-1}).

The fishery posed low risk to all but two species (i.e., $F_{2019} < F_{\text{msm}}$, Figure 10, Figure 11, and Table 14). Only *S. megalops* and *D. australis* were found to be at medium risk (i.e., $F_{\text{lim}} > F_{2019} > F_{\text{msm}}$); however, $F_{2019}/F_{\text{msm}} = 0.940$ for *U. kapalensis* (Figure 11). The instantaneous fishing mortality rate did not exceed unsustainable levels (i.e., $F > F_{\text{crash}}$) for any species. When assessing the precautionary risk posed to the 48 species, the fishery posed precautionary extreme high risk to *S. megalops* and *Mustelus antarcticus*, and precautionary high risk to *Parascyllium collare* and *D. australis*. Further, the fishery posed a precautionary medium risk to five species: *Orectolobus maculatus*, *Squatina albipunctata*, *Asymbolus rubiginosus*, *D. endeavouri* and *U. kapalensis*. The fishery posed a low precautionary risk to all remaining species.

6.4 Discussion

The results indicate that the QECOTF posed a low risk to the sustainability of all but two of the species assessed in the study area in the 2019 fishing year. The fishery posed low risk to all 17 species globally threatened with extinction, classified as Vulnerable, Endangered or Critically Endangered by the IUCN Redlist. Overfishing affects over 99% of the world's threatened chondrichthyans due to capture in fisheries targeting other species (Dulvy *et al.*, 2021). These authors reported that, of the 1199 species assessed as part of the IUCN Redlist, 391 (32.6%) are threatened with extinction (Vulnerable, Endangered or Critically Endangered). The sole threat to 267 of these 391 species is overfishing. To reduce the mortality of threatened species, arrest declines and enable recovery, Dulvy *et al.* (2021) proposed the imposition of regulatory objectives that: 1) restrict catch, 2) protect at-risk species, 3) minimise incidental catch, 4) improve post-release survival, and 5) provide refugia across significant portions of each species' distribution. Additionally, Kyne *et al.* (2020) suggested prohibiting the retention of shark products as a measure to mitigate risk for Rhinobatiformes, a strategy likely to benefit all chondrichthyans.

The possession of shark products by fishers operating in the QECOTF was prohibited in 2001, thus significantly reducing the level of fishing mortality from this sector to at-risk species, as recommended by Dulvy *et al.* (2021). The prohibition was one of a suite of management changes introduced between 2000 and 2003. During this time, a government buy-back resulted in a reduction in the number of vessels operating in the QECOTF, a six-week closure was introduced for waters < ~91 m in depth (50 fathoms) south of 22°S and an effort unitisation system limited the number of nights each vessel could fish annually (Helidoniotis *et al.*, 2020). These measures resulted in significant reductions in nominal effort, particularly in the shallow water EKP and scallop sectors (see Figure 9, Wang *et al.*, 2020), which account for the highest catch of chondrichthyans in the QECOTF (Courtney *et al.*, 2007). Further, as a result of low spawning biomass (Wortmann, 2021), fishery managers have significantly reduced fishing effort in the saucer scallop fishery since 2016, further reducing the discard mortality of chondrichthyans.

Prior to the prohibition of the retention of shark products, logbook data reveal an average of 1.5 t per calendar year (range: 0–4.8, S.D = 1.4) of chondrichthyans (all taxa) was landed in the period 1990–2001 by trawl fishers operating within the study area. However, under-reporting of chondrichthyan landings during this period is likely to have occurred as these species were of little economic value compared to targeted species. It is, therefore, difficult to quantitatively assess the effect of management interventions prohibiting shark products on the risk posed to chondrichthyans by the QECOTF. The changes, however, likely had positive impacts on the chondrichthyan populations in the study area, especially given the relatively high prices commanded for shark fins in recent years (D'Alberto *et al.*, 2019), which would have encouraged the retention of shark products, possibly increasing fishing mortality.

Although the retention of shark products by the QECOTF has been prohibited since 2001, chondrichthyans are subjected to fishing mortality from other sources in Queensland. An in-possession (bag) limit of one chondrichthyan of any species <1.5 m TL applies to recreational fishers and the retention of any species >1.5 m TL is prohibited. Survey data indicate a total of 77,000 chondrichthyans (all taxa) were caught by 660,000 recreational fishers on the east coast of Queensland between April 2019 and April 2020, 75,000 of which were released (data available at: https://www.daf.qld.gov.au/business-priorities/fisheries/monitoring-research/monitoring-reporting/statewide-recreational-fishing-surveys/dashboard). No information is available for the recreational catch specific to the study area. In the 2019 calendar year, commercial net fishers landed a total of 52.3 t of chondrichthyans in the study area, 90% of which were carcharhinids; *C. brevipinna* accounted for 28.2% of the total catch, whereas Rhinopristiformes accounted for <2% of total catch. A bather protection program, the Queensland Shark Control Program (QSCP), deploys nets and drumlines in Queensland with the aim of reducing the risk of interactions between bathers and dangerous sharks, primarily white (*Carcharodon carcharias*), tiger (*Galeocerdo cuvier*) and bull (*Carcharhinus leucas*) sharks. In the 2019 calendar year, the QSCP caught 179 individual sharks in the study area, including 45 *C. brevipinna*, 13 *S. lewini*, six *S. tigrinum*, three Rhinopristiformes and one *C. amboinensis*, many of which were dead on retrieval. The catch data from the QSCP is reliable; however, species identification issues are likely in both the commercial net and recreational fisheries. As such, it is difficult to determine cumulative risks posed to the species assessed herein from all fisheries and further work is required to ensure fishing mortality from all sources is sustainable for all species in the long term.

The mandatory use of TEDs in the QECOTF since 2001 has reduced the incidental catch of chondrichthyans, as recommended by Dulvy *et al.* (2021). Turtle excluder devices have been shown to decrease the catch of chondrichthyans in a range of penaeid-trawl fisheries (Fennessy and Isaksen, 2007; Garstin and Oxenford, 2018; Noell *et al.*, 2018), mitigating the risk to larger species (Griffiths *et al.*, 2006; Stobutzki *et al.*, 2002; Zhou and Griffiths, 2008). The escape of smaller species and small individuals of large species, however, is poor and the presence of a TED can result in increased catches of some species (Campbell *et al.*, 2020), resulting in the negative values of E in the current study. Reducing bycatch, via TEDs and other bycatch reduction devices, decreases codend drag throughout a trawl, allowing wing-end spread to be maintained (Eayrs, 2007). This has the counterintuitive effect of increasing the catch of smaller chondrichthyans, as reported by Campbell *et al.* (2020), due to an increase in the area swept by nets with the devices installed, compared to nets without the devices (Eayrs, 2007).

The bar spacing is the primary driver of escape from TEDs; as bar space decreases, escape increases (Belcher and Jennings, 2011; Campbell *et al.*, 2020; Garstin and Oxenford, 2018; Noell *et al.*, 2018). Current regulations require a maximum bar space of 120 mm in the QECOTF, which allows a high proportion of small chondrichthyans to pass through the TED and into the codend (Kyne *et al.*, 2002). However, a reduction in bar space is a controversial topic among QECOTF fishers as there is a perception that a reduction from 120 to 100 mm would result in significant loss of targeted penaeids and Moreton Bay bugs (*Thenus orientalis* and *T. parindicus*). This perception is currently unsubstantiated and further research is required to quantify the effects of reducing bar space on the marketable component of catch in this fishery, as well as chondrichthyan and other bycatch components.

Escape could not be quantified for 28 of the 48 species assessed herein due, in part, to a lack of available size information and further sampling is required to determine their median size. This is especially important for the 15 species from the four taxonomic orders where escape is estimable via the models derived by Campbell *et al.* (2020). Further research is also required to develop similar models of escape for the remaining taxonomic orders, particularly the Squaliformes and the Rajiformes which include *S. megalops* and *D. australis*, respectively. The predominant use of bottom-shooter TEDs in the current study facilitates the higher escape of Myliobatiformes, compared to top-shooter TEDs (Campbell *et al.*, 2020) and is likely to facilitate the higher escape of Rajiformes, Torpediniformes and bottom-dwelling sharks of conservation interest such as *Brachaelurus colcloughi* (IUCN Redlist-designated Vulnerable) and *Chiloscyllium punctatum* (Near Threatened). Escape estimates in the current study are reasonable given the information available, although further research is required to quantify species-specific escape, particularly for species of conservation interest.

Improving the post-release survival of incidentally caught chondrichthyans is the fourth measure recommended by Dulvy *et al.* (2021) to reduce fishing mortality. Although post-trawl survival (PTS) is species-specific, fishers can increase the survival of released chondrichthyans by altering fishing practices such as reducing trawl duration (Fennessy, 1994) and catch size (Enever *et al.*, 2010), and minimising air exposure by returning individuals to the sea as quickly as possible (Campbell *et al.*, 2018; Cicia *et al.*, 2012). Of the 48 species assessed herein, PTS estimates were available for only 12 species. Apart from the PTS rates for *A. rostrata* and *T. testacea*, quantified by Campbell *et al.* (2018), the survival rates are immediate or at-vessel survival. The at-vessel estimates should be regarded as maxima given they represent the survival upon capture, and ignore any delayed effects known to affect survival (e.g. Kaiser and Spencer, 1995; Van Beek *et al.*, 1990; Wassenberg and Hill, 1993). Enforcing prescribed rules that increase PTS is difficult and information outlining best practice handling should be provided to fishers through dedicated education and extension programs, or through observer programs, where in operation.

The final measure proposed by Dulvy *et al.* (2021) to reduce fishing mortality of at-risk chondrichthyans is the provision of refugia via the introduction of marine protected areas (MPAs). Two separate marine parks occur within, and adjacent to, the fishery area, both of which are designed to maintain or enhance biodiversity. The Great Sandy Marine Park (GSMP) and the Moreton Bay Marine Park (MBMP) were established in 2006 and 2009, respectively, and include inshore areas that are closed to all forms of fishing. These marine parks protect a wide range of habitats, some of which are known to support significant populations of chondrichthyans (Gutteridge *et al.*, 2013; Jacobsen and Bennett, 2011; Kyne and Bennett, 2002b; Pierce *et al.*, 2011). However, the area closed to all forms of fishing in the study area is ~540 km^2, representing just 2% of the fishery assessed. Given their small size and proximity to the coast (Figure 9), the closed areas are unlikely to have any significant benefits to the abundance of chondrichthyans (Edgar *et al.*, 2014; Kyne *et al.*, 2021). The lack of connectivity between the closed areas exposes individuals to fishing, reducing any benefit to transiting species (Chapman *et al.*, 2005).

The results of the current study provide further evidence that Australia is successfully implementing regulations that protect chondrichthyans from overexploitation (Dulvy *et al.*, 2017). Six species assessed as part of the current study are globally threatened with extinction (*S. montalbani*, *S. tigrinum*, *C. brevipinna*, *R. acutus*, *R. australiae* and *G. typus*), however, these species are not threatened in Australian waters (Kyne *et al.*, 2021). These six species are among 45 'lifeboat' species that are at low risk from fishing in Australia due to well-managed fisheries, along with low levels of fishing effort when compared to adjacent parts of the Indo–West Pacific (Kyne *et al.*, 2021). In contrast to the global assessment concluding that one-third of chondrichthyans are threatened with extinction (Dulvy *et al.*, 2021), only 11.9% (22 sharks, 17 rays, 0 chimaeras) of the 328 chondrichthyans occurring in Australia are faced with this threat (Kyne *et al.*, 2021). The presence of the World Heritage-listed GBRMP (Figure 9) within the boundaries of the QECOTF has necessitated management changes to reduce the catch of chondrichthyans and other bycatch components. These management changes have resulted in a reduction in discards by approximately 69% since the late 1990s (Wang *et al.*, 2020) and a concomitant reduction in fishing mortality on chondrichthyans in southern Queensland since the late 1990s.

Life history data were available for 22 of the 48 (~46%) species assessed. Of these 22 species, life history characteristics for only eight species were derived from samples collected within the study area. This represents a source of uncertainty in assessing risk that requires urgent attention as growth is known to vary both spatially (Moulton *et al.*, 1992) and temporally (Carlson and Baremore, 2003). For example, the growth parameters for *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. This species may grow faster in the warmer waters of southern Queensland resulting in an increased resilience to fishing mortality. Improvements to, and validation of, life history characteristics have been identified as areas of further research when determining ecological risk (Zhou *et al.*, 2016). The lack of life history information for most species in the current study reinforces the need for additional resources focusing on estimating region-specific growth for species interacting with fishing gear in this and other fisheries (Kyne *et al.*, 2021; Taylor *et al.*, 2016).

An additional source of uncertainty in the current study is the estimation of catch efficiency, Q. Estimates of Q for penaeid and scallop trawls were primarily derived from data collected in 2001 and 2002, whereas the estimates of Q for the Danish seine and fish trawl gear were derived from data collected between 2009 and 2010. Although Zhou and Griffiths (2008) found that bycatch populations were stable over a 20 year period in the NPF, reduced fishing effort, prevailing biological and physical oceanographic conditions, and climate change, are likely to impact the abundance of the species assessed herein. Further, the observed catches of each species were generally low during the sampling, necessitating the grouping of species into higher taxonomic classifications to ensure adequate non-zero catches for model fitting when estimating Q. Low sample size could bias Q estimation; Zhou and Griffiths (2008) reported catch efficiency estimates informed by Blaber *et al.* (1990), which were significantly higher than those used herein. Additionally, the sampling undertaken was not intended for the purposes of calculating Q and some chondrichthyans may have been overlooked when catch samples were obtained, particularly during the opportunistic, observer-based sampling described by Courtney *et al.* (2007) and Rowsell and Davies (2012). This may have resulted in the underestimation of Q and the discrepancy between the values of Q derived herein and those reported by Zhou and Griffiths (2008). The underestimation of Q will result in the underestimation of fishing mortality (F_{2019}) and risk. Further sampling, predominantly by way of a dedicated fishery observer program, is required to improve estimates of Q for future assessments.

Despite the deficiencies in some of the metrics used to determine risk in the current study, the SAFE method has been shown to be superior to qualitative ERAs (Zhou *et al.*, 2016). Generally, qualitative assessments are more conservative, resulting in more species being classified at medium or high risk. This is the case for a qualitative assessment by Jacobsen *et al.* (2018) of the risk posed to chondrichthyans by the QECOTF in the study area. These authors reported the fishery posed high risk to 15 chondrichthyans, all but one of which (*D. australis*) were classified as low risk in the current study. A distinct advantage of the SAFE method used in the current study was the availability of high-resolution trawl track information. These data facilitate the accurate calculation of trawl area and fishing effort within the distribution of each species assessed, which was absent in previous qualitative assessments. For example, Pears *et al.* (2012) reported that the QECOTF posed a high risk to *Dipturus apricus*, despite no trawling occurring within the species' published distribution (mainly in depths between 300 and 500m, Last and Stevens, 2009).

Increasing water temperature has been shown to reduce the suitability of habitats at the equatorward boundary of some chondrichthyans (Dulvy *et al.*, 2021). Along with the poleward movement commonly associated with climate change (Hammerschlag *et al.*, 2022; Osgood *et al.*, 2021), there is also evidence that some species respond to increasing temperatures by moving into deeper waters (Perry *et al.*, 2005). Such a shift may reduce fishing mortality for some species assessed in the current study, as waters deeper than 250 m generally offer refuge from trawling. For example, *S. megalops* is found to 580 m water depth and, as such, a significant proportion of this species' distribution is unaffected by trawling south of the GBRMP. This is consistent with Walls and Dulvy (2021), who found that deeper waters provided refuge from fishing for sharks and rays in the Northeast Atlantic Ocean and the Mediterranean Sea. Conversely, an eastward shift into depths >100 m of, for example, *T. testacea*, would result in significant increases in fishing mortality given the fishing pressure at these depths between 26°30'S and 27°15'S (Figure 9).

In conclusion, this study showed that the QECOTF posed low risk to all but two of the chondrichthyan populations in southern Queensland and most species are sustainable at 2019 levels of fishing effort in the long term. Management changes in the early 2000s, which mandated the use of TEDs in the QECOTF and significantly reduced the number of licensed operators, have reduced the risk of overfishing, particularly in the penaeid prawn and scallop sectors where the abundance of chondrichthyans is greatest. Further research is required to improve the estimates of both escape and post-release survival for most of the species assessed. Fishers can reduce the fishing mortality of chondrichthyans by reducing TED bar spacing and altering fishing practices to avoid excessive post-release mortality, such as decreasing trawl duration and minimising the air exposure of captured fish. The lack of region-specific life history information for most of the species assessed represents a source of uncertainty when estimating natural mortality and requires immediate attention, particularly for *S. megalops* and *D. australis*. Furthermore, subsequent assessments should be undertaken if future stock assessments indicate the recovery of the saucer scallop stock and FQ allow fishers to target scallops. Such a decision will result in increased trawled area in and around Hervey Bay, with concomitant increases of the fishing mortality of chondrichthyans.

7. General Discussion and Conclusions

Plate 7: The FV *C–Rainger* at the Southport wharves.

7.1 Discussion

This study demonstrates that the QECOTF poses low risk to all but two of the chondrichthyan species assessed in the study area (Figure 9) and medium risk to *S. megalops* and *D. australis*. In the period 2000–2003, management measures were introduced in Queensland that reduce the fishing mortality of at-risk chondrichthyans species. In 2001, the retention of chondrichthyan products by fishers operating in the QECOTF was prohibited, and the use of TEDs was made mandatory. Other changes, including a vessel buy-back, a six-week fishing closure in waters <91 m (September 20 to November 1, annually), and effort unitisation to limit annual fishing effort, were implemented to reduce the impacts of penaeid trawling in Queensland. These changes resulted in a 69% decrease in annual discards from 77,000 t in the late 1990s to 21,000 t in the period 2011–2014 (Wang *et al.*, 2020). A concomitant reduction in the catch of chondrichthyans over this period is a reasonable assumption. Further, significant reductions in fishing effort have occurred in the saucer scallop trawl fishery since 2016 due to low spawning biomass levels, which have also likely reduced fishing mortality in both the scallops and those species caught incidentally, including chondrichthyans.

The SAFE method produces more accurate measures of risk, compared to qualitative methods such as the Productivity and Susceptibility Analysis (PSA) used by Stobutzki *et al.* (2001a) (Zhou *et al.*, 2016). In the current study, high resolution trawl track information was used to quantify fishing effort within each species' distribution, which accurately calculates the level of interaction with the fishery. In contrast, Jacobsen *et al.* (2018) used terms such as 'some contact', 'moderate contact', and 'significant contact' to categorise the level of interaction between chondrichthyans and the QECOTF in the study area using PSA. These authors further used two qualitative measures of TED effectiveness ('effective' and 'not effective') and three qualitative measures of PTS ('good survival', 'moderate survival', and 'low survival'). Using these qualitative categories, Jacobsen *et al.* (2018) found the fishery posed high risk to 15 chondrichthyans in the study area. Conversely, all but one (*D. australis*) of the 15 chondrichthyan species deemed to be high risk by Jacobsen *et al.* (2018) were classified as low risk herein. This difference can be attributed to the increased information used to assess risk in the current study. For example, the high PTS of *A. rostrata* quantified in Chapter 5 significantly reduced the risk posed to this species by the QECOTF.

Post-trawl survival is unknown for the two species, *D. australis* and *S. megalops*, found to be at medium risk. Estimates of PTS were available for only 12 of the 48 species assessed herein and two of these were derived via dedicated short-term experiments. Conducting PTS experiments is expensive and logistically challenging (Dapp *et al.*, 2016). To overcome these issues, some authors have advocated for assessing survival as a function of capture condition (Benoît *et al.*, 2010); however, this metric can lead to erroneous PTS estimates. For example, 50% of the *A. rostrata* and 18.2% of the *T. testacea* categorised as 'dead or nearly dead' survived three days post-capture (Table 5). Further, 90.5% of *A.*

rostrata and 38% of *T. testacea* categorised with "vigorous wing and/or body movement; rapid spiracle movement" on capture, survived. This highlights the obvious need for experimentally derived, species-specific measures of PTS to ensure the accurate assessment of risk, despite the logistical and financial constraints. The two species found to be at medium risk, *S. megalops* and *D. australis*, co-occur and it is feasible to undertake PTS experiments on-board vessels under commercial conditions. Holding tanks should be fitted to appropriately-sized commercial vessels and scientists could assist fishers in developing robust experimental design to quantify PTS. Such an approach would result in cost-effective, empirically derived estimates of PTS, while providing interested fishers with ownership of results and increase their confidence in outputs. This approach would also demonstrate industry's commitment to the long-term sustainability of chondrichthyans.

Although the survival of chondrichthyans discarded from penaeid trawls is poorly understood, survival of chondrichthyans from trawl fisheries targeting fish has been studied (Kaiser and Spencer, 1995; Mandelman *et al.*, 2013; Mandelman and Farrington, 2007a). These studies demonstrate that PTS is species-specific and is affected by a range of factors such as trawl duration, time-on-deck, and catch weight. Fishers can, therefore, improve the survival of discarded chondrichthyans by returning individuals to the sea as quickly as practicable. Reducing trawl duration, resulting in smaller catch volumes, decreased sorting times and less time-on-deck will improve the survival of discarded individuals, although this may adversely affect operations and profit.

The estimates of escape derived in Chapter 4 are quantitative measures as a function of fish size. The models estimate the escape of species classified within four taxonomic orders: Carcharhiniformes, Myliobatiformes, Rhinopristiformes and Orectolobiformes. Provided a median fish size is known, the escape of any species is quantitatively estimable, along with a measure of variability, from both top- and bottom-shooter TEDs. The addition of TED orientation when quantifying fishing mortality is a significant improvement over previous quantitative ERAs. For example, Zhou and Griffiths (2008) assumed escape was zero for 20 species, whereas escape is estimable for each of these species with the models produced in Chapter 4. The escape of *S. megalops* and *D. australis* remains unquantified and efforts should be made to quantify escape for these species. Like PTS, quantifying escape through experimentation is costly, particularly for species that are rare; however, data can be generated in a cost-effective manner through collaboration between scientists and fishers.

The current maximum bar space in the QECOTF is 120 mm. A reduction to 100 mm, like that used in the south-east United States, would further reduce the capture of chondrichthyans. Reducing bar space, however, remains highly contentious among QECOTF operators, who are adamant that any decrease in bar space would result in significant loss of catch, particularly the scyllarid lobsters (*Thenus* spp. and *Ibacus* spp.) and large saucer scallops. The loss of these catch components can be quantified experimentally, in an approach similar to that used by Courtney *et al.* (2008) to test four identical TEDs simultaneously using a quad-rigged vessel. Bar space can be varied across the four identical TEDs, tested in each net position, to isolate the effects of bar space.

The assessment of risk is contingent on the estimation of natural mortality, which is generally correlated to the life history of individual species (Table 6). The life history information for most species assessed in Chapter 6 was derived during postgraduate research (Table 15) and this is a sensible pathway to quantifying life history for more species. There is scope for universities to work collaboratively with industry and government researchers in Queensland to obtain the samples required to improve life history metrics (Kyne *et al.*, 2021). Until this research is undertaken, the development of additional life history correlates, like those described in section 6.2.2, would improve future ERAs. The benefit of available life history correlates is demonstrated in Chapter 6: correlating k and L_{max} for Rajidae (Frisk *et al.*, 2001) led to the estimation of eight values of M (Table 15), despite the lack of life history information for these species.

The refugia distributed throughout the study area are unlikely to significantly reduce incidental fishing mortality on chondrichthyans. Edgar *et al.* (2014) found that marine protected areas should include at least four of the five following features to increase the biomass of chondrichthyans: no take, enforced, old, large and isolated (NEOLI). The areas closed to all fishing within the study area (Figure 9) are small and located close to the coast. Further, the closed areas are disconnected, exposing individuals to fishing as they move throughout the study area (Chapman *et al.*, 2005). The introduction of closed areas is a controversial measure (Caveen *et al.*, 2014), particularly in Queensland (Kenyon *et al.*, 2018), where affected fishers protested the increase in no-fishing zones from 0.5 to 16% within the Moreton Bay Marine Park in 2009. The intention of the closed areas within the study area was to protect or enhance marine biodiversity (Kenyon *et al.*, 2018) and not as a means of protecting fish stocks. Any expansion of the closed areas within the study area to specifically protect chondrichthyans would be difficult to argue given the low catch levels discussed in Chapter 6.

Performing a SAFE analysis is less data-intensive than a quantitative stock assessment: however, the SAFE requires more data than a PSA and is, consequently, more labour-intensive, and costly. A cost-effective approach to assess the risk posed to chondrichthyans by the QECOTF, or other fisheries, would be to undertake a PSA to derive a preliminary risk, before performing a SAFE for those species categorised at medium or high risk. For example, Jacobsen *et al.* (2018) found that the QECOTF posed high risk to 15 chondrichthyan species assessed within the same area as that assessed herein and 14 of these were found to be at low risk in Chapter 6. Pears *et al.* (2012) identified 11 chondrichthyan species as being at high risk within the GBRMP and it would be prudent to undertake a SAFE for these species to ensure that current levels of fishing mortality are sustainable.

7.2 Conclusion

This research has increased the scientific knowledge of two chondrichthyan species, *A. rostrata* and *T. testacea*. The growth and age-at-maturity of *A. rostrata* indicate this species is slow growing, long-lived and late to mature, life history strategies that are common among chondrichthyans. Prior to this study, only three of the eight species that comprise Trygonorrhinidae have published growth information and, as such, the current study increases the scientific knowledge of trygonorrhinids and the Rhinopristiformes more broadly. The under-sampling of larger, older individuals was overcome by using informative priors, reducing bias in both growth and maturity estimates. The models of escape developed herein enable quantitative estimates of escape via TEDs for any species classified within the Carcharhiniform, Myliobatiform, Orectolobiform and Rhinopristiform taxonomic orders. Provided a median size can be calculated for any species within these taxonomic orders, escape rates are estimable. Escape was found to increase with increasing fish size. Top-shooter TEDs increased the escape of the Carcharhiniformes and bottom-shooter TEDs increased the escape of Myliobatiformes. The post-trawl survival estimates of *A. rostrata* and *T. testacea* are the first experimentally derived, short-term (3 day) estimates published in the primary literature. The factors affecting PTS of *A. rostrata* and *T. testacea* are consistent with those affecting chondrichthyans caught in fish trawls in the northern hemisphere: fish size, sex, time-on-deck and tow duration were all found to affect PTS. The PTS of *A. rostrata* was at the upper bounds of published estimates, while the PTS of *T. testacea* was low. These results demonstrate that the assumption of zero survival where estimates are unavailable is clearly incorrect and experimentally derived estimates are required to ensure accurate estimates of risk. This research demonstrated the QECOTF posed medium risk to two species, *Squalus megalops* and *Dentiraja australis*, and low risk to the remaining 46 species, in the 2019 fishing year. *Aptychotrema rostrata* is the only species assessed where all relevant data are available for use in SAFE, and the lack of these data is a significant impediment to the accurate assessment of risk for the remaining species.

7.3 Fishery Implications and Recommendations

1. The results demonstrate that the level of incidental fishing mortality in the 2019 fishing year was unlikely to affect the long-term sustainability of 46 of the 48 species assessed. This allows fishery managers to demonstrate to the federal Department of Climate Change, Energy, the Environment and Water (DCCEEW), the department responsible for granting export approval, that the QECOTF poses low risk to the majority of chondrichthyans that interact with the fishery in the study area.
2. Two species, *Squalus megalops* and *Dentiraja australis*, were assessed to be at medium risk. This categorisation implies that the level of fishing mortality for the 2019 fishing year resulted in the overfishing of the population of these species within the study area. For both species, escape and PTS are unknown and it is recommended that these metrics be quantified to improve accuracy in the assessment of risk. It is important to note that, while the results indicate the fishery lowered

recruitment for both species, the level of fishing mortality did not present significant risk of extinction.

3. To ensure accuracy when assessing risk, estimates of PTS and escape are required for all species: however, cost and logistical constraints prohibit the assessment of these metrics via dedicated research charters for all species. As such, collaboration between researchers and industry is recommended to collect data during commercial fishing operations. This would also foster ownership of the findings, and confidence in subsequent decision making, by fishers. This strategy would be cost-effective and produce scientifically robust measures of PTS and escape. Furthermore, data generated from this sampling could also be used to improve estimates of catch efficiency (Q), which is a source of uncertainty in this study.
4. Quantifying the bar space that will maximise escape of chondrichthyans while maintaining the catch of target or permitted species, particularly large saucer scallops, blue swimmer crabs (*Portunus armatus*), Moreton Bay bugs (*Thenus* spp.) and Balmain bugs (*Ibacus* spp.) is required. The models developed in this study demonstrate that decreasing bar space will result in fewer chondrichthyans being caught. Multiple TEDs, with varying bar space, could be tested simultaneously to determine their effect on catch. Any regulatory change, resulting from such experiments, would be costly to fishers and a phase-in period would be required, should such a change be implemented.
5. Life history data derived from samples collected within the study area were available for only eight species. The lack of region-specific estimates of growth and age-at-maturity represent a source of uncertainty when assessing risk. Collaboration between post-graduate students, scientists and industry is recommended to improve the life history information of the species assessed and chondrichthyans in general.
6. Low spawning biomass of saucer scallops affected the level of fishing effort in the 2019 fishing year. This was particularly the case in Hervey Bay, where fishing effort has decreased since 2016. Should the saucer scallop biomass increase in future due to management intervention, and the level of fishing effort in the scallop fishery increase, a SAFE should be undertaken for the chondrichthyans assessed in this study to ensure the increase in effort is sustainable.
7. At present, the effects of climate change on the distribution of the species assessed herein is unknown. The collaborative sampling strategy recommended previously would provide information to determine any geographical shifts resulting from a change in climate. Shifts to deeper water due to increasing water temperature would be beneficial for some species and detrimental for others. The effects of these changes should be quantified in future studies.
8. In future assessments of risk to chondrichthyan populations distributed throughout Queensland, it is recommended that a PSA be undertaken to provide preliminary estimates of risk. Any species found to be at either medium or high risk via a PSA should then be assessed using the SAFE

method. This is recommended as a cost-effective method to assess risk and identify species for which further research is required.

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Appendix - Supplementary figures and tables

Chapter 2

Table 8: Summary of the studies published in the primary literature reviewed to determine the factors affecting escape of chondrichthyans via turtle excluder devices (TEDs) and bycatch reduction devices (BRDs) from otter trawls. Type denotes the study type where E is experimental trawls and C is commercial trawls.

Author	Species of interest	Type	Target species	Area	TEDs tested (bar space, mm)	Grid orientation	Effects
Belcher and Jennings (2011)	*Rhizoprionodon terraenovae*, *Carcharhinus isodon*, *Carcharhinus brevipinna, Carcharhinus limbatus, Sphyrna lewini*, *Sphyrna tiburo*	C	Penaeid prawns	Georgia, USA	Super Shooter and Georgia jumper (102)		None: flat trawls resulted in lower catch rates
Brčić *et al.* (2015)	*Galeus melastomus*	E	*Nephrops norvegicus*	Italy	Super Shooter (90)		Only larger individuals excluded
Brewer *et al.* (1998)	*Maculabatis toshi, Pastinachus sephen, Rhynchobatus djiddensis, Rhinobatus typus, Anoxypristus cuspidata*	E	Penaeid prawns	Northern Australia	Super–Shooter, Nordmore, AusTED, NAFTED, Fisheye BRD, Radial Escape Section BRD, Square mesh panel BRD (90 – 120+)	Both	Trawls with TEDs caught 11 large (>5kg) chondrichthyans while standard nets caught 57.
Brewer *et al.* (1998)	*Maculabatis toshi, Pastinachus sephen*	C	Penaeid prawns	Northern Australia	Super–Shooter TED,	Both	Trawls without TEDs caught 12 large (>5kg) sharks and no large stingrays, while

					NAFTED (90 – 120+)		standard nets caught 32 large sharks and 17 large stingrays.
Brewer *et al.* (2006)	Various. Mostly (64%) *Carcharhinus tilstoni*, *Carcharhinus dussumieri*, *Rhynchobatus djiddensis*, *Gymnura* australis, *Himantura toshi*	C	Penaeid prawns	Northern Australia	Various including Super Shooter, NAFTED, Nordmore (90–120+)	Both	Large (>1m) sharks and rays: 86% and 94% exclusion, respectively. Small (<1m) sharks and rays: 4.9% and 25% exclusion, respectively. Top-shooters: 20% of sharks and 26.9% of rays excluded. Bottom-shooters: 8.8% of sharks and 34.8% of rays.
Courtney *et al.* (2014)	*Dentiraja endeavouri, Asymbolus rubiginosus, Figaro boardmani*	E	*Melicertus plebejus*	Queensland, Australia	Wicks TED (120)	Top	No significant difference in catch rates between control nets and nets containing TEDs
Courtney *et al.* (2008)	*Aptychotrema rostrata, Neotrygon trigonoides, Neotrygon leylandi*	E	Saucer scallop *Ylistrum balloti*	Queensland, Australia	Bent bar Wicks TED (120)	Top	No significant difference in catch rates between control nets and nets containing TEDs
Courtney *et al.* (2006)	*Aptychotrema rostrata, Urolophus kapalensis, Trygonoptera testacea*	E	*Melicertus plebejus*	Queensland, Australia	Wicks TED (120)	Top	No significant difference in catch rates between control nets and nets containing TEDs
Fennessy and Isaksen (2007)	None described	E	Penaeid prawns	Mozambique	Nordmore (100) with and without a square mesh panel	Top	Square mesh panel had no effect on the catch of chondrichthyans. Significant reduction of large sharks (>70cm PCL) and rays (>30cm disc width)
Garcıa–Caudillo *et al.* (2000)	*Rhinobatus productus*	E	Penaeid prawns	Gulf of California, Mexicco	Super-Shooter (100) plus Radial Escape Section BRD	Bottom	Significant reduction (37%) of *R. productus*
Gorman and Dixon (2015)	Sharks and rays	C	*Melicertus latisulcatus*	South Australia	Modified Wicks TED (50)	Top	73% reduction of large (>1m) rays and 86% reduction of large sharks

Isaksen *et al.* (1992)	*Somniosus microcephalus*, skates (Rajidae)		*Pandalus borealis*	Norway	Rectangular (19)	Top	All *S. microcephalus* excluded. No skates were excluded.
Jaiteh *et al.* (2014)	Various species	C	Lutjanids	North-west Australia	Oval, semi-rigid (150)	Bottom	40°- sharks: 63% reduction; rays: 53% reduction. 70°- sharks: 39% reduction; rays: 17% reduction
Kendall (1990)	"Small" sharks and rays, *Nequaprion brevirostris*, *Carcharhinus leucas*	C	*Penaeus aztecus, Penaeus setiferus, Penaeus duorarum*	Florida, USA	Morrison soft TED	Bottom	Control net caught more small sharks and rays that the TED net. Control net also caught one *N. brevirostris* and one *C. leucas*
Lomeli and Wakefield (2013)	*Raja binoculata, Raja rhina*	E	*Hippoglossus stenolepis*	Washington, USA	Soft TED (mesh size 19.1cm^2)	Bottom	*R. binoculata*: 94.5% reduction; *R. rhina*: 86.1% reduction
Lucchetti *et al.* (2016)	*Raja asterias*	E	*Scomber* spp., *Merluccius merluccius, Mullus barbatus, Melicertus kerathurus, Solea solea, Sepia officinalis*	Italy	Flexigrid (96)	Top	No significant reduction detected
McGilvray *et al.* (1999)	Large stingrays	E	Penaeid prawns	Queensland, Australia	AusTED II	Top	Unspecified number of large stingrays caught in the control net but none in the TED net.
Ordines *et al.* (2006)	*Raja miraletus, Raja radula, Scyliorhinus canicular, Leucoraja naevus, Raja brachyura, Raja clavate*	E	Multi-species fish trawl	Balearic Islands, Spain	Square mesh codend (40)	-	No significant reduction detected for any species
Queirolo *et al.* (2011)	Various species	E	*Heterocarpus reedi*	Chile	Nordmøre (35)	Top	39% across all species.
Raborn *et al.* (2012)	*Carcharhinus acronotus*, *Rhizoprionodon terraenovae*, *Sphyrna lewini*	E, C	Penaeid prawns	Gulf states, USA	Various (100)	Bottom	*C. acronotus*: 94%; *S. lewini*: 31%; *R. terraenovae*: no difference.

Robins–Troeger (1994)	*Glaucostegus typus, Neotrygon trigonoides*	E	Penaeid prawns	Queensland, Australia	Morrison soft TED (150)	Top	*R. batillum*, *N. trigonoides* and *Caretta caretta* absent from net fitted with the TED.
Robins–Troeger *et al.* (1995)	*Neotrygon trigonoides*	E	Penaeid prawns	Queensland, Australia	AusTED (150)	Top	Catch rate of *N. trigonoides* was significantly higher in the control net.
Robins and McGilvray (1999)	*Glaucostegus typus*, *Rhyncobatus australiae, Carcharhinus* sp.	E	Penaeid prawns	Queensland, Australia	AusTED (150)	Top	Three *G. typus* and one *R. australiae* caught in the control net. One "small" *Charcharhinus* sp caught in the TED net.
Wakefield *et al.* (2016)	Various species	C	Lutjanids	Western Australia	TED with curved edges top and bottom (150)	Both	Underwater video analysis. Benthic sharks: 80%; rays and skates: 66%; shark-like rays: 31%; benthopelagic sharks: 30%. Top-shooter TED: 20–30% reduction in both shark-like rays and benthopelagic sharks.
Willems *et al.* (2016)	*Dasyatis geijskesi*, *Dasyatis guttata*, *Gymnura micrura*, *Rhinoptera bonasus*, *Urotrygon microphthalmum*	E	*Xiphopenaeus kroyeri*	Suriname	Super-shooter TED (100)	Bottom	Overall: 36.1%; >80% for animals with a disc width >50cm; *D. geijskesi*: 76.6%; *D. guttata*: 40.2%; *G. micrura*: 32.1%.

Table 9: Summary of the studies published in the primary literature reviewed to determine the methods used and factors assessed when quantifying post-trawl survival of chondrichthyans.

Author(s)	Gear type	Species of interest	Area	Method	Time Held (hours)	Factors assessed	Best predictor(s) and (effect)	Survival
Benoît *et al.* (2010)	Fish trawl	Rajidae	Canada	On-board tanks	48	Catch weight, time-on-deck, depth, trawl duration, air temperature	Catch weight (-), air exposure (-), surface temperature (-)	Not stated explicitly but high.
Benoît *et al.* (2012)	Fish trawl	Rajidae	Canada	On-board tanks	14–110	Vitality level, time-on-deck	All animals with "excellent" or "good/fair" vitality score survived. Survival of animals with vitality score "moribund" was negatively correlated to time-on-deck	97%.
Benoît *et al.* (2013)	Fish trawl	*Leucoraja ocellata*, *Malacoraja senta*, *Amblyraja radiata*	Canada	AVM - Time to 50% mortality	-	Air temperature, body mass, depth, activity level	*L. ocellata*: body mass (+), depth (–) and temperature (–). *M. senta*: body mass and temperature. *A. radiata*: body mass, depth and temperature	Time to 50% mortality *L. ocellata*: 90 minutes; *M. senta*: 60 minutes; and *A. radiata*: 60 minutes.
Cicia *et al.* (2012)	Fish trawl	*Leucoraja erinacea*	New Hampshire, USA	Land-based tanks	120	Time-on-deck, temperature gradient (bottom temp. v surface temp)	Time-on-deck (–), temperature (–)	Winter: 100%, 82% and 73% for air exposure of <1 min, 15 min and 50 min, respectively. Summer: 63%, 14% and 0% for air exposure of <1 min, 15 min and 50 min, respectively.
Depestele *et al.* (2014)	Fish beam trawl	Rajidae	North Sea	On-board tanks	60	Catch weight, depth, salinity,	Length (+) and physical injury (–)	72%.

						air temperature, length, physical damage		
Enever *et al.* (2009)	Fish trawl	*Leucoraja naevus*, *Raja microocellata*, *Raja brachyura*, *Raja clavata*	United Kingdom	On-board tanks	72	Catch weight, sex, length, health status, depth, tow duration, position of tank in stack	Tow duration (–) and sex (♀+)	Commercial tows (3.6 hr): *L. naevus*: 33%; *R. microocellata*: 51%; *R. brachyura*: 55%; and *R. clavata*: 59%. Mean: 55%. Experimental trawls (0.8 hr): *R. brachyura*: 67%; and *R. clavata*: 91%. Mean: 87%.
Enever *et al.* (2010)	Fish trawl	*Leucoraja naevus*, *Raja microocellata*, *Raja brachyura*, *Raja clavata*	United Kingdom	On-board tanks	48	Catch weight, sex, length and health status	Catch weight (–) and sex (♀+)	80mm diamond mesh codend: 56%; 100mm diamond mesh: 59%; and 100mm square mesh codend: 65%.
Fennessy (1994)	Prawn trawl	Various species	South Africa	AVM	-	Depth, trawl duration, catch weight, length	Trawl duration (–), catch weight (–) and length (?)	*Mean = 57%.* *G. natalensis*: 53.6%; *Himantura* spp.: 57.4%; *D. chrysonata*: 82.3%; *H. uarnak*: 75%; *D. thetidis: 30%;* *S. lewini*: 2.4%; *M. mosis*: 71.4%; *C. brevipinna*: 44%; *R. acutus*: 70.8; *H. lineatus*: 80.8%; *A. leucospilus*: 47%; *R. djiddensis*: 81.8%; and *S. africana*: 40%.
Frick *et al.* (2010)	Fish trawl	*Heterodontus portjacksoni*, *Mustelus antarcticus*	Southern Australia	Trawl simulation	-	Trawl duration, air exposure and crowding	None	*H. portjacksoni: 100%*; *M. antarcticus*: 15% after 120 minute trawl, 100% after 60 minute trawl, 75% after 60 minute trawl and air exposure of 10 minutes

Heard *et al.* (2014)	Fish trawl	*Urolophus paucimaculatus*	Southern Australia	Trawl simulation	96	Trawl duration, air exposure and crowding	Air exposure increased plasma lactate concentrations	85%
Jaiteh *et al.* (2014)	Fish trawl	Sharks and rays, species unspecified	North-western Australia	AVM	-	None	None	Sharks: 9% and Rays: 34%
Kaiser and Spencer (1995)	Flatfish beam trawl	*Scyliorhinus canicula*, *Leucoraja naevus*	North Wales, UK	On-board tanks	144	None	None	*S. canicula*: 97% AVM and 94% after 120 hours; 100% AVM and 90% after 144 hours. *R. naevus*: 100% AVM, 59% after 120 hours.
Laptikhovsky (2004)	Squid trawl	Various species	Falkland Islands	On-board tanks	Up to 200 minutes	Depth, sex	Depth (–) and sex (♀+)	*B. albomaculata*: 71.4%; *B. brachyurops*: 54.6%; *B. griseocauda*: 0%; *B. macloviana*: 0%; *B. magellanica*: 60%; *Bathyraja* sp.: 75%; and *Psammobatis* sp.: 60%. Mean = 59.1%.
(Mandelman and Farrington, 2007a)	Fish trawl	*Squalus acanthias*	Massachusetts, USA	Pens anchored to sea floor	72	Length, sex, catch weight, tow duration	Catch weight (–)	71%
Mandelman and Farrington (2007b)	Fish trawl	*Squalus acanthias*	Massachusetts, USA	Land-based tanks	30 days	None	None	94.1%
Mandelman *et al.* (2013)	Fish trawl	*Amblyraja radiata*, *Malacoraja senta*, *Leucoraja ocellata*, *Leucoraja erinacea*	North-east USA	Pens anchored to sea floor	72	Tow duration, sex, length, catch weight, temperature change	*L. erinacea*: catch weight (–) and temperature change (–); *L. ocellata*: temperature change (–) and sex (♀+); *A. radiata*: catch weight (–) and length (+). Insufficient sample size for *M. senta*.	19% for all species combined.
Revill *et al.* (2005)	Fish beam trawl	*Scyliorhinus canicula*	Devon, UK	On-board tanks	36–60	None	None	98.3%

Rodríguez–Cabello *et al.* (2005)	Fish trawl	*Scyliorhinus canicula*	Spain	On-board tanks	1	Time-on-deck, tow duration, sex, length, depth	None	90% after 40 minutes time-on-deck and 30 minute tow duration. 78% after 36 minute time-on-deck and tow duration of 4.5 hours.
Rulifson (2007)	Fish trawl	*Squalus acanthias*	Massachusetts, USA	Pens anchored to sea floor	48	Tow duration	None	100%
Saygu and Deval (2014)	Fish trawl	*Raja clavata*, *Raja miraletus*	Turkey	On-board tanks	48	Catch weight, tow duration, length, sex, temperature	*R. clavata*: tow duration (–) and length (+); *R. miraletus*: catch weight (–), length (+), tow duration (–); both: length (+) and tow duration (–)	*R. clavata*: 81%; *R. miraletus*: 21%; both: 59%.
Stobutzki *et al.* (2002)	Prawn trawl	Various species	Northern Australia	AVM	-	Length, sex	Length (+), sex (♀+)	Sharks: ♀=78%, ♂=34; rays: ♀=44%, ♂=33%; *C. dussumieri*: ♀=52%, ♂=42%; *C. sorrah*: ♀=27%, ♂=50%; *C. tilstoni*: ♀=22%, ♂=15%; *R. acutus*: ♀=75%, ♂=14%; *N. leylandi*: ♀=73%, ♂=5%; *M. toshi*: ♀=57%, ♂=22%; *G. australis*: ♀=69%, ♂=25%; *H. microstoma*: ♀=56%, ♂=36%; *R. australiae*: ♀=56%, ♂=36%;

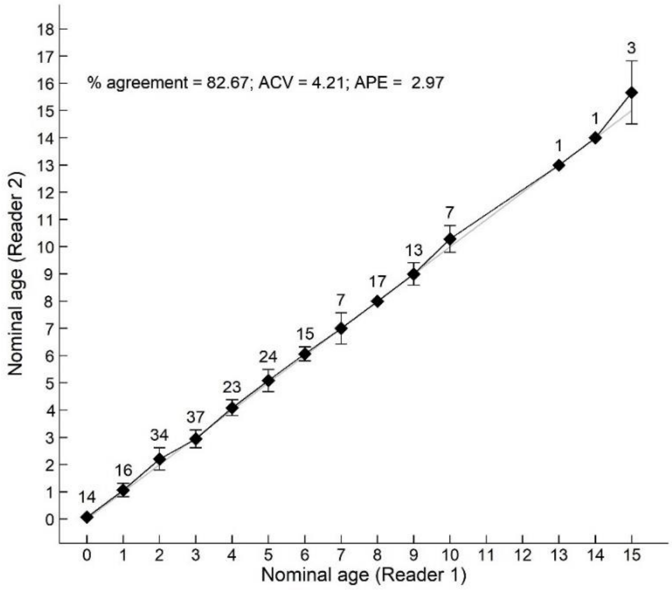

Figure 12: Age bias plot for two readers of 212 *Aptychotrema rostrata* centra. Also shown are relevant indices of agreement between the two readers. The grey line represents the line of equivalence. Numbers atop each point are the number of animals assigned the respective nominal ages.

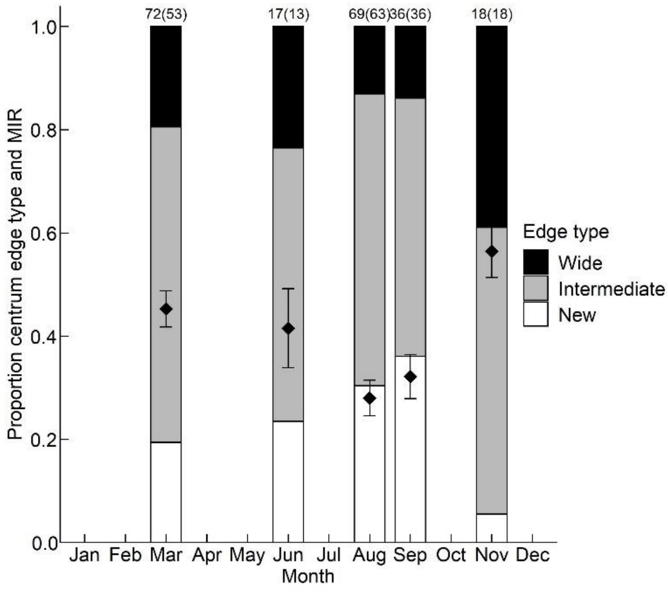

Figure 13: Variation in edge type and mean marginal increment ratio (MIR, ± S.E.) as a function of month for *Aptychotrema rostrata*. The number above each bar is the sample size for edge classification (total n = 212). The number in parentheses is the monthly sample size to assess MIR quantified for animals ≥2 years of age.

Table 10: Comparison of mean observed and mean back calculated lengths-at-age for individuals aged between one and ten. Those centra where a 'wide' edge occurred were excluded from the analyses. Note: *n* is the sample size; and *t* and *P* are the *t*-statistic and the *P*-value ($\alpha = 0.05$), respectively, from the two-sample *t*-tests where the hypothesised difference between the two means was zero.

Age	Observed		Back calculated		*t*	*P*
	n	mean	*n*	mean		
1	15	260.1	175	245.3	1.543	0.121
2	30	328.6	175	307.6	1.869	0.063
3	24	352.6	144	343.7	1.132	0.257
4	16	411.2	111	391.6	1.089	0.278
5	19	439.6	87	420.5	1.683	0.095
6	14	480.6	64	455.3	1.456	0.149
7	4	499.5	48	481.2	0.649	0.519
8	14	510.4	39	512.8	–0.139	0.89
9	9	522.1	22	539.9	–0.784	0.44
10	6	652.2	11	590	0.838	0.415

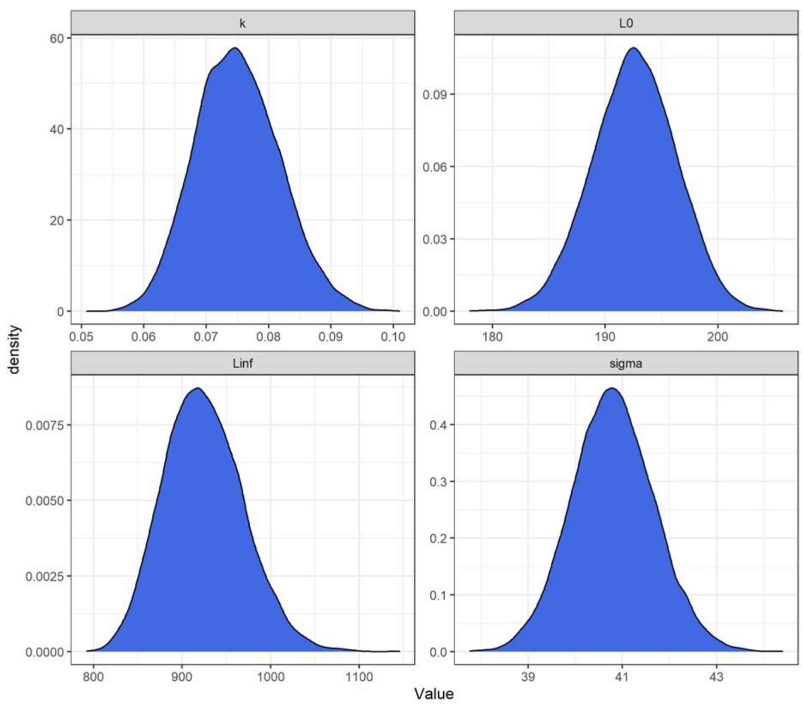

Figure 14: Posterior distributions of the VBGF parameter estimates. Priors were set at $L_\infty \sim N(1200, 50)$ and $L_0 \sim N(140, 10)$. A non-informative prior was used for σ (maximum value of $\sigma = 100$) and k (maximum value of $k = 0.3$ year^{-1}).

Table 11: Design aspects of the various TEDs used during 720 comparative trawls during the Northern Prawn Fishery tiger prawn season (August to November) of 2001. Grid type represents the 14 separate TEDs used by the 22 vessels participating in the sampling. The * represents TEDs that were broadly similar but had one or more design aspects that were changed during sampling. Grid angle is measured from the horizontal.

Grid type	1	2	3*	4*	5	6	7	8	9	10	11	12*	13	14	Total
Total no. of trawls	81	24	87	81	32	47	20	41	19	69	34	108	43	34	720
Size (cm)															
Height	120	135	118	110	130	110	120	120	115	110	119	96	120	120	
Width	180	160	148	140	150	120	125	115	130	140	174	102	150	145	
Grid orientation															
Bottom-shooter	–	24	40	–	–	47	–	41	19	69	5	108	43	34	430
Top-shooter	81	–	47	81	32	–	20	–	–	–	29	–	–	–	290
Grid shape															
Circular	–	–	–	–	–	–	20	–	–	–	–	–	–	–	20
Elliptical	–	–	87	81	–	47	–	–	19	–	–	–	–	–	234
Rectangular	81	–	–	–	32	–	–	41	–	–	34	–	–	–	188
Tombstone	–	24	–	–	–	–	–	–	–	69	–	108	43	34	278
Grid angle															
<=45	–	–	5	29	–	–	–	41	19	69	29	56	–	–	248
46 – 50	34	–	–	32	–	47	20	–	–	–	–	–	–	34	167
51 – 55	47	–	2	–	32	–	–	–	–	–	–	–	–	–	81
>=60	–	24	33	–	–	–	–	–	–	–	–	38	43	–	138
Not recorded	–	–	47	20	–	–	–	–	–	–	5	14	–	–	86
Bar spacing															
<=100	–	–	–	–	–	47	20	41	19	–	–	56	–	–	183
101 – 119	–	24	87	81	32	–	–	–	–	–	–	38	43	–	305
120	81	–	–	–	–	–	–	–	–	69	34	14	–	34	232
Guide panel present?															
No	–	–	69	–	–	–	–	–	–	–	–	–	43	–	112
Yes	34	–	–	61	32	47	–	41	19	69	–	94	–	34	431
Not recorded	47	24	18	20	–	–	20	–	–	–	34	14	–	–	177
BRD present?															
No	38	5	16	40	16	16	–	19	–	35	24	43	22	19	293
Yes	43	19	71	41	16	31	20	22	19	34	10	65	21	15	427
Bent deflector bars?															
No	81	–	31	49	32	47	20	41	19	69	34	108	43	34	608
Yes	–	9	56	–	–	–	–	–	–	–	–	–	–	–	65
Not recorded	–	15	–	32	–	–	–	–	–	–	–	–	–	–	47
Meshes past grid															
<10	–	24	2	32	–	–	20	41	–	–	–	14	–	–	133
10 – 20	81	–	33	29	–	47	–	–	–	69	–	–	43	34	336
>20	–	–	5	–	32	–	–	–	19	–	34	56	–	–	146
Not recorded	–	–	47	20	–	–	–	–	–	–	–	38	–	–	105

Table 12: Species name and common name of the 34 species of elasmobranch caught during 720 paired comparisons during the 2001 tiger prawn season (August–November) in the Northern Prawn Fishery, Australia. IUCN classifications are as follows: NE = not evaluated, LC = least concern, NT = near threatened, VU = vulnerable and EN = endangered. All IUCN classifications represent the global classification. Also shown are the number of each species or species group caught in total, numbers caught in control nets and treatment nets. Size (total length for all except disc width for Myliobatiformes, in cm) range and median size of animals caught in control nets only. $\hat{p}$ is the estimated proportion caught in the treatment nets and the *P*-value from the exact binomial tests undertaken with a null hypothesis of $p = 0.5$.

Species or species group	Common name	IUCN	Number caught			Size (cm)		$\hat{p}$ (95% CI)	*P* value
			Total	Control	Treatment	Range	Median		
Carcharhiniformes			**2508**	**1283**	**1225**	**6 – 302**	**76**	**0.488 (0.469 – 0.508)**	**0.255**
Carcharhinidae			2160	1086	1074	22 – 302	75	0.497 (0.476 – 0.519)	0.813
Carcharhinus coatesi	Whitecheek shark	NT	1218	590	628	22 – 90	73	0.516 (0.487 – 0.544)	0.289
Carcharhinus sorrah	Spottail shark	NT	25	13	12	41 – 118	88	0.480 (0.278 – 0.687)	1.000
Carcharhinus tilstoni	Australian blacktip shark	LC	634	339	295	37 – 159	76	0.465 (0.426 – 0.505)	0.088
Carcharhinus spp.	Requiem sharks unspecified		80	50	30	37 – 297	94	0.375 (0.269 – 0.490)	0.033
Galeocerdo cuvier	Tiger shark	NT	3	3	0	148 – 302	270		
Negaprion acutidens	Sharptooth lemon shark	VU	1	1	0	220	–		
Rhizoprionodon acutus	Milk shark	LC	198	89	109	32 – 92	81	0.551 (0.478 – 0.621)	0.177
Triaenodon obesus	Whitetip reef shark	NT	1	1	0	136	–		
Hemigaleidae									
Hemigaleus australiensis	Australian weasel shark	LC	267	140	127	6 – 96	69	0.476 (0.414 – 0.537)	0.463
Hemipristis elongata	Fossil shark	VU	6	3	3	82 – 154	136		
Scyliorhinidae									
Atelomycterus fasciatus	Banded catshark	LC	1	0	1	38	–		
Sphyrnidae			39	28	11	48 – 302	115	0.282 (0.150 – 0.449)	0.009
Eusphyra blochii	Winghead shark	EN	3	3	0	110 – 140	137	0.000 (0.000 – 0.708)	0.250
Sphyrna lewini	Scalloped hammerhead	EN	32	22	10	48 – 302	100	0.312 (0.161 – 0.500)	0.050
Sphyrna mokarran	Great hammerhead	EN	2	1	1	110 – 215	162		
Sphyrna spp.	Hammerheads unspecified		3	2	1	61 – 200	131		
Orectolobiformes			**489**	**306**	**183**	**21 – 290**	**74**	**0.374 (0.331 – 0.419)**	**< 0.001**
Ginglymostomatidae									
Nebrius ferrugineus	Tawny nurse shark	VU	3	3	0	215 – 290	236		
Hemiscylliidae									
Chiloscyllium punctatum	Grey carpetshark	NT	395	221	174	21 – 150	70	0.441 (0.391 – 0.491)	0.021
Orectolobidae									
Eucrossorhinus dasypogon	Tasselled wobbegong	LC	1	1	0	38	–		
Orectolobidae	Wobbegongs unspecified		1	0	1	63	–		
Parascyllidae									
Parascyllidae	Collared carpetsharks unspecified	–	21	14	7	32 – 56	43	0.333 (0.146 – 0.570)	0.189
Stegostomidae									
Stegostoma fasciatum	Zebra shark	EN	68	67	1	63 – 192	170	0.015 (0.000 – 0.079)	< 0.001
Myliobatiformes			**2742**	**1563**	**1179**	**4 – 230**	**30**	**0.430 (0.413 – 0.450)**	**< 0.001**
Dasyatidae			2030	1190	840	4 – 230	24	0.414 (0.392 – 0.436)	< 0.001
Himantura uarnak	Coach whipray	VU	13	13	0	28 – 210	95	–	–
Himantura leoparda	Leopard whipray	VU	35	33	2	33 – 153	126	0.057 (0.007 – 0.192)	< 0.001
Maculabatis toshi	Brown whipray	LC	662	383	279	11 – 122	40	0.421 (0.384 – 0.460)	< 0.001
Neotrygon annotata	Plain maskray	NT	461	270	191	4 – 77	20	0.414 (0.369 – 0.461)	< 0.001
Neotrygon australiae	Australian bluespotted maskray	NE	67	28	39	11 – 38	29	0.582 (0.455 – 0.702)	0.222
Neotrygon leylandi	Painted maskray	LC	627	313	314	7 – 38	17	0.501 (0.461 – 0.541)	1.000
Pastinachus ater	Broad cowtail ray	LC	64	62	2	24 – 153	127	0.031 (0.004 – 0.108)	< 0.001
Taeniura meyeni	Blotched stingray	VU	1	1	0	138	–		
Urogymnus asperrimus	Porcupine ray	VU	1	1	0	109	–		
Urogymnus granulatus	Mangrove whipray	VU	1	1	0	111	–		
Dasyatidae	Whipray unspecified		98	85	13	9 – 230	121	0.137 (0.075 – 0.223)	< 0.001
Gymnuridae									
Gymnura australis	Australian butterfly ray	LC	641	336	305	17 – 81	44	0.476 (0.437 – 0.515)	0.236
Myliobatidae			71	37	34	21 – 159	69	0.479 (0.359 – 0.601)	0.813
Aetobatus ocellatus	Spotted eagle ray	VU	5	5	0	124 – 149	130		
Aetomylaeus caeruleofasciatus	Bluebanded eagle ray	LC	61	28	33	21 – 151	48	0.541 (0.408 – 0.669)	0.609
Myliobatididae	Eagle rays unspecified		5	4	1	22 – 159	57		
Rhinopristiformes			**607**	**361**	**246**	**27 – 320**	**66**	**0.405 (0.366 – 0.446)**	**< 0.001**
Glaucostegidae									
Glaucostegus typus	Giant guitarfish	VU	4	4	0	72 – 221	208		
Pristidae			23	14	9	126 – 300	211	0.391 (0.197 – 0.615)	0.405
Anoxypristis cuspidata	Narrow sawfish	EN	16	12	4	126 – 300	211	0.250 (0.073 – 0.524)	0.077
Pristidae	Sawfish unspecified		7	2	5	172 – 240	210		
Rhinidae									
Rhina ancylostoma	Shark ray	VU	9	9	0	137 – 213	190		
Rhynchobatus australiae	Bottlenose wedgefish	VU	571	334	237	27 – 320	65	0.415 (0.374 – 0.457)	< 0.001

Table 13: The size (total length for all except disc width for Myliobatiformes, in cm) at which escape and retention were equal (S_0), the percentage of animals in control nets at sizes greater than S_0 (i.e., $S > S_0$), the size at 50% escape (S_{50}) and the percentage of animals in control nets at sizes greater than S_{50} (i.e., $S > S_{50}$). Values in parentheses are ranges between upper and lower 95% confidence intervals from Figure 4 and Figure 5.

Order	Orientation	S_0	$S > S_0$ (%)	S_{50}	$S > S_{50}$ (%)
Carcharhiniformes	Top:	61 (48–72)	88	113 (99–133)	5
	Bottom:	76 (67–84)	42	127 (111–153)	3
Orectolobiformes	–	53 (43–63)	84	81 (73–90)	32
Myliobatiformes	Top:	31 (22–40)	64	69 (60–78)	22
	Bottom:	17 (10–23)	82	54 (47–62)	18
Rhinopristiformes	–	48 (24–64)	92	109 (92–135)	24

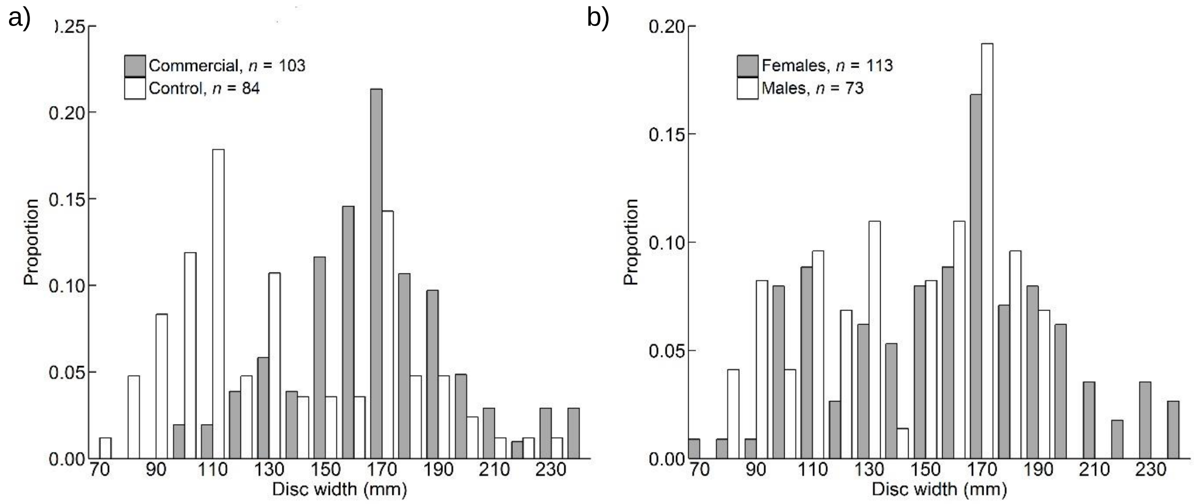

Figure 15: Length–frequency distributions for 187 common stingarees (*Trygonoptera testacea*) caught during three post-release survival experiments as a function of (a) sampling type (control v. commercially trawled) and (b) sex. Note, sex was not recorded for five individuals.

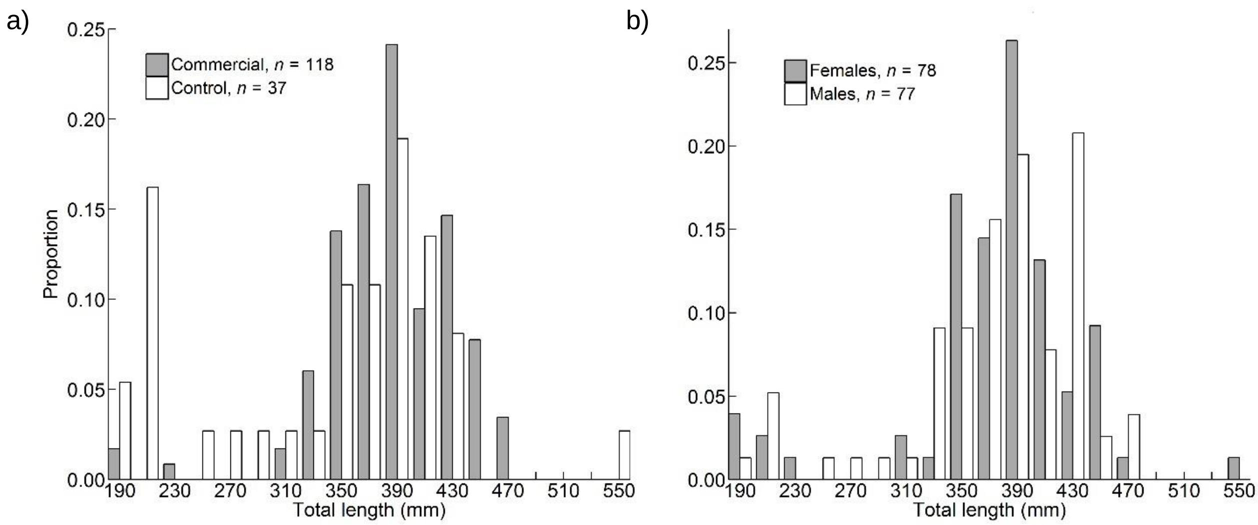

Figure 16: Length–frequency distributions for 155 eastern shovelnose rays (*Aptychotrema rostrata*), caught during three post-release survival experiments, as a function of (a) sampling type (control v. commercially trawled) and (b) sex.

Table 14: Ecological risk categories and precautionary risk categories for 48 species assessed in the study area. F_{2019} (year^{-1}) is the level of fishing mortality applied by the trawl fishery in the 2019 fishing year, F_{2019}+90%CI is the upper 90% confidence interval of F_{2019} and F_{msm} is the maximum sustainable fishing mortality.

Species	F_{2019}	F_{2019}+90%CI	F_{msm}	Risk category	Precautionary risk category
Aetomylaeus nichofii	0.006	0.007	0.183	Low	Low
Aptychotrema rostrata	0.002	0.005	0.083	Low	Low
Asymbolus analis	0.016	0.063	0.165	Low	Low
Asymbolus rubiginosus	0.076	0.170	0.166	Low	Precautionary medium risk
Bathytoshia brevicaudata	0.140	0.169	0.176	Low	Low
Bathytoshia lata	0.088	0.107	0.144	Low	Low
Brachaelurus colcloughi	0.034	0.071	0.108	Low	Low
Carcharhinus amboinensis	0.035	0.072	0.099	Low	Low
Carcharhinus brevipinna	0.029	0.060	0.136	Low	Low
Carcharhinus coatesi	0.025	0.049	0.280	Low	Low
Chiloscyllium punctatum	0.042	0.088	0.187	Low	Low
Dentiraja australis	0.152	0.170	0.152	Low	Precautionary high risk
Dentiraja endeavouri	0.143	0.162	0.169	Low	Precautionary medium risk
Dipturus melanospilus	0.000	0.000	0.116	Low	Low
Figaro boardmani	0.011	0.037	0.103	Low	Low
Glaucostegus typus	0.006	0.007	0.137	Low	Low
Gymnura australis	0.017	0.023	0.160	Low	Low
Hemigaleus australiensis	0.055	0.119	0.180	Low	Low
Hemitrygon fluviorum	0.013	0.015	0.078	Low	Low
Heterodontus galeatus	0.080	0.173	0.187	Low	Low
Hydrolagus lemures	0.027	0.030	0.097	Low	Low
Hypnos monopterygius	0.086	0.096	0.166	Low	Low
Loxodon macrorhinus	0.034	0.070	0.231	Low	Low
Maculabatis astra	0.057	0.069	0.086	Low	Low
Maculabatis toshi	0.009	0.011	0.197	Low	Low
Mustelus antarcticus	0.064	0.142	0.074	Low	Precautionary extreme high risk
Neotrygon picta	0.006	0.007	0.122	Low	Low
Neotrygon trigonoides	0.035	0.042	0.108	Low	Low
Orectolobus maculatus	0.057	0.117	0.088	Low	Precautionary medium risk
Parascyllium collare	0.127	0.272	0.170	Low	Precautionary high risk
Pastinachus ater	0.007	0.008	0.112	Low	Low
Rhizoprionodon acutus	0.020	0.040	0.270	Low	Low
Rhizoprionodon taylori	0.024	0.049	0.378	Low	Low
Rhynchobatus australiae	0.003	0.005	0.137	Low	Low
Sphyrna lewini	0.029	0.062	0.088	Low	Low
Squalus grahami	0.000	0.000	0.110	Low	Low
Squalus megalops	0.065	0.143	0.058	Medium	Precautionary extreme high risk
Squalus montalbani	0.000	0.000	0.044	Low	Low
Squatina albipunctata	0.068	0.146	0.132	Low	Precautionary medium risk
Stegostoma tigrinum	0.003	0.005	0.080	Low	Low
Tetronarce nobiliana	0.082	0.092	0.093	Low	Low
Trygonoptera testacea	0.020	0.022	0.189	Low	Low
Trygonorrhina fasciata	0.031	0.038	0.185	Low	Low
Urolophus bucculentus	0.085	0.103	0.141	Low	Low
Urolophus flavomosaicus	0.086	0.105	0.143	Low	Low
Urolophus kapalensis	0.169	0.192	0.180	Low	Precautionary medium risk
Urolophus sufflavus	0.099	0.119	0.160	Low	Low
Urolophus viridis	0.082	0.100	0.146	Low	Low

Table 15: Relevant life history information and resultant estimates of natural mortality (M) used to calculate F_{msm}. Depth, in metres, is the midpoint of the published depth range and °C is the mean annual temperature at this depth, L_{max} is the maximum length (mm) from the literature, L_{∞} (mm), W_{∞} (g) and k (year^{-1}) are the von Bertalanffy growth parameters, t_{max} is the maximum age (years), and t_{mat} is the age-at-maturity (years). Values in red are those that were calculated from the equations cited in section 5.2.2.

Species	L_{max}	L_{∞}/W_{∞}	k	t_{max}	t_{mat}	Reference	Depth	°C	Natural mortality (M)							
									Hoenig (1983)	Then et al. (2015)	Then et al. (2015)	Fishbase	Jensen (1996)	Pauly (1980)	Frisk et al. (2001)	Frisk et al. (2001)
Aetomylaeus nichofii	72	74.44					57	22.2				0.45				
Aptychotrema rostrata		92.3	0.08	22.5	10.50	Campbell et al. (2021)	50	22.5	0.20	0.28	0.15	0.45	0.15	0.23	0.15	0.15
Asymbolus analis	60	62.21					112	20.0				0.40				
Asymbolus rubiginosus	55	57.11					112	20.0				0.40				
Bathytoshia brevicaudata	210	213.45					75	21.5				0.43				
Bathytoshia lata	260	263.38					180	17.5				0.35				
Brachaelurus colcloughi	85	87.60	0.13	19	8.52	Kyne et al. (2015)	50	22.5	0.23	0.33	0.22	0.45	0.19	0.32	0.19	0.18
Carcharhinus amboinensis		267.20	0.15	18.4	8.28	Tillett et al. (2011)	50	22.5	0.24	0.34	0.16	0.45	0.20	0.25	0.19	0.18
Carcharhinus brevipinna		200.00	0.14	11	4.58	Carlson and Baremore (2005)	37	23.1	0.40	0.54	0.17	0.46	0.36	0.27	0.19	0.26
Carcharhinus coatesi		80.00	0.83	7	2.00	Baje et al. (2019)	85	21.1	0.67	0.88	0.85	0.42	0.83	1.05	0.40	0.36
Chiloscyllium punctatum	132	135.20					45	22.8				0.46				
Dentiraja australis	50	51.99	0.30	13.2	5.88		175	17.7	0.34	0.46	0.47	0.36	0.28	0.57	0.26	0.22
Dentiraja endeavouri	37	38.66	0.36	12.1	5.26		205	16.7	0.37	0.50	0.58	0.34	0.31	0.67	0.28	0.24
Dipturus melanospilus	78	80.54	0.23	15.2	6.89		470	11.2	0.29	0.41	0.33	0.23	0.24	0.34	0.23	0.20
Figaro boardmani	61	63.23					385	12.4				0.25				
Glaucostegus typus		277.00	0.30	13.3	5.94	White et al. (2014)	50	22.5	0.33	0.46	0.27	0.45	0.28	0.40	0.26	0.22
Gymnura australis	94	96.77					125	19.5				0.39				
Hemigaleus australiensis	110	113.00					65	21.9				0.44				
Hemitrygon fluvorium		147.50	0.03	21	13.40	Pierce and Bennett (2011)	15	24.1	0.21	0.30	0.06	0.48	0.12	0.11	0.10	0.13
Heterodontus galeatus	150	153.30					45	22.8				0.46				
Hydrolagus lemures	82	84.60					430	11.7				0.24				
Hypnos monopterygius	62.5	64.76					110	20.1				0.40				
Loxodon macrorhinus		84.20	0.41	7.6	1.90	Gutteridge et al. (2013)	50	22.5	0.58	0.77	0.50	0.45	0.87	0.68	0.30	0.37
Maculabatis astra		82.20	0.07	29	8.67	Jacobsen and Bennett (2011)	70	21.7	0.15	0.22	0.14	0.44	0.19	0.22	0.15	0.18
Maculabatis toshi	74	76.47					20	23.9				0.48				
Mustelus antarcticus		186.1	0.04	18	14	M. Campbell (unpublished data)	225	16.1	0.25	0.35	0.06	0.32	0.12	0.08	0.10	0.12
Neotrygon picta		36.10	0.08	18	4	Jacobsen and Bennett (2010)	50	22.5	0.25	0.35	0.20	0.46	0.41	0.29	0.15	0.28
Neotrygon trigonoides		44.10	0.08	13.4	6	Jacobsen and Bennett (2010)	45	22.8	0.33	0.45	0.19	0.46	0.28	0.28	0.15	0.22
Orectolobus maculatus		163.00	0.09	22	9.58	Huveneers et al. (2013)	110	20.1	0.20	0.29	0.13	0.40	0.17	0.20	0.16	0.16
Parascyllium collare	87	89.68					97	20.6				0.41				
Pastinachus ater		156	0.16	17.68	8.00	O'Shea et al. (2013)	30	23.4	0.25	0.20	0.47	0.21	0.31	0.35	0.20	0.19
Rhizoprionodon acutus		86.10	0.63	8	1.80	Harry et al. (2010)	100	20.4	0.55	0.73	0.68	0.41	0.92	0.85	0.36	0.38
Rhizoprionodon taylori		65.20	1.01	7	1.33	Simpfendorfer (1993)	50	22.5	0.62	0.82	1.05	0.45	1.24	1.31	0.44	0.41
Rhynchobatus australiae		204.50	0.41	19	7	White et al. (2014)	30	23.4	0.23	0.33	0.37	0.47	0.24	0.54	0.30	0.20
Sphyrna lewini		319.70	0.25	35	13	Chen et al. (1990)	142	18.8	0.13	0.19	0.22	0.38	0.13	0.31	0.24	0.13
Squalus grahami	73	75.46					335	13.3				0.27				
Squalus megalops		82.90	0.03	28	22	Braccini et al. (2007)	305	14.0	0.16	0.23	0.08	0.28	0.08	0.11	0.11	0.09
Squalus montalbani		362.40	0.01	30	11.81	Rigby et al. (2016a)	480	11.1	0.15	0.22	0.02	0.22	0.14	0.02	0.05	0.14
Squatina albipunctata	130	133.10					225	16.1				0.32				
Stegosoma fasciata	246	249.00	0.04	28	6	Dudgeon et al. (2019)	30	23.4	0.16	0.23	0.06	0.47	0.28	0.10	0.11	0.22
Tetronacre nobiliana	180	183.40					463	11.3				0.23				
Trygonoptera testacea	52	54.04					45	22.8				0.46				
Trygonorrhina fasciata	120	123.03					50	22.5				0.45				
Urolophus bucculentus	89	91.70	0.27	14	5	Kyne et al. (2019)	165	18.0	0.32	0.44	0.36	0.36	0.33	0.45	0.25	0.25
Urolophus flavomosaicus	59	61.19					190	17.2				0.35				
Urolophus kapalensis	52	54.04					70	21.7				0.44				
Urolophus sufflavus	42	43.80					132	19.2				0.39				
Urolophus viridis	51	53.02					180	17.5				0.36				

Table 16: The total area trawled (in km^2) within each species' distribution assessed as a function of gear type (prawn trawl, scallop trawl, Danish seine and fish trawl) and escape-hole orientation (bottom- or top -shooter) of the turtle excluder devices used in prawn and scallop trawl nets. Also shown is the area of each species' distribution in the area south of the Great Barrier Reef Marine Park. *E* is the escape from bottom- and top-shooter turtle excluder devices given the median size of these species (disc width for Myliobatiformes and total length for all other species, in cm), *S* is the estimate of post-trawl survival and n is the sample size used to derive estimates of *Q*. The IUCN Redlist categories are correct as of 5 November 2021.

Species	Family	Order	IUCN Redlist Status	Prawn		Scallop		Danish seine	Fish Trawl	Distribution Area	*E*		Median size	*S*	*n*
				Bottom-shooter	Top-shooter	Bottom-shooter	Top-shooter				Bottom-shooter	Top-shooter			
Aetomylaeus nichofii	Myliobatidae	Myliobatiformes	Vulnerable	163	100	496	55	0	0	6,884	0	0	-	0	18
Aptychotrema rostrata	Trygonorrhinidae	Rhinopristiformes	Least Concern	5,568	1,751	533	58	329	141	23,031	-0.029	-0.029	44	0.87[1]	3,933
Asymbolus analis	Scyliorhinidae	Carcharhiniformes	Least Concern	3,092	1,328	0	0	304	0	3,822	-0.492	-0.205	47	0.88[2]	31
Asymbolus rubiginosus	Scyliorhinidae	Carcharhiniformes	Least Concern	3,057	1,319	0	0	304	0	3,842	-0.545	-0.248	44	0.54[2]	21
Bathytoshia brevicaudata	Dasyatidae	Myliobatiformes	Least Concern	4,597	1,735	0	0	307	7	5,622	0	0	-	0	46
Bathytoshia lata	Dasyatidae	Myliobatiformes	Vulnerable	7,657	2,674	1	0	307	20	9,749	0.370	0.218	43	0	16
Brachaelurus colcloughi	Brachaeluridae	Orectolobiformes	Vulnerable	5,568	1,751	533	58	329	141	23,031	0.029	0.029	54	0	8
Carcharhinus amboinensis	Carcharhinidae	Carcharhiniformes	Vulnerable	5,336	1,712	533	58	328	141	22,332	0	0	-	0	1
Carcharhinus brevipinna	Carcharhinidae	Carcharhiniformes	Vulnerable	4,261	1,315	526	58	328	141	21,517	0	0	-	0	7
Carcharhinus coatesi	Carcharhinidae	Carcharhiniformes	Least Concern	2,270	511	517	58	19	107	13,944	-0.053	0.150	73	0	0
Chiloscyllium punctatum	Hemiscylliidae	Orectolobiformes	Near Threatened	5,326	1,713	532	58	329	141	22,560	-0.218	-0.218	44	0	46
Dentiraja australis	Rajidae	Rajiformes	Near Threatened	3,057	1,319	0	0	304	0	4,709	0	0	-	0	6
Dentiraja endeavouri	Rajidae	Rajiformes	Near Threatened	3,858	1,152	0	0	0	0	4,221	0	0	-	0.24[2]	128
Dipturus melanospilus	Rajidae	Rajiformes	Data deficient	1	0	0	0	0	0	3,019	0	0	-	0	6
Figaro boardmani	Scyliorhinidae	Carcharhiniformes	Least Concern	1,202	385	0	0	0	0	3,050	-0.558	-0.259	43	0.82[2]	30
Glaucostegus typus	Glaucostegidae	Rhinopristiformes	Critically Endangered	5,326	1,713	532	58	329	141	22,560	0.843	0.843	208	0	16
Gymnura australis	Gymnuridae	Myliobatiformes	Least Concern	10,742	3,381	533	58	330	141	27,915	0.376	0.225	43	0.59[3]	17
Hemigaleus australiensis	Hemigaleidae	Carcharhiniformes	Least Concern	10,690	3,358	533	58	330	141	25,836	-0.628	-0.315	41	0.38[3]	114
Hemitrygon fluviorum	Dasyatidae	Myliobatiformes	Near Threatened	583	154	420	31	118	9	8,857	0	0	-	0	46
Heterodontus galeatus	Heterodontidae	Heterodontiformes	Least Concern	2,431	1,078	0	0	307	6	4,355	0	0	-	0	13
Hydrolagus lemures	Chimaeridae	Chimaeriformes	Near Threatened	671	241	0	0	0	0	5,397	0	0	-	0	1
Hypnos monopterygius	Hypnidae	Torpediniformes	Least Concern	10,742	3,381	533	58	330	141	27,915	0	0	-	0	62
Loxodon macrorhinus	Carcharhinidae	Carcharhiniformes	Near Threatened	5,336	1,712	533	58	329	141	23,031	0	0	-	0	37
Maculabatis astra	Dasyatidae	Myliobatiformes	Least Concern	8,653	2,680	533	58	329	141	24,511	0	0	-	0	2
Maculabatis toshi	Dasyatidae	Myliobatiformes	Least Concern	1,336	308	502	58	271	47	10,570	0.341	0.182	40	0.47[3]	2
Mustelus antarcticus	Triakidae	Carcharhiniformes	Least Concern	6,530	2,053	20	0	2.2	4	13,279	0	0	-	0	25
Neotrygon picta	Dasyatidae	Myliobatiformes	Least Concern	163	99	496	55	0	0	6,874	0	0	-	0.41[3]	4
Neotrygon trigonoides	Dasyatidae	Myliobatiformes	Least Concern	5,326	1,713	532	58	329	141	22,560	0.169	-0.032	27	0	385
Orectolobus maculatus	Orectolobidae	Orectolobiformes	Least Concern	10,739	3,381	533	58	330	141	26,577	0	0	-	0	7
Parascyllium collare	Parascylliidae	Orectolobiformes	Least Concern	5,385	1,965	0	0	304	6	5,804	0	0	-	0	28
Pastinachus ater	Dasyatidae	Myliobatiformes	Vulnerable	3,403	951	524	58	326	141	17,551	0.874	0.842	127	0	0
Rhizoprionodon acutus	Carcharhinidae	Carcharhiniformes	Vulnerable	2,379	551	521	58	18	108	14,062	0.041	0.226	81	0.18[3]	12
Rhizoprionodon taylori	Carcharhinidae	Carcharhiniformes	Least Concern	2,766	614	532	58	22	121	16,978	0	0	-	0	12
Rhynchobatus australiae	Rhinidae	Rhinopristiformes	Critically Endangered	3,447	962	524	58	327	141	17,974	0.117	0.117	65	0.90[3]	16
Sphyrna lewini	Sphyrnidae	Carcharhiniformes	Critically Endangered	10,742	3,381	533	58	330	141	27,915	0.450	0.556	122	0	53
Squalus grahami	Squalidae	Squaliformes	Near Threatened	3	0	0	0	0	0	2,504	0	0	-	0	0
Squalus megalops	Squalidae	Squaliformes	Least Concern	10,159	3,228	113	27	212	132	20,791	0	0	-	0	0
Squalus montalbani	Squalidae	Squaliformes	Vulnerable	0	0	0	0	0	0	1,934	0	0	-	0	7
Squatina albipunctata	Squatinidae	Squatiniformes	Vulnerable	10,159	3,228	113	27	213	132	20,085	0	0	-	0	5
Stegostoma tigrinum	Stegostomatidae	Orectolobiformes	Endangered	3,447	962	524	58	327	141	17,974	0.935	0.935	170	0	5
Tetronarce nobiliana	Torpedinidae	Torpediniformes	Data Deficient	10,742	3,381	533	58	330	141	27,915	0	0	-	0	0
Trygonoptera testacea	Urolophidae	Myliobatiformes	Near Threatened	5,163	1,613	36	3	329	141	16,151	-0.028	-0.275	16	0.17[1]	1,270
Trygonorrhina fasciata	Trygonorrhinidae	Rhinopristiformes	Least Concern	3,368	1,263	12	0	313	33	9,572	0	0	-	0	3
Urolophus bucculentus	Urolophidae	Myliobatiformes	Vulnerable	1,055	437	0	0	2.2	0	2,067	0	0	-	0	3
Urolophus flavomosaicus	Urolophidae	Myliobatiformes	Least Concern	3,476	1,003	9	0	1.1	0	6,174	0	0	-	0	1
Urolophus kapalensis	Urolophidae	Myliobatiformes	Near Threatened	2,071	994	0	0	306	0	3,230	-0.002	-0.243	17	0	121
Urolophus sufflavus	Urolophidae	Myliobatiformes	Vulnerable	1,542	750	0	0	54	0	2,806	0	0	-	0	1
Urolophus viridis	Urolophidae	Myliobatiformes	Vulnerable	1,054	437	0	0	2.2	0	2,145	0	0	-	0	1

[1]Campbell et al. (2018); [2]Peter Kyne unpublished data; [3]Stobutzki et al. (2002)