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Characteristics of east Australian demersal trawl elasmobranch bycatch as revealed by short-term latitudinal monitoring

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ABSTRACT: Elasmobranchs are being depleted on a global scale, caused mainly by fisheries. Demersal trawling is a component of mortality but is often not assessed. This could pose risk to benthic/demersal elasmobranchs which are often endemic and therefore vulnerable to fisheries when species ranges are within (or mainly within) trawl footprints. Northern New South Wales (NSW) is an area with endemism but also an area with fisheries such as the ocean prawn trawl (OPT) (penaeid sector). The OPT may interact with elasmobranchs, but this has never been comprehensively studied. To identify high assessment-priority species, determine spatiotemporal stratification for designing future monitoring, and to report catch rates of individuals caught during a trip (i.e. form baseline), we implemented an observer programme (2017 to 2019). To test for stratification of assemblages, we used model-based multivariate analysis. On 435 trawl trips, observers identified elasmobranchs from ~54 species, 13 orders and 34 families from variable catches. Only 2 elasmobranchs were protected in NSW, ~7% qualified for conservation listing, and ~33 and ~17% were endemic and lifeboat (listed elsewhere) species, respectively. Models suggested common elasmobranch assemblages were significantly affected by all strata (geographic zone, season and depth). Elasmobranch catch rates were low compared to other taxonomic groups (e.g. teleost fish), with 2 species captured at >10, 5 species at >2, and the remaining species <2 individuals per trip. The occurrence of endemism and spatiotemporal assemblage variation was explained by mesoscale climate transitions and oceanography. This study forms a timely baseline which can be used to assess the impact of the OPT on elasmobranchs in the future.

KEY WORDS: Commercial fisheries monitoring · Conservation · Penaeid trawling · Benthic · Multivariate modelling · Endemic · Biodiversity · CPUE · Stratification

1. INTRODUCTION

Fishing-based decline of elasmobranchs is occurring globally (e.g. oceanic shark and ray abundance has declined by 71% since 1970) (Pacoureau et al. 2021), and consequently, there has been recent media and research attention (e.g. AMCS 2021, Dulvy et al. 2021). Markets for elasmobranch products have increased in the last 2 decades, increasing pressure on this species group (Dent & Clarke 2015). Also, finning of elasmobranchs (particularly sharks) may have amplified the fishing impact (Dulvy et al. 2014), as this has not only created a luxury market, but also illegal activity due to the high monetary value. Moreover,

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many regulated fisheries, including pelagic longline, and demersal longline and trawl, are also not consistently assessed for contribution to mortality (Daley & Gray 2021). Globally, it is estimated that ~50% of the total catch of elasmobranchs is taken as bycatch which is mainly unreported and unmanaged (Stevens et al. 2000). Therefore, there is fishing mortality that is unquantified (Last & Stevens 2009, Carr et al. 2013). Elasmobranch life history traits mean they are not very productive (Dulvy et al. 2008, Last & Stevens 2009, James et al. 2016) and this makes them very susceptible to fishing-based depletion (i.e. compared to some teleost fish) (Ellis et al. 2008, Campbell et al. 2020). Further, benthic and demersal (herein demersal) endemic elasmobranchs may be at risk of extinction if fishing occurs over a large proportion of their range (Zhou & Griffiths 2008, Daley & Gray 2021, Kyne et al. 2021) and have been shown to be depleted quickly (<2 decades via demersal trawling in south-eastern Australia) (Graham et al. 2001). The effect of unmanaged fishing on demersal and other elasmobranchs also has potential to adversely affect the wider marine ecosystem (Stevens et al. 2000).

Demersal trawling has the capacity to interact with a large volume of non-target species termed bycatch (Ye et al. 2000, Kennelly 2020). Demersal penaeid trawls often have large bycatches due to unselective gear that is often deployed in sub-tropical to tropical climatic zones and near diverse habitats (Watson & Goeden 1989, Broadhurst 2000, Ye et al. 2000). Such trawling interacts with and may be very impactful on demersal elasmobranch species (Macbeth et al. 2008), whereas faster-swimming pelagic elasmobranchs are more likely to be occasional captures (Campbell et al. 2020, Barnes et al. 2022b). Therefore, it is particularly important to assess the risk demersal trawling poses to elasmobranchs, but this is rarely done (Daley & Gray 2021). Trawling may cause more problematic depletion in areas of high endemism which is mainly associated with demersal species (Manes et al. 2021). For example, recolonisation to aid the rebuilding of depleted populations via migration may be limited for endemic species (Stevens et al. 2000, Kyne et al. 2021).

An area of high marine species richness and endemism is the New South Wales (NSW) north coast (Fig. 1, Nelson 1978, Harrison & Noss 2017) due to particular climatic and oceanographic conditions (Zann 2000, Malcolm et al. 2011, Harrison & Noss 2017). In this area, the NSW inshore and offshore prawn sector of the 'ocean trawl fishery' operates (herein termed ocean prawn trawl [OPT]), targeting mainly eastern king prawn *Melicertus plebejus*. Typical of penaeid trawl fisheries operating in warm temperate to sub-tropical environments, bycatch of the OPT exceeds retained catch and includes a diverse range of species (Kennelly et al. 1998, Barnes et al. 2022b). However, fishery-level OPT bycatch has not been quantified for ~30 yr (Kennelly et al. 1998). Further, Kennelly et al. (1998) only described commer-

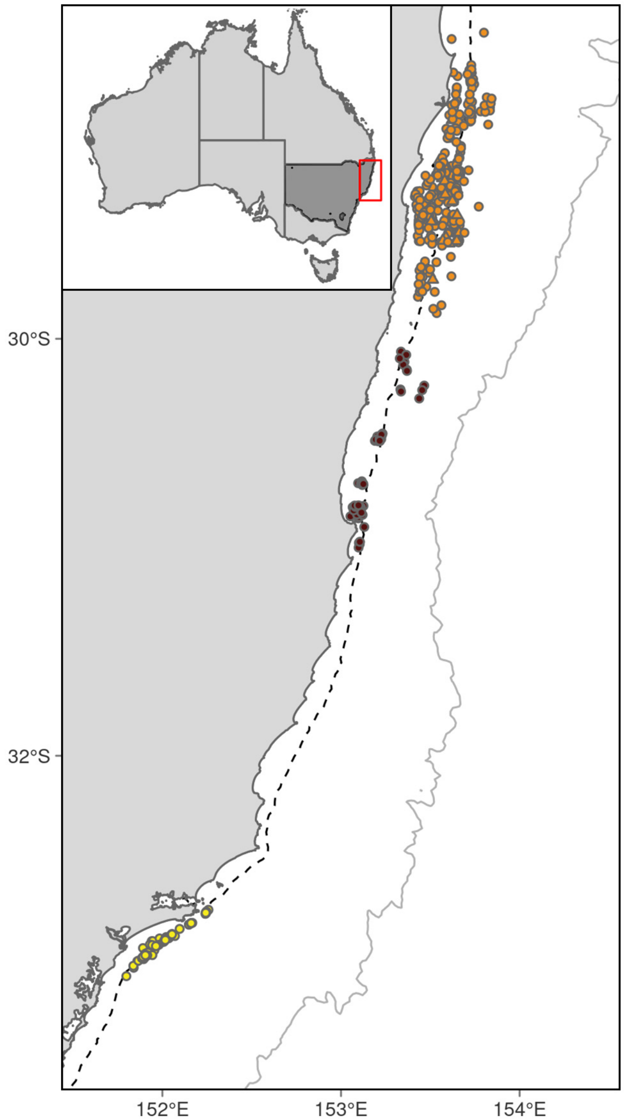

Fig. 1. Study location with the inset showing the state of New South Wales (dark gray shade) in relation to Australia and the extent of the ocean prawn trawl fishery (red box), enlarged in the main map. The 435 observed ocean prawn trawl trips (all trips i.e. prior to any subsetting) from 2017 to 2019 are indicated. Circles: trips recording elasmobranchs; triangles: no records (see zoomed map Fig. S1 in the Supplement). Zones: (orange) north; (dark brown) central; (yellow) south. Dashed line: the 64 m contour differentiating the shallow (<64 m) and deep (≥64 m) depths; solid line: approximate location of the eastern fishery boundary (4000 m isobath)

cially important elasmobranchs, meaning that catches of bycatch species are poorly understood. This is despite the OPT having a large spatiotemporal footprint (i.e. 5 degree latitudinal gradient [Fig. 1] and all seasons) and being in an elasmobranch endemism hotspot (Heupel 2018, Stein et al. 2018), and therefore posing a potential risk to demersal elasmobranchs (NSW MEMA 2016, Daley & Gray 2021). For a detailed description of the OPT, see Barnes et al. (2022b). Briefly, the fishery is divided into 2 spatial management zones (north and south of 31° S) and 6 latitudinal spatial zones for catch reporting. There are multiple marine protected areas that restrict trawling and there are also several spatiotemporal managed areas permanently and temporarily closing trawling (Taylor et al. 2020, 2021). The OPT operates from the natural shoreline offshore to the 4000 m depth isobath (Fig. 1) north of 33.5° S. From 31° to 33.5° S, the area of operation of the OPT overlaps with the permitted fishing area of the northern fish trawl (NFT) sector of the OTF. *M. plebejus* from the same biological stock are targeted by fishers in the adjacent Queensland East Coast Otter Trawl Fishery (ECOTF) managed by the state of Queensland (QLD). Triple rigged trawl gear is the main gear in NSW (Macbeth et al. 2008) and fishing trips occur at night and typically involve three 3 h net tows (or trawl shots, herein trawls). Only a small number of elasmobranch species are sometimes retained for sale by the OPT.

For the OPT (but also many fisheries), there is a need to collect basic fisheries information on elasmobranch bycatch (Kennelly 1995, 2020, Daley & Gray 2021). Such information includes the identification of interacting species (Daley & Gray 2021) and quantification of spatiotemporal variation in both assemblages and catch rates (Kennelly et al. 1998). This can provide information about the risk to species from trawling (e.g. Threatened, Endangered and Protected species [TEPS]) and any strata (e.g. spatiotemporal) requirements for future monitoring, and provide a baseline to monitor changes in relative abundance. Currently, observer surveys provide the best means to acquire this information (Kennelly 2020). Observer-based monitoring is expensive but can be streamlined by considering stratification requirements (spatiotemporal) and optimal sampling fractions (Borges et al. 2004, Johnson & Barnes 2023). Given the global plight of elasmobranchs and the unknown impact of the OPT, there is a need for assessment and acquisition of knowledge facilitated via more regular monitoring. Thus, any requirements for intervention and mitigation can be assessed based on rigorous scientific information (Daley & Gray 2021).

Given the above and international, Australian and NSW-based legislation or recommendations on elasmobranch harvest and bycatch sustainability (Shark Advisory Group & Lack 2004, Fischer et al. 2012, NSW MEMA 2016), we sought to characterise the elasmobranch bycatch community from the OPT which, despite the significant impact this fishery potentially poses to elasmobranchs, has not been done comprehensively before. This was done using the data from an observer survey via 3 specific objectives. First, identify species that form the elasmobranch assemblage and assess if any species are at urgent risk from the OPT via classification of extinction risk and endemism. Second, assess spatiotemporal variability in the elasmobranch bycatch assemblage to inform future monitoring. Third, estimate catch rates to form a proxy for baseline and temporal comparisons of relative abundance. As this work is an early step in developing an understanding of the risk the OPT poses to elasmobranchs, we discuss the results in terms of the logical next step based on this preliminary but comprehensive data analysis.

2. MATERIALS AND METHODS

2.1. Data collected

Observers (n = 10) attended trawling operations over 2 yr (2017 to 2019), incorporating all 4 austral seasons. The latitudinal extent of the OPT was divided into 3 geographic zones for analytical purposes, comprising fishery reporting zones 1–2, 3 and 4, herein, north, central and south, respectively (www.dpi.nsw.gov.au/fishing/commercial/catch-effort, Fig. 1), The number of trips sampled was proportional to the typical commercial effort (sampling fractions) within the geographic zones (herein zones) and therefore sampling had a north to south negative gradient to reflect the spatial differences in fishery effort (Barnes et al. 2022b). All organisms in catches were identified to species whenever possible and enumerated; a small number of catches were not enumerated due to those being very large. In the rare case of a very large catch from a given trawl, observers randomly subsampled 30 individuals from each species and measured their mass. Species level total numbers were then extrapolated based on the subsample weight and the total weight of the catch from the large catch trawl (Courtney et al. 2014, Barnes et al. 2022b). Observers also recorded the mean trawl depth (m), trawl start and end geographic position (decimal degrees, used to calculate the linear trawl distance in km) and trawl start and end time.

2.2. Data analysis

2.2.1. Species identification, enumeration and classification

To begin to understand the potential impact of the OPT on bycatch elasmobranchs, species identified by observers were classified by the risk of extinction according to a variety of domestic and national policies, programmes, and action plans. These included the NSW state government (NSW Fisheries Management Act 1994, https://legislation.nsw.gov.au/view/html/inforce/current/act-1994-038), Commonwealth Government (www.dcceew.gov.au/environment/epbc) and the Action Plan for Sharks and Rays (APASR) via the International Union for the Conservation of Nature (IUCN) Red List of Threatened Species criteria (Kyne et al. 2021). The IUCN Red List was not considered as this would have been largely captured by Kyne et al. (2021). Elasmobranch bycatch was also classified by endemism at various spatial scales including the NSW jurisdiction, Australian national and cosmopolitan (see Kyne et al. 2021). Species that potentially are not threatened with extinction in Australian waters but are in other parts of the world were classified as lifeboat species (Kyne et al. 2021).

2.2.2. Spatiotemporal stratification of assemblages

All data analysis was done using R v.4.2.1 (R Core Team 2022). Preliminary analysis was the same as that described by Barnes et al. (2022b). To address Objective 2 (and 3), a subset of commonly captured species was used. We used this subset to avoid having very rare species in the model as they cause fitting difficulties and removing rarely results in a loss of signal but removes a lot of noise (Warton 2022). Taxa present in 5 % of trips has been used (Stobutzki et al. 2001), but our sensitivity analysis (i.e. removing species with the lowest abundance until the model would converge) showed that contemporary multivariate models could be fit appropriately with species present in ~1.5 % of trips (≥5 trips) and >95 total individuals. Seven species groups and 1 unlikely species level identification were caught regularly according to the common criteria (≥5 trips and >95 individuals) but were not retained for statistical analysis due to their undifferentiated or unconfirmed status. Trawl trips were removed that did not report elasmobranch catches.

Catch and other data from separate trawls were combined to form trip level summaries, thus removing some potential for spatial autocorrelation, as fishers tend to target a specific ground in a trip (Barnes et al. 2021). Controlling for spatial autocorrelation is important for model-based analysis (Zuur et al. 2010). Fishing trips targeting certain grounds also normally ensures that trawls that constitute a trip were generally done at a narrow depth range. Occasional trips with a depth standard deviation of >30 m violated the narrow depth rule and were removed (see Section 3.1). The relationship between mean number and the variance of individual species captured per trip was explored visually to assess which exponential family would be appropriate for modelling. This was also done due to the multivariate nature of this analysis and to determined which type of analysis to perform and thus avoiding problematic inherent errors (e.g. Type 2 errors, Warton et al. 2012).

We modelled spatiotemporal variation in common elasmobranch species multivariate abundance by fitting a multivariate generalised linear mixed model (MVGLMM) from the mvabund R package (Wang et al. 2020). We used a negative binomial distribution as the most flexible and appropriate for count data with mean variance relationship (O'Hara & Kotze 2010). Categorical predictors of variation included 'zone', including north, central and south (Fig. 1), 'season' (cool water <20°C [winter and spring] and warm water ≥20 to 29°C [summer and autumn]) taken from monthly mean water temperatures (www.bom.gov.au), and 'depth' based on the mean observed trawl depth over a trip (shallow <64 m and deep ≥64 m, Fig. 1). Depth analysis was performed only on the north zone as the other 2 zones were only trawled at a narrow range of depths (Fig. 1, see Fig. S1 in the Supplement at www.int-res.com/articles/suppl/n052p149_supp.pdf). This reduced some common species level replication but still satisfied assumptions and diagnostics (Warton 2022). The rationale for the demarcation of stratum within categories includes separating the zones due to the latitudinal positioning combined with bioregional differentiation (IMCRA Technical Group 1998), proximity to the shelf break (i.e. the central zone has less shelf area, Fig. 1) and the aforementioned differential in trawl effort. The season demarcation was based on the northern NSW marine environment being dominated by cool water in winter and spring and warm water in summer and autumn (Barnes et al. 2022a). The depth was demarked shallow and deep at ~64 m due to this being the transition zone between benthic habitats in northern NSW (Boyd et al. 2004, NSW MEMA 2016). For example, the shallow depth has more rocky reefs and input of terrestrial sediments and the deep is dominated by Cenozoic sediment (Boyd et al. 2004). Further, the shallow depth has photophilic habitat-forming organisms (NSW MEMA 2016). Also, 64 m was a central point in the disribution of trawl trips in the north (Fig. 1). All categories (main effects) were treated as fixed factors with the swept area estimated as the distance trawled multiplied by 80% of the combined net headline length (Sterling 2005) as an offset and vessel names as a random effect via the 'anova.manyglm' function with correction for multiple tests using the 'p.uni' function (test = 'LR') with 100 000 permutations. All combinations of categories were tested for interactions, and either were not significant (zone and season) or the model would not converge (zone and depth [north, central and south depths combined], and season and depth [again all zone depths combined]). As such, categories are only reported as main effects. Pairwise post hoc tests (free stepdown corrected for multiple tests) were performed to determine which levels of a category were significantly different, and a species level scores test was done to indicate which individual common species were influential. MVGLMM performance was assessed using diagnostic visualisation of residuals and tested for zero inflation in the response variable using a comparison of expected and realised responses (zeros) (Campbell 2021). Variograms were used to test model residuals for a signal of spatial autocorrelation via the gstat R package (Pebesma 2004). Model-based ordination was performed on common species assemblages to visualise the categorical effects in 2-dimensional space via the ecocopula R package (Popovic et al. 2022). Ordination of this type allows full model conditions (e.g. inclusion of offset, Popovic et al. 2022).

2.2.3. Catch rates

The overall number of common species individuals per trawl trip and for each significant category level determined by the models described in the above section (e.g. zone), was calculated. The trip unit was again chosen as this is how the last OPT observer bycatch survey was largely analysed (Kennelly et al. 1998), thereby facilitating comparison, and this weighting can easily be converted approximately to haul (trip/3) and hour (haul/3), making it comparable with a range of studies. We tabulated this information as means and SE to provide a catch rate (relative abundance) and index of variability. This was only done on the subset of common species (n = 360 total trips), as it would not have been informative for the remaining ~43 species as some identifications were difficult or catches were too low and variable.

3. RESULTS

3.1. Observer effort and technical data

A total of 435 ocean prawn trawl trips via 30 vessels were observed over the 2 yr monitoring programme (Table 1). Overall, most observed trips were in the north (66.44%) zone compared to south (17.24%) and central (16.32%) zones, with the common species subset showing very similar zonal proportions (62.78, 20.28 and 16.94%) (n = 360 trips, Fig. 1, Table 1), which is commensurate of the OPT effort. There was a moderate difference in the number of trips across the season stratum (20.00% more trips observed in the warm water season, Table 1). The number of trips was only slightly lower in the deeper section (46.46%). The mean swept area (weighted to the trip level) trawled (m^2) was greater in the north zone compared to the central and south zones (by 14.33 and 16.13% respectively, Table 1), but was quite similar across depth (9.68% greater in the deep) and seasons.

3.2. Catches

3.2.1. Species identification, enumeration and classification

Considering all identified species and higher taxonomic groupings due to identification difficulty (i.e. not just our common species), a total of ~54 elasmobranch species or higher taxon were captured, from approximately 13 orders and 34 families (Table S1). Most species (61.11%) were captured in multiple trawl trips and the remaining only in single trips or no more than 3 trawls (Table S1). A small number of trips (7.76%) did not catch elasmobranchs, and these were all in the north zone (Fig. S1). Elasmobranch catches included records of species protected in NSW (as per the Fisheries Management Act 1994), including both nurse sharks, sand tiger or Herbst's nurse, *Odontaspis ferox*, and grey nurse *Carcharias taurus* (Table S1). *C. taurus* is also listed at the Australian national level (Environment Protection and Biodiversity Conservation [EPBC] Act 1999, https://www.dcceew.gov.au/environment/epbc). Another EPBC-listed species was

Table 1. Technical and biological variables from the New South Wales ocean prawn trawl fishery during the 2017 to 2019 observer programme. Trips are the numbers observed, vessels are the number that participated in the study and mean swept area is the trip weighted mean ± SD area (m^2) trawled. The common species are listed with number of individuals caught. All data are reduced post filtering for common species, breach of the >30 m SD of the depth trawled per trip or not catching elasmobranchs (relevant to the stratification of assemblages and the weighting of catch rates), except for trips in parentheses which are unfiltered (relevant to the quantification and classification of all elasmobranch bycatch). Data are presented by spatiotemporal strata (zone, season and depth [north zone only]) and combined totals

Subset variable	North	Central	South	Combined	Warm	Cool	Deep	Shallow
Technical								
Trips	226 (289)	61 (71)	73 (75)	360 (435)	200	160	105	121
Vessels	15	10	7	32	22	26	13	13
Mean (±SD) swept area (m^2 × 10^5)	534 (148)	447 (178)	395 (99)	491 (156)	496 (157)	484 (156)	589 (143)	485 (129)
Biological								
Aptychotrema rostrata (eastern shovelnose ray, Rhinobatidae)[a]	1880	1375	2042	5297	1995	3302	501	1379
Urolophus viridis (greenback stingaree, Urolophidae)[a]	3684	0	10	3694	1287	2407	476	3208
Hypnos monopterygius (coffin ray, Hypnidae)[a]	1871	607	387	2865	2154	711	681	1190
Trygonorrhina fasciata (eastern fiddler ray, Rhinobatidae)[b]	133	429	1163	1725	534	1191	23	110
Trygonoptera testacea (common stingaree, Urolophidae)[b]	4	1329	245	1578	319	1259	0	4
Heterodontus portusjacksoni (Port Jackson shark, Heterodontidae)[a]	24	39	1006	1069	258	811	18	6
Dentiraja australis (Sydney skate, Rajidae)[b]	32	104	641	777	499	278	32	0
Mustelus antarcticus (gummy shark, Triakidae)[c]	264	33	81	378	174	204	147	117
Neotrygon australiae (bluespotted maskray, Dasyatidae)	112	6	0	118	8	92	0	112
Parascyllium collare (collared carpetshark, Parascylliidae)[b]	0	92	8	100	8	92	0	0
Asymbolus analis (Australian spotted catshark, Scyliorhinidae)[b]	96	0	0	96	9	87	96	0

[a]Australian endemism; [b]New South Wales endemism; [c]Unique NSW gene pool

recorded: a single school shark *Galeorhinus galeus*, listed as conservation dependent. A small proportion of elasmobranchs were suggested for listing based on IUCN criteria (7.41% or 2 Endangered [EN] and 2 Vulnerable [VU]) according to the APASR (Kyne et al. 2021): thresher shark *Alopias* spp. and school shark *G. galeus* are EN and Sydney skate *Dentiraja australis* and greenback skate *Urolophus viridis* are VU (Table S1). All EN species were captured on single trips and VU more regularly (13.79% trips). The data suggests a relatively large proportion of species were Australian endemic (33.33%) and of these, a large proportion were mainly confined to NSW (38.89%) (Table S1). The data also indicates a small proportion (12.96%) of all species were cosmopolitan, a large proportion (71.02%) of these were only caught on 1 observed trip and the remainder on 2 observed trips (Table S1). A small proportion (16.67%) of all species were classified as lifeboat, with catches of these rare (<0.70% trips) (Table S1). Interactions with other species of conservation concern may have occurred but could not be analysed due to the undifferentiated nature of some (e.g. generic or higher taxonomic classification such as Urolophids not identified to the species level, Table S1).

3.2.2. Spatiotemporal stratification of assemblages

Individuals (potentially common) not identified to species were from the following families or orders: Torpediniformes (torpedo and coffin [numb] rays), Myliobatiformes (eagle and stingrays), Urolophidae (stingarees), Scyliorhinidae (catsharks) and Rajidae (skates). *Aulohalaelurus labiosus* (blackspotted catshark) was a dubious identification to species (Table S1). Eleven species (20.37 %, from 10 families) that could be identified and were common formed the model subset response variable and represented most (96.86 %) individuals across the entire assemblage (Table 1). Nearly all common species were Australian endemic (91.10 %) and of these, 70.00 % were endemic to NSW (Table 1). The most frequently captured species was the eastern shovelnose ray *Aptychotrema rostrata*, whilst the Australian spotted catshark *Asymbolus analis* was the least captured common species (Table 1). Four species were not captured in all zones including *Parascyllium collare*, *U. viridis*, bluespotted maskray *Neotrygon australiae* and *A. analis* (Table 1). Common species *P. collare* was not included in the depth analysis as it was not captured in the north, 2 common species were only captured in the deep (*D. australis* and *A. analis*) and 2 only captured in the shallow section (*N. australiae* and common stingaree *Trygonoptera testacea*) (Table 1). All common species were recorded across seasons (Table 1).

MVGLMMs revealed a significant effect of zone on the common elasmobranch bycatch assemblage ($p < 0.001$, Table 2). All combinations of spatial zone stratum had significant variations in bycatch assemblages ($p < 0.001$, Table 3). Ordination of the common elasmobranch bycatch assemblage showed that most separation by zone occurred on 1 axis (Axis 2, Fig. 2a), with assemblages from the south zone mainly separated from the north and central. However, there was a small area that lacked strong spatial separation of assemblages (i.e. there was an overlap of all 3 zones) in the centre of the ordination plot. MVGLMMs revealed a weak significant effect of season ($p < 0.01$, Table 2). Ordination of the effect of season on common elasmobranch bycatch assemblages lacked separation of stratum (Fig. 2b). An effect of depth was found by MVGLMMs ($p < 0.001$, Table 2). Ordination of the effect of depth showed reasonable separation of some shallow trips on 1 axis (Axis 1, Fig. 2c). Simulation to test zero inflation did not produce excess zeros and diagnostic plots of residuals did not show any concerning shape or pattern (Fig. S2). Thus, the selection of negative binomial family and logit link was the correct method to model common species response data with a mean variance relationship (Fig. S3). Variograms of model residuals showed no structure suggesting no significant spatial autocorrelation (data not shown).

Table 3. Pairwise post hoc analysis to test for differences between significant spatial zones (north, central and south) from the multivariate generalised linear mixed model (Model 3). Shown are the test statistic and multiple test adjusted p-value. All variables were sampled during the 2017 to 2019 observer programme of the New South Wales ocean prawn trawl fishery. ***$p < 0.001$

Factor levels	Test statistic	Adjusted p-value
North vs. south	438.38	***
Central vs. north	228.76	***
Central vs. south	97.42	***

A significant contribution to the variation in elasmobranch bycatch assemblage due to spatial zone was from 5 of the 11 common species (Table 4). The greatest contribution to the zone variation was from *Heterodontus portusjacksoni* (23.90 %) and the least *N. australiae* (0.40 %) (Table 4, Fig. 2). *Aptychotrema rostrata* and eastern fiddler ray *Trygonorrhina fasciata* also made large contributions to the zone variation (21.30 % each). *D. australis*, *H. portusjacksoni* and *T. fasciata* showed the greatest association with trawling in the south (*D. australis* contribution to the zone variation was 10.10 %). *A. rostrata* had a relatively high association with trawling in both the central and south (Fig. 2a). *Trygonoptera testacea* had a significant association with trawling in the central zone (15.82 % contribution to the zone variation).

A. rostrata was the only elasmobranch to contribute to the significant variation in elasmobranch bycatch assemblages because of season (Table 4, Fig. 2b). This species was associated with a group of ~12 trips during the cool water season with a contribution of 35.62 % to the seasonal difference.

Table 2. Summary of ANOVA comparing multivariate generalised linear mixed models of common elasmobranch assemblages. Models 2 to 4 were compared to Model 1 separately to test effects of season (cool water <20°C [winter and spring] and warm water ≥20 to 29°C [summer and autumn]), zone (north, central, south) and depth (shallow <64 m, deep ≥64 m, north only) on the common elasmobranch bycatch community. Shown are the residual degrees of freedom (df res.), degrees of freedom difference (df diff.), Wald statistic and p-value. All variables were sampled during the 2017 to 2019 observer programme of the New South Wales ocean prawn trawl fishery. na: not applicable; **$p < 0.01$; ***$p < 0.001$

Model	Factors	df (res.)	df (diff.)	Wald statistic	p (<Wald)
1	Intercept only	359 225[a]	na	na	na
2	Water temperature	358	1	7.612	**
3	Geographic zone	358	2	26.3	***
4[a]	Depth	224	1	7.143	***

[a]North subset

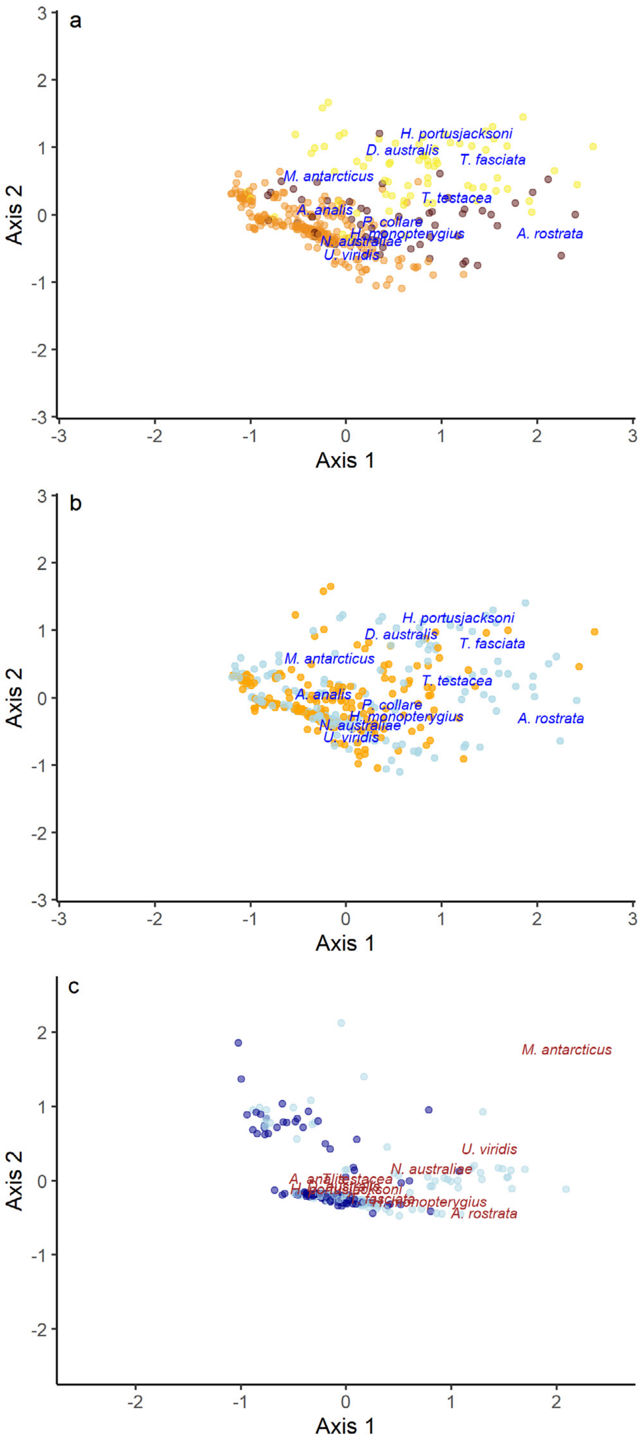

Fig. 2. Model-based ordination of common elasmobranch bycatch assemblages from the New South Wales ocean prawn trawl fishery during 2017 to 2019, with each observed trip represented by a circle. (a) The effect of spatial zone: north (orange); central (dark brown); south (yellow). (b) Season: cool water <20°C (winter and spring, light blue); warm water ≥20 to 29°C (summer and autumn, orange). (c) Depth (north only): shallow (<64 m, light blue); deep (≥64 m, dark blue). Position of species names indicate degree of association with spatiotemporal stratum

All other species contribution to the variation in assemblages due to the season was <15% (Table 4).

A. rostrata and *T. fasciata* drove the variation in common elasmobranch bycatch assemblages because of the depth (Table 4, Fig. 2c). The ordination suggested the strongest depth-associated species was *A. rostrata*, with this species positioned near a relatively large group of shallow trips (~20) (Axis 1, Fig. 2c). This species was also identified as the strongest contributor to the variation in common assemblages due to depth (40.70%), with *T. fasciata* the next strongest (15.66%). All other species contributions were relatively low (<12%, Table 4).

3.2.3. Catch rates

Overall mean catch rates varied among the common species, with *A. rostrata* and *U. viridis* captured at a rate of >10 per trip (Table 4). *Hypnos monopterygius* was captured at ~5 per trip, with *Trygonorrhina fasciata*, *Trygonoptera testacea*, *H. portusjacksoni* and *D. australis* all >2 per trip. Catch rates were very small (<2 per trip) for *A. analis*, *Mustelus antarcticus*, *N. australiae* and *P. collare* (Table 4). Catch rates were highly variable at the species level for all except *A. rostrata* and *H. monopterygius* (i.e. as indicated by standard errors, Table 4). Given the results of the multivariate modelling, it was not surprising that catch rates varied at the common species level among strata. For example, *A. rostrata* was captured at a rate of ~28 per trip (the highest catch rate recorded) in the south and only ~8 per trip in the north (Table 4). *U. viridis* was captured at a relatively high rate in the north (~16 per trip) but was highly variable and not caught in the central zone (see Section 3.2.2). *T. testacea* had a relatively high catch rate (>20) in the central zone (Table 4). *A. rostrata* were caught at a much higher (~50% greater) rate during the cool season and 7 other species were caught at a higher rate during this season, although this was often also highly variable (e.g. *U. viridis*). Also, *A. rostrata* and *U. viridis* were captured at much higher rates in the shallow depth (~50 and ~75% greater, respectively, Table 4), but again this was this was highly variable for *U. viridis*.

4. DISCUSSION

The present study identified and quantified catches of potentially vulnerable elasmobranch taxa. This work was done to begin the assessment of a fishery

Table 4. Common species from the New South Wales ocean prawn trawl fishery during the 2017 to 2019 observer programme with indicator species significance (p), contribution (Contr., %) to the spatiotemporal stratification and catch rates ± SE weighted at the common species subset trip level (n = 360 total trips). All are presented by statistically significant stratum within zone (Z), season (water temperature, W) and depth (D, north only). na: not applicable; ns: not significant p > 0.05; *p < 0.05; **p < 0.01; ***p < 0.001

Subset species	p (Z/W/D)	Contr. (Z)	Contr. (W)	Contr. (D)	North	Central	South	Cool	Warm	Overall	Deep	Shallow
Aptychotrema rostrata (eastern shovelnose ray)	***/***/***	21.30	35.62	40.70	8.34 ± 0.90	22.54 ± 2.92	28.01 ± 4.95	20.64 ± 2.45	9.97 ± 1.25	14.73 ± 1.33 5.19	4.77 ± 0.63	11.4 ± 1.53
Urolophus viridis (greenback stingaree)	ns/ns/ns	1.20	3.17	5.68	16.31 ± 5.09	na	0.14 ± 0.14	15.04 ± 6.51	6.44 ± 2.58	10.26 ± 3.24	4.53 ± 3.32	26.51 ± 8.91
Hypnos monopterygius (coffin ray)	ns/ns/ns	0.84	4.35	11.45	8.29 ± 1.06	9.95 ± 1.26	5.3 ± 0.61	4.44 ± 0.67	10.77 ± 1.13	7.97 ± 0.71	6.49 ± 1.42	9.83 ± 1.52
Trygonorrhina fasciata (eastern fiddler ray)	***/ns/*	21.30	13.58	15.66	0.59 ± 0.12	7.03 ± 1.86	15.99 ± 2.21	7.44 ± 1.17	2.67 ± 0.63	4.8 ± 0.65	0.22 ± 0.07	0.91 ± 0.21
Trygonoptera testacea (common stingaree)	***/ns/ns	15.82	14.67	2.06	0.02 ± 0.01	21.8 ± 4.18	3.37 ± 1.62	7.87 ± 1.92	1.59 ± 0.48	4.39 ± 0.92	na	0.03 ± 0.03
Heterodontus portusjacksoni (Port Jackson shark)	***/ns/ns	23.90	13.83	3.69	0.11 ± 0.04	0.64 ± 0.22	13.82 ± 2.6	5.07 ± 1.25	1.29 ± 0.36	2.98 ± 0.61	0.17 ± 0.06	0.05 ± 0.0
Dentiraja australis (Sydney skate)	**/ns/ns	10.10	0.77	5.77	0.14 ± 0.08	1.7 ± 0.76	8.82 ± 2.13	1.74 ± 0.47	2.5 ± 0.81	2.17 ± 0.50	0.30 ± 0.17	na
Mustelus antarcticus (gummy shark)	ns/ns/ns	1.15	2.26	3.69	1.17 ± 0.3	0.54 ± 0.16	1.11 ± 0.24	1.28 ± 0.4	0.87 ± 0.15	1.05 ± 0.20	1.4 ± 0.51	0.97 ± 0.34
Neotrygon australiae (bluespotted maskray)	ns/ns/ns	0.40	0.19	3.79	0.5 ± 0.28	0.1 ± 0.1	na	0.21 ± 0.11	0.42 ± 0.3	0.33 ± 0.17	na	0.93 ± 0.51
Parascyllium collare (collared carpetshark)	ns/ns/ns	3.08	3.91	na	na	1.51 ± 0.51	0.11 ± 0.11	0.58 ± 0.21	0.04 ± 0.04	0.28 ± 0.1	na	na
Asymbolus analis (Australian spotted catshark)	ns/ns/ns	0.88	7.65	10.50	0.42 ± 0.16	na	na	0.54 ± 0.22	0.04 ± 0.03	0.27 ± 0.1	0.91 ± 0.34	na

and to determine if spatiotemporal stratification differentiates demersal elasmobranch bycatch assemblages to inform future monitoring (and potentially streamline). The information can be used to aid and extend future monitoring and assessment of the potential impact of the OPT on elasmobranchs.

We identified ~54 species, with a portion of these listed conservation priorities by NSW agency legislation. Also, some were listed by other jurisdictions (e.g. the Australian Commonwealth Government), proposed for listing in Australia and listed elsewhere (lifeboat) (Kyne et al. 2021). The OPT elasmobranch bycatch has high endemism, which is expected here (Last & Stevens 2009), and high diversity compared to other similar studies (e.g. similar or higher number of species described) (Stobutzki et al. 2001, Kyne et al. 2002). There are climatic, geomorphological and oceanographic reasons for this high diversity and endemism (Nelson 1978, Boyd et al. 2004, Malcolm et al. 2011, Harrison & Noss 2017). Extractive activities in an area with high diversity and endemism can be problematic as this creates a macroevolutionary sink dynamic (Goldberg et al. 2005). For example, the sink creates flow on pressure on surrounding sources and thus can promote a loss of genetic fitness (Goldberg et al. 2005, Cure et al. 2017). However, the reduction in OPT effort (Barnes et al. 2022b, Johnson & Barnes 2023) coupled with the relatively new NSW and Australian Commonwealth Marine Protected Area network (NSW MEMA 2016) may well counteract the force of a sink in such an area (Samaai et al. 2020). These uncertainties demonstrate the importance of monitoring, and listed and endemic species should be prioritised for follow up research. The identification and conservation classification of species is only an early step in delivering a thorough assessment.

Elasmobranch interactions in the OPT has not been comprehensively investigated before but it has been done on a smaller scale (e.g. Silburn et al. 2020) or just on certain groups (e.g. commercial or byproduct species) of elasmobranchs (Kennelly et al. 1998, Macbeth et al. 2008). Large-scale traditional observer-based quantification of fishery bycatch is impactful on resources, possibly explaining but not justifying, the lack of such programmes (Borges et al. 2004, Johnson & Barnes 2023). Unfortunately, the significant stratified spatiotemporal variation in elasmobranch assemblages in our common species subset means that future observer sampling cannot be readily streamlined in a robust and representative manner. However, the present study is just a short-term snapshot, therefore this could change with further work in the near future. Spatiotemporal stratification of bycatch assemblages has previously been found here (Kennelly et al. 1998) and elsewhere (e.g. Stobutzki et al. 2002), and is also due to the same drivers listed above (e.g. climatic, etc.) amongst other possibilities (i.e. the OPT zone differential in trawl effort). The unbalanced depth sampling across the spatial zone stratum could also have an influence on the significant variation between zones. Unfortunately, the lack of model convergence when testing an interaction between zone and depth means it is necessary to treat these 2 categories as separate at present. Kennelly et al. (1998) did find significant spatial variation of an all-bycatch species assemblage during their study of the OPT. *Aptychotrema rostrata* was the primary indicator and contributor to strata differentiation identified by all 3 of our models; this species has also been previously reported as a relatively common catch in the OPT (Kennelly et al. 1998) and in neighbouring fisheries (Kyne et al. 2002).

The reported low catch rates of common species showed patterns generally consistent with the elasmobranch assemblage spatiotemporal stratification. These rates should be used as a foundation to assess relative abundance and help monitor the elasmobranchs over time. Whilst there was some favourable comparison with Kennelly et al. (1998) for 2 species, the Rhinobatids (*A. rostrata* [the most relatively abundant] and *Trygonorrhina fasciata*), too much has potentially changed in ~30 yr, both operationally (i.e. effort reduction, increased fishing power, etc.) and environmentally (Hobday & Lough 2011, Melbourne-Thomas et al. 2021), to speculate about stability in relative abundance.

The very nature of trawl gear means that nontarget catches are inevitable (Broadhurst 2000). Trawling relatively fine mesh nets near a variety of habitats in warm water promotes interactions with diverse bycatch assemblages. The literature suggests that demersal penaeid trawling is no exception (Watson & Goeden 1989, Robins et al. 1999, Ye et al. 2000), as do our results. Unfortunately, the results presented here cannot be extended to determine the immediate impact of OPT on demersal elasmobranchs. Rather it suggests a small number of species should be prioritised for research and assessment. For example, *Dentiraja australis* is conservation-listed and NSW endemic, and a large portion of its range may be trawled by the OPT (Daley & Gray 2021, Kyne et al. 2021). More broadly, another comprehensive bycatch quantification study should be implemented soon (<5 yr). Only when these next programmes are completed will the OPT impact on elasmobranchs be realised.

Fishery interactions with a species does not necessitate unsustainable depletion (Campbell et al. 2018). A large number of the OPT bycatch elasmobranchs are released (Kennelly et al. 1998, Barnes et al. 2022b) and some of those retained are periodically stock assessed (e.g. Peddemors 2015). Therefore, future research should determine the post release survival and nonlethal capture impacts (e.g. Adams et al. 2018), particularly for those identified as assessment priorities or commonly retained. This has already been done for 2 species (Campbell et al. 2018), with survivorship perhaps reasonable (>50%) but species specific and influenced by a variety of factors. Other studies have found survivorship to be ~50% (Stobutzki et al. 2002, Laptikhovsky 2004, Enever et al. 2009). Further, turtle excluder devices can also release non-reptilian large-bodied animals before landing and have been shown to be reduce elasmobranch bycatch (Campbell et al. 2020). However, these are not deployed in the OPT due to the lack of turtles, and efficacy is limited to larger animals and those with particular behavioural traits (Broadhurst 2000). However, they could still possibly reduce OPT elasmobranch interactions and deserve thorough investigation. Other new gear and vessel technology is now available to mitigate or quantify bycatch such as image-based release during trawling and monitoring of landings (Sokolova et al. 2021, Abangan et al. 2023). Integrating contemporary technology into monitoring could reduce the expense of such programmes. This is particularly relevant for relatively low monetary value but important fisheries such as the OPT (Johnson & Barnes 2023). Initiatives to balance the use of resources but to ensure representativeness of monitoring the OPT have begun in the form of an empirically optimised sampling programme (Johnson & Barnes 2023). Such programmes use data and analysis to optimise sampling fractions and generally provide a range of precision statistics and equivalent sampling effort (Borges et al. 2004) which can demonstrate to financial managers the return for a given investment. Basic biological information such as age and growth data is lacking for many elasmobranch bycatch species (Kyne et al. 2021, Mejía et al. 2023) and should be addressed to provide productivity, and longevity (etc.) information. Also, NSW observer identification could be improved, especially for less common elasmobranchs, which will increase replication during traditional observer studies (currently the most comprehensive).

More than 50 elasmobranch species were found to be interacting with the OPT, with some listed and endemic. Also found were a spatiotemporally stratified common elasmobranch bycatch assemblage and mainly relatively low catch rates. Thus, we have provided an early step to understanding the impact of the OPT. The nontarget impact of the OPT and other similar fisheries is often not investigated. This is concerning as the OPT and other demersal trawlers have the potential to deplete elasmobranchs. It is recommended that follow-up research and monitoring is done in a timely manner to build on the foundation of this study. Ultimately, further investment must be made to understand the risk this and other fisheries may pose with technology now available to partly alleviate the financial burden. Future complimentary work may reveal that the OPT is not a substantial risk to elasmobranchs. However, if mitigation is required, research is needed to target it in order to efficiently remove the risk, thus helping to arrest the global decline of elasmobranchs.

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