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Anchoring from shipping as a disturbance agent to temperate rocky reef fish: marked shifts observed in trophic and taxonomic guilds

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Anchoring impacts to marine environments from large, ocean-going ships is increasingly recognized as a global threat to marine biota. To date, no replicated assessment examining anchor disturbance to fish assemblages exists at the scale of ocean-going vessels. Here we aim to fill this important knowledge gap, using the Port Kembla Anchorage in SE Australia as a case study. We predicted that demersal fish on temperate rocky reefs (>30m) exposed to anchoring activities would differ significantly to those that were 'anchor-free'. Using Baited Remote Underwater Video (BRUV) we assessed species and functional groups using a full-subsets generalized additive mixed modelling approach, including fine-scale reef variables as covariates to account for natural spatial variability and to improve estimates. Reefs exposed to anchoring (ie. disturbed) was the most important predictor for the total abundance of fish, with twice as many individuals when compared to undisturbed reefs (anchor-free). Abundance measures were largely driven by the shoaling zooplanktivore; *Atypichthys strigatus*, with near four-fold increases of this trophic group on anchored reefs. In contrast, the abundance of other taxa including, *Meuschenia freycineti* and demersal elasmobranchs combined decreased two to three-fold on disturbed reefs. These results indicate anchoring activities can have ecosystem-wide impacts to fish assemblages underscoring the importance of better managing anchoring near ports globally.

1 Introduction

Maritime transport is the backbone of modern trade and globalized economies with >80% of goods transported by large container ships and bulk carriers, summing to more than 120,000 port visits per annum (UNCTAD [United Nations Conference on Trade and Development], 2024). With this heavy reliance on shipping, comes increasing potential for impacts to marine environments and their associated biota. Numerous stressors associated with shipping have been well documented, although the focus has largely been on shipping as a vector for invasive species as well as a source of chemical and underwater noise pollution (Jägerbrand et al., 2019; Qi et al., 2024). However, evidence is building on the adverse impacts of anchor and chain scour from ships as an important source of disturbance to marine systems (Davis et al., 2016; Broad et al., 2020). Spatial analyses using ship tracking data indicate that this disturbance can extend far beyond designated anchorages near ports (Steele et al., 2017) and disturbances from this routine operation can occur over very large spatial scales (Davis et al., 2022; Watson et al., 2022; Broad et al., 2023), contributing to more than 48% of all seabed disturbance in some coastal areas (Watson et al., 2020).

Rocky reefs and the biota associated with them, constitute significant reservoirs of biodiversity vital for temperate marine systems (Bennett et al., 2016). They offer abundant food resources, as well as habitat refugia attractive to economically important fish assemblages (Tuya et al., 2009; Gaylard et al., 2020). At mesophotic depths (>30m), temperate reefs are primarily dominated by sponge fauna, ahermatypic cnidarians (non-reef builders) and bryozoans (lace corals) which have collectively been referred to as 'Marine Animal Forests' (Rossi et al., 2017). Researchers have previously suggested that these offshore, mesophotic reefs are largely exempt from the common stressors actively degrading shallow nearshore reefs, serving as refugia and sustaining shallow reef fish communities (Bongaerts et al., 2010; Assis et al., 2016). However, assessments of many mesophotic systems remain in their infancy, as they are logistically difficult and expensive to investigate. Therefore, in many temperate areas, the condition of these types of reefs has yet to be determined. Recent research has uncovered chronic degradation of mesophotic reefs at some locations from marine industries (Bennett et al., 2016; Broad et al., 2023), which are unlikely to support the requirements of diverse fish assemblages and may compromise their functional traits, limiting their range (McCauley et al., 2015).

Natural disturbances, often associated with weather, are commonplace in many shallow marine systems and can shape communities, as well as the species within them (Pickett and White, 1985). In deeper, relatively more stable waters, however, disturbances from offshore marine industries, such as demersal fishing, mining and shipping have, in many cases, transformed marine ecosystems over large spatial scales (Ibon, 2013; Broad et al., 2020; Gaylard et al., 2020). Mechanical operations and disturbances from marine industries can simplify complex environments, reducing the tri-dimensional complexity of coral or rocky reefs and the ecosystem engineers they house (Alvarez-Filip et al., 2009;

Broad et al., 2023). Therefore, chronic damage to reefal habitat or removal of biogenic structures by anchor scour would be expected to have flow-on effects for associated motile fauna such as fishes (Forrester et al., 2015; Roberts et al., 2016; Flynn and Forrester, 2019).

Research examining anchor disturbance to date has largely focused on impacts from recreational boating which consistently reveal decreases in the biogenic structure and habitat complexity which inevitably impact demersal fish assemblages (Lanham et al., 2018; Flynn and Forrester, 2019; Broad et al., 2020). Anchor damage from recreational vessels on tropical coral reefs have resulted in wide-spread changes to the structure of reef fish assemblages, with reductions in species richness, as well as alterations in the distribution and abundance of several trophic guilds (Flynn and Forrester, 2019). In temperate regions, impacts of anchoring activities on fish assemblages are contradictory; seagrass loss can negatively affect some guilds (Kiggins et al., 2018), while recreational boat mooring research examining soft sediment environments reported impacts to fish assemblages as subtle and only detectable at the smallest of scales (m's) (Lanham et al., 2018). A key point is the marked discrepancy in the scale of disturbance generated by small recreational boats versus large ocean-going commercial vessels, with the later representing a significant gap in our knowledge (Deter et al., 2017; Broad et al., 2020; Watson et al., 2020, 2022).

Despite clear evidence that anchor scour over reef from ocean-going vessels damages and removes sessile biota such as sponges (Broad et al., 2023), the effects on associated fauna, such as fishes, have not yet been assessed. Here, we seek to fill this important knowledge gap and test the hypothesis that, fish on temperate mesophotic rocky reefs frequently 'anchored' on will differ significantly relative to 'anchor-free' reefs and examine effects across species, taxonomic groups and trophic guilds. We sought to reduce the impacts of potential confounding factors by incorporating localized seascape variables (Rees et al., 2018a, b) including reef structure, depth and the proportion of reefal habitat at sample locations. This study extends the knowledge of demersal reef fish distribution and abundance on temperate mesophotic reefs generally, though notably is the first quantitative assessment of anchor scour impacts from maritime trading ships (>100m in length) to any demersal fish population. Importantly, the outcomes of this work will directly inform future decision-making for the management of shipping in marine estates locally (BMT WBM, 2017) and across the globe.

2 Methods

2.1 Study region and sampling design

This study was done along a ~25km stretch of coastline in south-eastern, Australia (Figure 1). At the time of sampling, ships anchored in an unregulated anchor roadstead in this region, over an area spanning ~220km^2 (Davis et al., 2022) in water depths ranging from 35-60m. The seabed in this depth range is characterized by
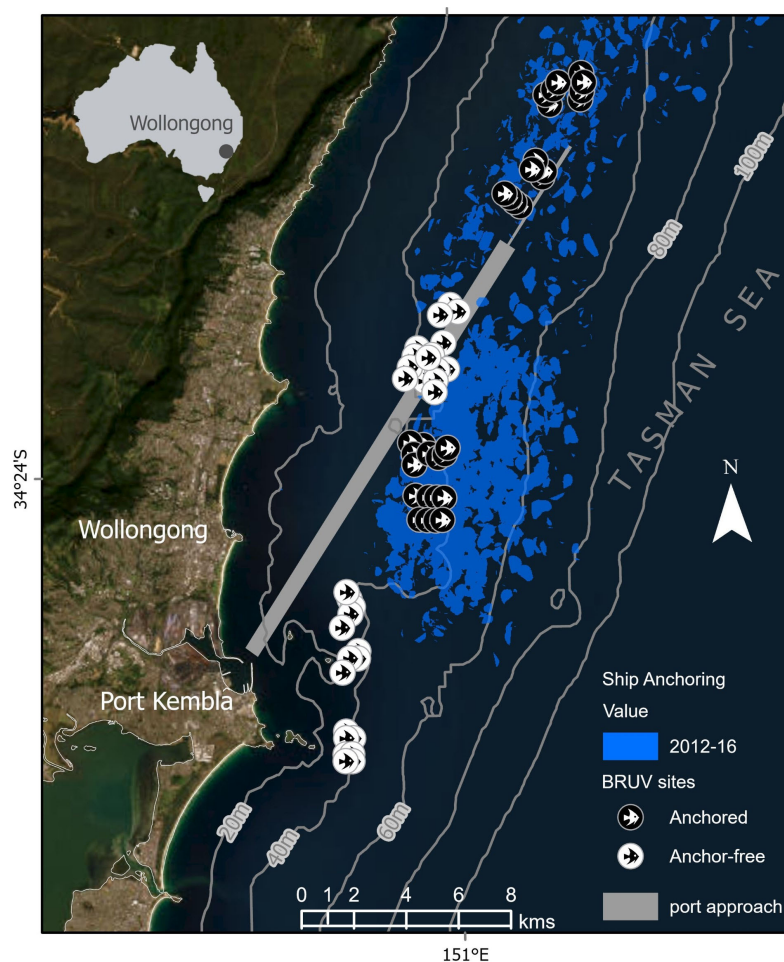
FIGURE 1
Map showing the study sites offshore from Wollongong, Australia. Baited Remote Underwater Video (BRUV) sites on reefs anchored on (black circles) and those that were anchor-free (white circles). The anchor roadstead, depicted in blue, reflects anchoring events between Sept 2012 and June 2015. Note the shipping approach to the Port Kembla Harbor is depicted as a linear grey feature. Depth contours along the continental shelf are depicted as thin grey lines.

extensive, predominantly low-profile rocky reef, including channels containing mixed reef with boulders, gravel and sand (Linklater et al., 2019).

To delineate the anchor-roadstead and identify anchor-free reference areas we used a similar approach to Broad et al. (2023). We identified vessels at anchor using positional information (Automated Identification Systems (AIS) between September 2012 to June 2016 – just prior to fish sampling. This spatial information was then overlaid on high resolution maps with 5m gridded bathymetry derived from multibeam echosounder (MBES) surveys (www.aodn.org.au) allowing us to identify 'anchored' and appropriate 'anchor-free' reference locations on rocky reefs within a depth range ~35-50m. Conservative estimates based on the AIS data confirmed that reference areas had not been anchored upon for > 4 years.

Baited Remote Underwater Video (BRUV) was used to test the hypothesis that fish assemblages would differ across reefs 'anchored' and 'anchor-free' areas. BRUV has been used routinely for sampling fish assemblages as it is non-destructive and provides useful

measures of species richness as well as relative abundance estimates of diver-shy and economically important fish assemblages (Watson et al., 2005; Kelaher et al., 2014; Rees et al., 2018a, b; Rees et al., 2021). As bait attracts fish, BRUVs also reduce the problem of zero-inflated datasets (Schramm et al., 2020). BRUV was particularly suitable in this study as units can be deployed in water depths exceeding 30m and considered too deep for diver-based assessments, particularly over large spatial scales (km's).

The BRUV system used in this study consisted of a single GoPro HERO3+ camera (www.gopro.com) mounted within a waterproof housing camera housing. BRUV's were deployed across four sites: two sites of 'anchored' reef and two sites that were 'anchor-free'. Anchored treatments were interspersed in space and time and within each site we deployed 16 BRUV replicates (Figure 1) for a total of 64 BRUV deployments in the study. Each BRUV was deployed for 35 minutes to allow for settlement time and to achieve a 30-minute video sample. Previous research has demonstrated that 30 minute is an appropriate set time to provide a representative sample of temperate rocky reef fish

assemblages (Harasti et al., 2015). Each sample was separated by a minimum distance of 200 m, consistent with other assessments using this method in the region (Kelaher et al., 2014; Rees et al., 2018a; Knott et al., 2021). Sampling was interspersed over several days during the late Austral winter and early spring (July to September, 2016) with all collections done in daylight hours (0900 to 1600) to alleviate potential effects of crepuscular feeding behaviors (Wraith et al., 2013). BRUV deployments used ~500 grams of thawed and crushed pilchards (*Sardinops sagax*) which has been determined as the optimal bait across a range of feeding guilds (including herbivores) (Harvey et al., 2007; Wraith et al., 2013) and was replenished for each subsequent deployment.

2.2 Analysis of video footage and guild identification

Videos were analyzed in the laboratory by a single experienced observer (AB) using the BRUV analysis software *EventMeasure* (http://www.seagis.com.au). All fish were recorded if they swam within 2 meters of the bait bag, providing a standardized (FOV) ~9 m^3 (Malcolm et al., 2007). Species richness, total Max N, and Max N of each fish species were recorded for each video sample. Species richness (SR) is the sum of all the species recorded for each video sample and Max N is an estimate of the total abundance observed in one frame of a given species. Fish that displayed sexual dimorphism (e.g. Labrids) or protruding sexual organs (e.g. claspers in male elasmobranchs) provided improved estimates of these metrics and we adjusted Max$N$ counts of those species (see Supplementary Table 1). The total relative abundance for each sample, Total Max$N$, was determined by summing the Max$N$ for all species over the 30 min sample. After video analysis, fish were classified into taxonomic groups (subclass/family) and trophic guilds to test for differences (for a full list with definitions see Supplementary Tables 1, 2). The later was based on their diet, previous research (Wraith et al., 2013; Swadling et al., 2019) or review of other supporting literature (Bell et al., 1978; Choat and Ayling, 1987; Bulman et al., 2001). In instances where no trophic information was available for a particular species, a similar, closely related species was used for inference. Species were grouped into one of three trophic groups; (i) herbivores; (ii) zooplanktivores and (iii) generalist carnivores.

2.3 Benthic classification

Fine-scale habitat features for each BRUV replicate were analyzed using *TransectMeasure* (www.seagis.com.au; see Supplementary Tables 3, 4) following the method described in McLean et al. (2016). A 5 x 4 grid was overlaid on a high definition still frame from each BRUV deployment delineating cells to be scored (n=20). For each of the 20 cells, the dominant habitat type and vertical relief was scored following the CATAMI classification scheme (Althaus et al., 2015). The main habitat classifications were defined as either; unconsolidated sediments

(sand or gravel), consolidated rock, macroalgae (any >5cm), sponges, ascidians and open water. For each cell containing benthos, an estimate of vertical relief was scored between 0 – 5 (Wilson et al., 2007). The BRUV Field of View (FoV) was also scored as 'open water', 'video facing-down', 'facing-up' or 'limited FoV' e.g. obstruction by a rocky outcrop. Data were exported from *TransectMeasure* using R scripts available from Langlois (2017). Cells of open water were removed before calculating the percentage cover of habitat and the mean vertical relief for each deployment. Depth (m) was recorded for each BRUV deployment in the field using the research vessel's sounder.

2.4 Habitat mapping

Multibeam surveys (MBES) in the vicinity of the Port Kembla roadstead were completed by NSW state government during the period Oct 2014 – May 2017 and made available via the Australian Oceanographic Data Network (AODN: https://www.aodn.org,au). Survey methods are detailed in Linklater et al. (2019) utilizing both a Geoswath 125khz (Teledyne GeoAcoustics, U.K) and an R2Sonic 2022 (R2Sonic, USA) with a POS MV vessel reference system for motion correction. Bathymetric data were issued as depth-weighted averages of cleaned (Cube-modelled) multibeam soundings binned at 5m resolution and reported relative to Australian Height Datum. Data were imported to ArcMap (ESRI, USA) and additional layers of hillshade, slope and ruggedness calculated using tools including Benthic Terrain Modeller. These layers were then used to inform the selection of BRUV deployment sites over reef across the northern impact, southern impact and southern control locations. Bathymetric data for the section of the seabed in the northern control location were unavailable at the time of fieldwork planning. In this instance, a random location for each BRUV drop was nominated and then the presence of reef confirmed using the vessel's onboard echosounder when on site. To calculate measures of reef structural complexity, the standard deviation of bathymetry values within 25 m, 50 m and 100 m radius buffers were calculated from the MBES data for each BRUV deployment.

2.5 Data analysis

The influence of anchoring and fine-scale habitat variables on the fish assemblage was analyzed using Generalized Additive Mixed Models (GAMMs; Hastie and Tibshirani, 1986; Hastie, 1990). GAMMs were chosen over Generalized Linear Mixed Models (GLMMs) to allow for non-linear relationships between the response and explanatory variables. A full-subsets modelling approach was used to determine which of the available explanatory variables were most important in influencing the fish assemblage (Fisher et al., 2018). Using this approach a complete model set was constructed and the selection of models was based on Akaike Information Criterion optimized for small sample sizes (AICc). Models within two delta AICc units (ΔAICc < 2) of the

model with the lowest AICc were selected (Burnham and Anderson, 2002). If there were multiple models within two delta AICc unit, we selected the most parsimonious model(s) with the fewest explanatory variables. The relative importance of each explanatory variable was calculated by summing the AICc weight values for all models. Variables with higher summed values represent increased importance of that explanatory variable (Burnham and Anderson, 2002). To avoid issues of collinearity, candidate models could only contain explanatory variables with Pearson correlation coefficients < 0.28 (Graham, 2003). The maximum number of explanatory variables for each candidate model was limited to three to reduce over-fitting and 'site' was included as a random effect in all models to account for overdispersion and correlation in the data (Harrison, 2014). In all candidate models, explanatory variables were fitted with smoothing splines, with the 'k' argument limited to four.

Models were run on species richness, total relative abundance (total Max*N*), abundance of trophic guilds, taxonomic groups and the abundance or presence/absence of common species present in >30% of samples (~16 spp.). Untransformed measures of the fish assemblage were used as response variables. The models for species richness were fitted with a Gaussian error distribution while models for total abundance included a negative binomial error distribution. GAMMs for the abundance of *Ophthalmolepis lineolata* and *Bodianus unimaculatus* were fitted with a Poisson error distribution. A Tweedie error distribution was fitted to all other relative abundance models due to the large number of zeroes (Tweedie, 1984). Taxa observed in low relative abundances were transformed to presence/absence and modelled using a binomial error distribution; this method has been shown to be an effective method for extracting ecologically meaningful data in rare or cryptic species (Joseph et al., 2006). Prior to analysis, the distribution of explanatory variables was plotted. Explanatory variables displaying a skewed distribution were transformed to ensure an even spread of values across the observed range. All data manipulation, analyses and plots were developed using the R language for statistical computing with the 'dplyr' (Wickham et al., 2020), 'ggplot2' (Wickham, 2016), 'mgcv' (Wood, 2011) and 'FSSgam' (Fisher et al., 2018) packages.

3 Results

3.1 Overall assessment

In total, we observed 68 species of temperate reef fish across 35 families from 64 BRUV deployments (4684 individuals; Supplementary Table 1). Overall, there was very strong evidence that 'anchored' rocky reefs influenced the total abundance of temperate reef fish observed, with both 'anchored' treatment and consolidated rock (%) included in the top model (Table 1; Figure 2). There were twice as many fish on 'anchored' reefs (98 ± 12 SE) compared to those that were 'anchor-free' (48 ± 6 SE) (Figure 3), with ~ 40% of the fish taxa examined containing the 'anchored'

treatment in the top model (Table 1). Despite trends in fish abundance, anchor treatment had no effect on the richness of species occurring on rocky reefs, although as reefs increased in structural complexity, so did fish diversity (Figure 2).

3.2 Trophic guilds

Fishes within trophic guilds showed contrasting responses to anchor treatments. The zooplantivores, showed a distinct, almost four-fold increase in abundance on 'anchored' reefs (75 ± 12 SE) compared to 'anchor-free' reefs (~20 ± 4 SE). Increasing reefal habitat was also a strong predictor of abundance for this guild (Figure 3). Similarly, herbivores showed preference for greater rocky reef habitat (Figure 3), although they were indifferent to anchor treatments. In contrast, the null model was the most parsimonious for the abundance of generalist carnivores, indicating that neither anchoring treatment, nor any habitat features had any effect on the distribution of this trophic guild (Table 1; Figure 2).

3.3 Common taxa

Just six of the sixteen common taxa examined (observed in >30% of samples) revealed 'anchored' reefs to be a strong predictor of their abundance (Table 1; Figure 2). The top model for the common shoaling, zooplanktivore, *Atypichthys strigatus* included 'anchored' reefs explaining 20% of their distribution with abundance measures more than three times greater (40 ± 14 SE) than 'anchor-free' reefs (12 ± 5 SE) (Figure 4). Notably, there were three top models within 2 AICc for *A. strigatus* with all models including the 'anchored' treatment (Table 1). In contrast, the probability of detection of benthic elasmobranchs and the leatherjacket, *Meuschenia freycineti* were three and two times more likely to be observed on 'anchor-free' reefs (Table 1; Figure 4).

While the most parsimonious model for the occurrence of the economically valuable sparid - snapper, *Chrysophrys auratus* similarly included 'anchored' reefs and the habitat variables, mean relief and structural reef complexity (<25m) derived from bathymetric data (Table 1; Figure 2). Overall, these variables explained 30% of the abundance of snapper. Although, 'anchored' reefs was included in the top model, the effect size was small, and estimates had high variability (Figure 5).

In comparison, the Grey Morwong *Nemadactylus douglasii* included 'anchor-free' reefs as well as reef structural complexity (<100m) in its top model and these variables explained 38% of its abundance (Table 1; Figure 2). Evidence for preference of grey morwong for undisturbed reefs was very weak, with little distinction in abundance between 'anchored' (0.94 ± 0.2) and 'anchor-free' reefs (1.14 ± 0.2) (Figure 5). Likewise, for the Maori wrasse, *Ophthalmolepis lineolatus*, the top model included 'anchored' reefs, however the effect was weak with their abundance on 'anchor-free' reefs being 3.3 (± 1) compared to 2.5 (± 0.7) for

TABLE 1 Top generalized additive mixed models (GAMMs) for predicting the distribution of species, taxonomic or functional groups from full subset analyses based on Akaike Information Criterion corrected for small sample sizes.

Response Variable	ΔAICc	ωAICc	R^2	edf	Best Model
Overall fish assemblage					
Species richness	0.154	0.143	0.404	7.96	Mean relief
Total abundance	0.409	0.2	0.150	4	Anchored + Consolidated rock
	1.758	0.102	0.156	4.66	Anchored + mean relief
Functional group					
Zooplanktivores	0	0.238	0.1882	4.66	Anchored + Consolidated rock
Herbivores	0	0.195	0.196	4.07	Consolidated rock
Generalist carnivore	1.159	0.158	0.162	9.43	Null
Taxonomic group					
Elasmobranchs	0.155	0.12	0.137	3	Anchor-free
Species					
Atypichthys strigatus	1.632	0.113	0.197	3.2	Anchored
Aulopus purpurissatus	0.337	0.043	0.148	2	Null
Bodianus unimaculatus	0	0.236	0.442	4.73	Depth
Chyrsophyrs auratus	0	0.994	0.298	13.46	Anchored + Mean relief + < 25m
Caesioperca lepidoptera	1.686	0.204	0.430	13.99	Consolidated rock + < 100m
Hypoplectrodes maccullochi	0	0.992	0.471	9.21	Consolidated rock
Mecaenichthys immaculatus	0	0.135	0.187	4.09	Consolidated rock
	1.516	0.063	0.155	7.34	Mean relief
Meuschenia freycineti	0	0.12	0.222	3	Anchor-free
Meuschenia scaber	0	0.214	0.380	5.46	Null
Nemadactylus douglasii	0	0.641	0.372	9.09	Anchor-free + < 100m
Ophthalmolepis lineolatus	0	0.101	0.525	10.78	Depth + Mean relief
	1.994	0.037	0.492	10.83	Anchor-free + Depth
	0.547	0.077	0.5497	11.49	Depth + Consolidated Rock
	1.896	0.039	0.55796	12.81	Depth + < 100m
Parma microlepis	0	0.721	0.172	2	Null
Pseudocaranx dentex	0	0.403	0.047	4.31	Consolidated rock + Depth
Scorpis lineolata	0.835	0.255	0.344	15.69	Depth + < 100m
Trachurus novaezelandiae	0	0.586	0.167	5.51	Depth + Mean relief
Upeneichthys lineatus	0	1	0.282	6.43	Null

The delta AICc (ΔAICc; differences in Akaike Information Criterion), AICc weights (ωAICc), explained variances (R^2) and effective degrees of freedom (EDF) are reported for model comparison. Models were selected based on the most parsimonious model (fewest variables) within ± two units of the model with the lowest AICc. The explanatory variables included; Consolidated rock (% of rock within the BRUV Field of View (FOV)); Reef structural complexity (SD of bathymetry within 25m, 50m, 100m of replicate); Treatment (Anchored versus Anchor-free); Water Depth (30-50m); Mean (Vertical) Relief in BRUV (0-5). Visual representations of these data are illustrated in Figures 3-6.

those disturbed (Table 1; Figure 2). In contrast, water depth was a much stronger predictor, where the abundance of Maori wrasse declined with increasing depth (Figure 5).

Importance scores for two of the zooplanktivores, *Caesioperca lepidoptera* and *Scorpis lineolata* indicated weak support for 'anchored' reefs as a predictor for their distribution (Table 1;

Figure 2). Although habitat variables best explained their distributions with the top models for *C. lepidoptera* including increasing consolidated rock and reef structural complexity <100m (Figure 6). While the top models for *S. lineolata* included shallower reefs and increased reef structural complexity <100m (Table 1; Figure 2).

FIGURE 2
A heat map highlighting the relative importance (0.0-1.0) of each explanatory variable (x-axes) for individual species, taxonomic or functional groups (y-axis). Relationships with response variables are indicated across a gradient; positive (red), zero (white) and negative (blue). The top models selected for the most parsimonious models are indicated (X, see Table 1). Relief: rating ranging from '0' (~0°flat substrate) to '5' (Vertical wall. ~90° substrate elevation) adapted from Wilson et al. (2007). Depth: ranged from 35 to 50m; Reef structural complexity (SD of bathymetry within 25m, 50m, 100m of replicate) derived from bathymetric maps; Treatment ('anchored' versus 'anchor-free').

Anchoring was not an important predictor for several species, including *Pseudocaranx dentex*, *Trachurus novaezelandiae*, *Mecaenichthys immaculatus*, *Hypoplectrodes maccullochi*, and *Bodianus unimaculatus* with habitat metrics providing more explanatory value in their spatial distribution (Table 1; Figures 2, 6). In contrast, no habitat variables included in our assessment, nor either anchor treatment, were of explanatory value in the abundance of several taxa, including *Aulopus purpurissatus*, *Meuschenia scaber*, *Parma microlepis* and *Upeneichthys lineatus* (Table 1; Figure 2).

4 Discussion

These findings point towards a complex restructure of the demersal reef fish assemblage on mesophotic rocky reefs in response to anchoring activities over large spatial scales (10's km). Forty percent of the taxa we examined were affected by anchoring activity on reefs in some way. Negative impacts on fishes were anticipated given that mechanical disturbance degrades reef ecosystems (Broad et al., 2023) with flow-on effects expected as fishes lose habitat refugia as well as food resources (Aburto-Oropeza et al., 2015; Forrester et al., 2015; Flynn and Forrester, 2019). Surprisingly however, we observed no differences in species richness between treatments and marked increases in fish

abundance; largely inflated by the zooplanktivore trophic guild. These outcomes are in stark contrast with the findings of anchoring research on tropical coral reefs; there researchers reported declines in both species' diversity and fish abundance, with 95% declines in the abundance of sponge-feeding fishes on disturbed reefs (Flynn and Forrester, 2019).

More surprising was our observation that 'anchored' reefs benefitted some taxa. Fishes in the zooplanktivore trophic guild typically dominate fish assemblages in this region, making up almost half of the biomass on temperate rocky reefs in SE Australia (Champion et al., 2015; Truong et al., 2017), yet their abundance increased a further three-fold on reefs disturbed by anchors. There is clear evidence from several studies that taxa from this trophic guild respond positively to seabed disturbance (Pauly et al., 1998). For example, using manipulative experiments in SE Australia, researchers report increases in planktivorous fish >200% in response to water discharged from desalinization plants installed on rocky reefs (Kelaher et al., 2020). Similarly, zooplanktivores in the Gulf of California, Mexico also proliferated in response to increased disturbance from industrial bottom fishing on rocky reefs (Aburto-Oropeza et al., 2015). An important local representative of the zooplankivore guild - mado, *Atypichthys strigatus*, the dominant zooplanktivore we observed, saw near four-fold increases in response to 'anchored' reefs. We suggest two possible mechanisms contributing to these findings. First,
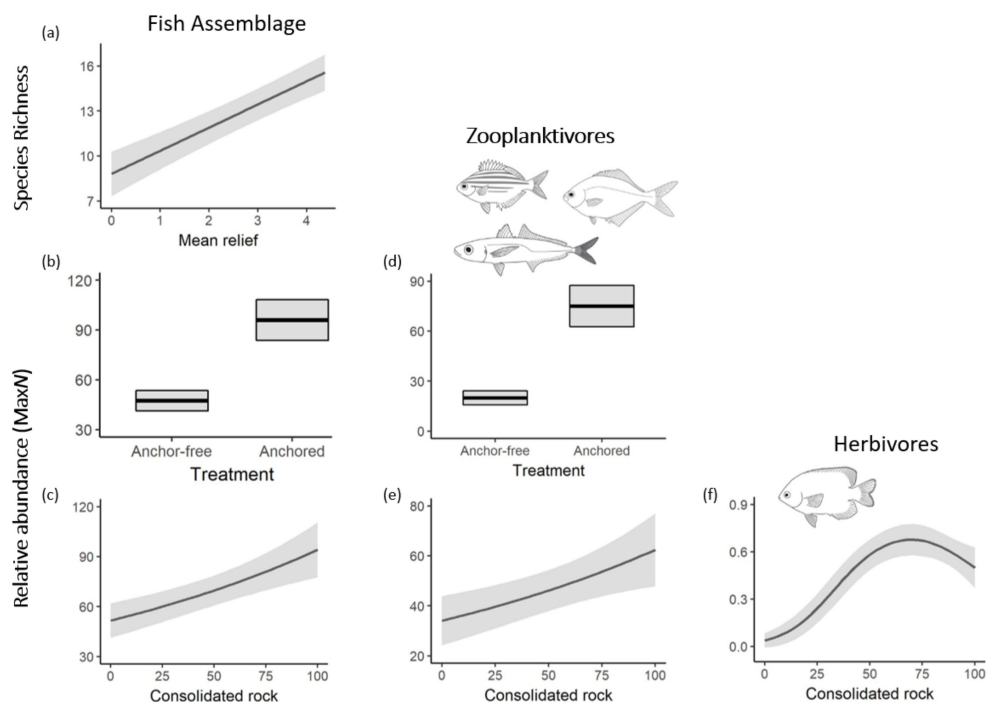
FIGURE 3

Mean predictions from the best fit models (lowest AICc) for temperate reef fish across 'anchored' (treatment) and 'anchor-free' reefs for **(A)** species richness (SR) and **(B, C)** total relative abundances (MaxN) of the overall fish assemblage; **(D, E)** zooplanktivores and **(F)** herbivores; from full subset Generalized Additive Mixed Model (GAMM). Solid lines represent fitted GAMM predictions and shaded areas define standard errors ~25 and 75 quartiles around the predictions. The summary of each model is provided in Table 1.

there is experimental evidence that mado respond to mechanical disturbances with increased feeding rates (Glasby and Kingsford, 1994), which suggests movement of anchor gear over the substratum mobilizes small benthic organisms into the water column, thereby increasing the foraging potential for zooplanktivores. Secondly, the provision of emergent structures extending into the water column associated with the presence of large vessels (Røstad et al., 2006) and their anchor chains may

provide protective shelter for this group accounting for their increased abundance (Champion et al., 2015).

Although zooplanktivores were more abundant on 'anchored' reefs, strong evidence shows that the presence of vessels and anchoring processes negatively affects certain taxa, with distinct responses to disturbed reefs. Mechanical disturbances from anchoring and benthic fishing trawls consistently show evidence of removal of complex biogenic habitats on reefs (Forrester et al.,
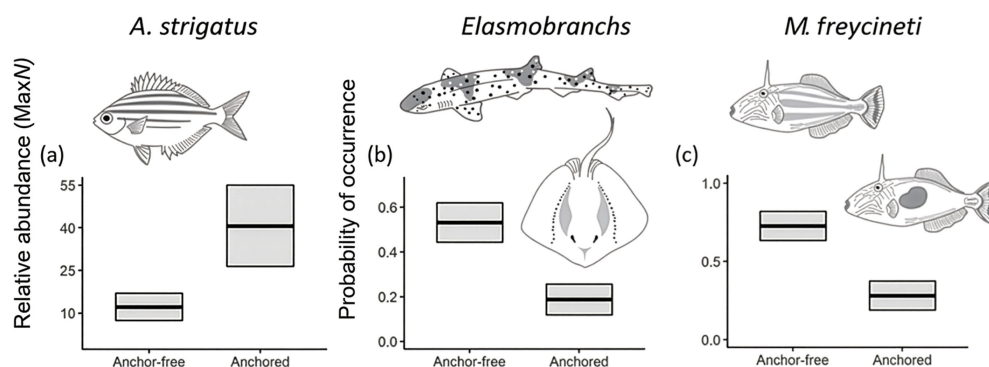

FIGURE 4

Relationships of the most parsimonious model found to predict the *relative abundances (MaxN) of the species of interest; or °probability of occurrence of two taxa of interest **(A)** *Atypichthys strigatus*; **(B)** Elasmobranchs; **(C)** *Meuschenia freycineti*; from full subset Generalized Additive Mixed Models (GAMM). Solid lines represent fitted GAMM predictions and shaded areas define standard errors ~25 and 75 quartiles around the predictions. The summary of each model is provided in Table 1.
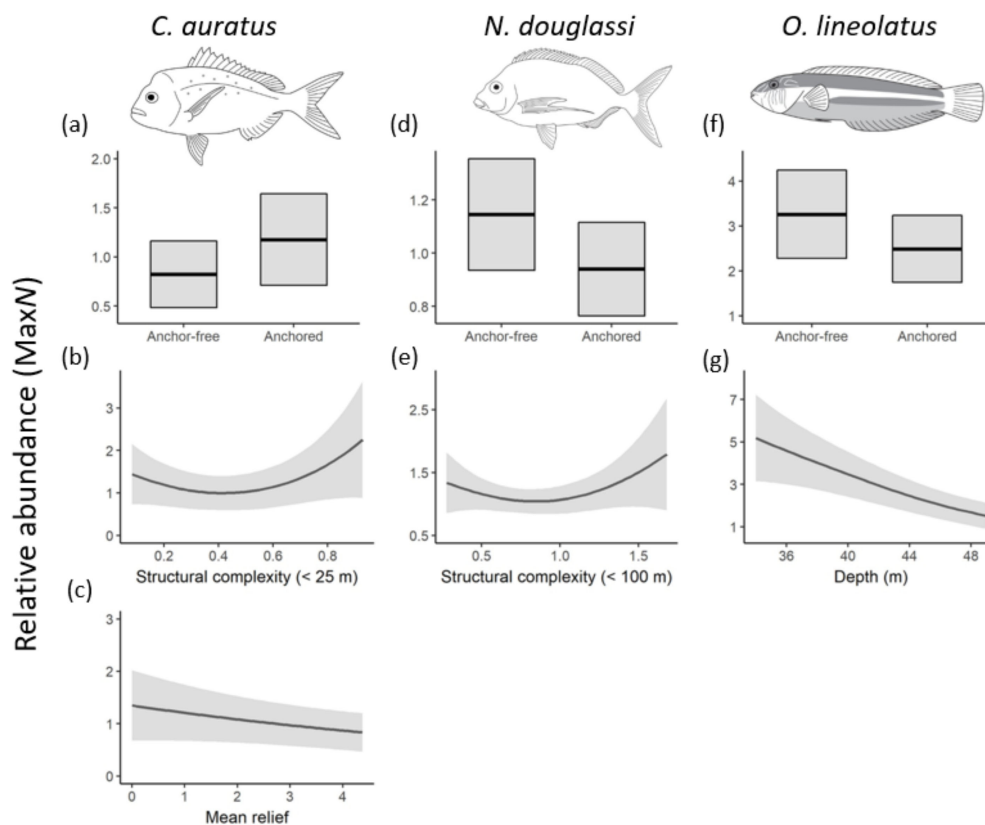
FIGURE 5
Relationships of the most parsimonious model found to predict the relative abundances (MaxN) of the species of interest; **(A-C)** *Chyrosophyrs auratus*; **(D, E)**; *Nemadactylus douglasii*; **(F, G)** *Ophthalmolepis lineolatus*; from full subset Generalized Additive Mixed Models (GAMMs). Solid lines represent fitted GAMM predictions and shaded areas define standard errors ~25 and 75 quartiles around the predictions. The summary of each model is provided in Table 1.

2015; Aburto-Oropeza et al., 2015; Flynn and Forrester, 2019; Broad et al., 2023) with negative implications for macrofauna on reefs (eg. the 'sponge-loop' - see Bart et al., 2021) as well as the demersal reef fish associated with them (Kenchington et al., 2013). In our study, several demersal reef fish were more likely to occur on 'anchor-free' reefs that are known to support up-to seven times higher densities of Marine Animal Forest (MAF) taxa when compared to reefs disturbed by anchoring (Broad et al., 2023). We observed distinct 3-fold reductions in the likelihood of observing demersal elasmobranchs and the six-spine leatherjacket *Meuschenia freycineti* on 'anchored' reefs that are largely devoid of epifauna (Broad et al., 2023). Similar, yet weaker trends, were also observed for the maori wrasse, *O. lineolatus* and grey morwong *Nemadactylus douglasii* – unsurprising, given that all of these fishes are closely tied with MAF taxa such as sponges, hydroids and bryozoans (Curley et al., 2013; Henderson et al., 2020).

Significant reductions of MAF taxa can reduce animal 'fitness' for fish that forage within these systems, expending greater energy searching for food in a depleted 'foodscape' as well as avoiding perceived threats where protective habitat has been lost (Roberts et al., 2016; Chapuis et al., 2019; Dwinnell et al., 2019). As an example, benthic elasmobranchs are often closely associated with Marine Animal Forest taxa on reefal habitats throughout all their

life stages (Kenchington et al., 2013; O'Neill et al., 2024). Reductions in sessile biota will likely affect this groups capacity to complete their life cycle, with decreases in habitat refugia or food availability. In field experiments, small benthic sharks have been shown to utilize sessile biota for position-holding to sustain alignment with water currents (rheotaxis; Peach, 2002). In more novel findings, high abundances of juvenile catsharks have been reported inhabiting the internal filtration structures of large sponges (O'Neill et al., 2024). Moreover, numerous elasmobranchs secure their egg-casings on reefal habitats, with research indicating their preference for egg deposition on sessile biota over other substrata (Ellis and Shackley, 1997; Carraro and Gladstone, 2006; Vasquez et al., 2018). In addition, many of the elasmobranchs in this study are known or are predicted to feed on the egg casings deposited on sessile biota (Powter and Gladstone, 2008) further reducing the availability of food resources on denuded reefs.

To date, anchoring disturbances from global shipping remain largely unmanaged, which has consistently shown evidence of alterations to the structure of marine assemblages across a range of habitats (Davis, 1977; Backhurst and Cole, 2000; Rogers and Garrison, 2001; Forrester et al., 2015; Lanham et al., 2018; Broad et al., 2023; Davis et al., 2025). We encourage marine managers and decision makers across all levels of government and private industry
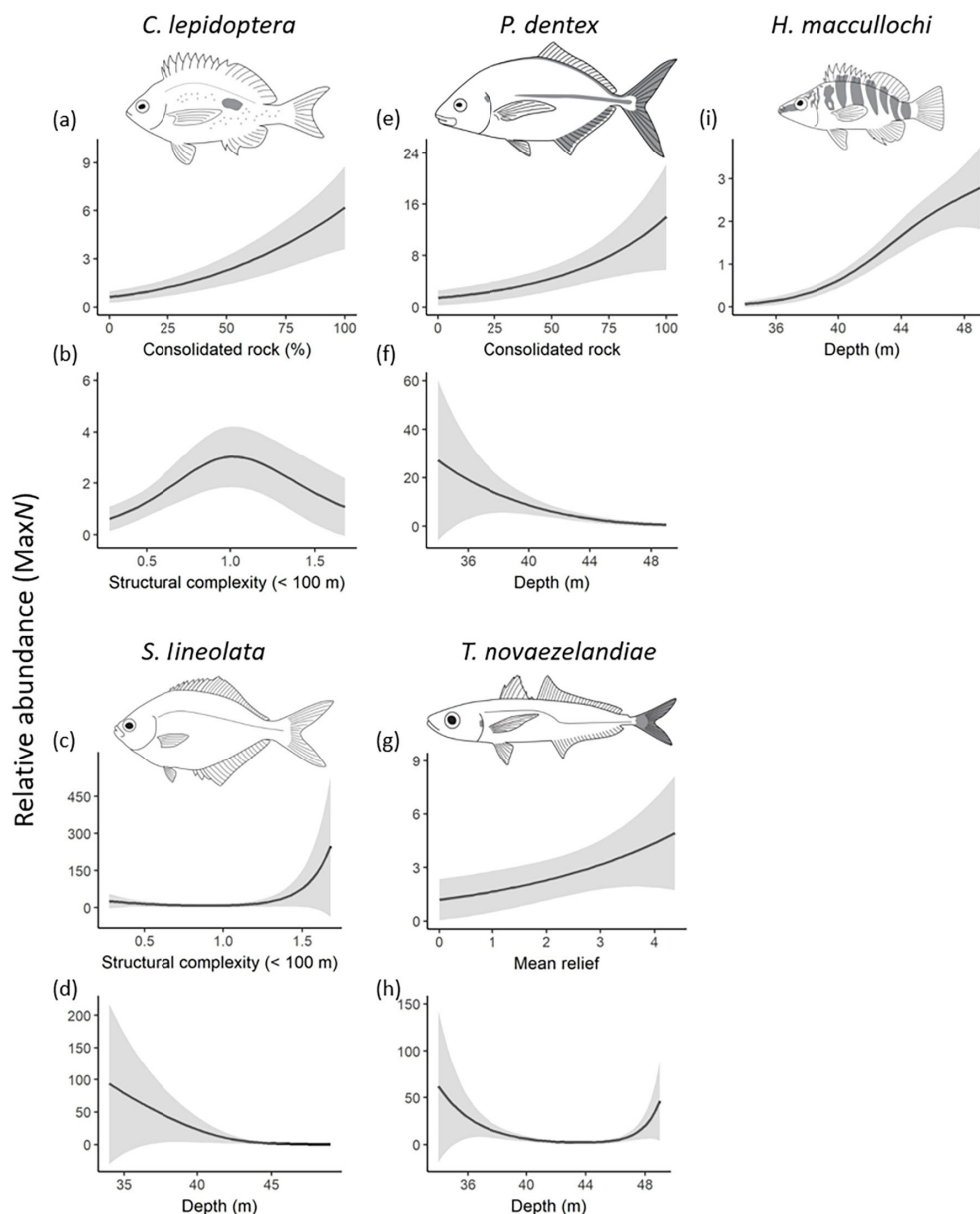
FIGURE 6
Relationships of the most parsimonious model found to predict the relative abundances (MaxN) of the species of interest. Neither anchor treatment were useful to predict the distribution of these taxa. **(A, B)** *Caesioperca lepidoptera*; **(C, D)** *Scorpis lineolata*; **(E, F)** *Pseudocaranx dentex* **(G, H)** *Trachurus novaezelandiae*; and **(I)** *Mecaenichthys immaculatus* from full subset Generalized Additive Mixed Models (GAMM). Solid lines represent fitted GAMM predictions and shaded areas define standard errors around the predictions.

to focus on reducing the need to anchor wherever possible and when necessary, aim to reduce the spatial area disturbed by anchoring (Davis et al., 2016; Steele et al., 2017). Immediate steps that can be considered to reduce anchoring impacts on marine systems include:

1. Raise Awareness: Promote understanding of anchoring-related disturbances among maritime industries and regulators, as this longstanding practice has traditionally been overlooked.

2. Enforce Designated Anchorages: Ensure vessels anchor within designated zones, with active monitoring and communication from port authorities to improve compliance (Steele et al., 2017).

3. Plan for Emergency Anchorages: Designate anchorage zones for use during exceptional circumstances (Davis et al., 2022).

4. Implement Vessel Arrival Systems (VAS): Encourage single-commodity ports to coordinate ship arrivals, minimizing unnecessary anchoring (Heaver, 2021).

5. Promote Dynamic Positioning: In high conservation areas, advocate for dynamic positioning systems over short-term anchoring (Mulrennan et al., 2025).

6. Use Seabed Mapping for Environmental Risk Assessment: Identify sensitive habitats and their biota to inform the designation of well-planned, new anchorages that avoid 'at-risk' environments.

Finally, detailed knowledge of local seabed habitats of high conservation value should be identified and avoided at all costs, and this requires regulatory support for marine managers through well-enforced policies (Jimenez et al., 2025). It is pertinent to balance environmental protection with social and economic needs, however this can only be achieved through collaboration amongst stakeholders—including governments, international agencies, and the shipping industry.

5 Conclusions

Anchoring associated with global trade is disturbing seabed environments over large areas (>1000's m^2) (Davis et al., 2022; Watson et al., 2022), yet quantitative studies examining impacts to biota have been largely overlooked (Broad et al., 2020) and are in their infancy (Broad et al., 2023). Here, we present the first, replicated empirical examination of the abundance of demersal reef fishes in response to anchoring activities from commercial trading ships (>100m). We provide evidence that anchoring activities are likely to have population-level effects on demersal reef fish across taxonomic (species and subclass levels) and trophic guilds on temperate reefs. These findings indicate potential changes in ecological function within the anchor roadstead. Given that shipping is the pillar of global trade, we encourage future research to investigate a range of anchoring stressors (eg. noise generated by ships at anchor) and their impacts to a range of biota and seabed habitats. In addition, research should seek to characterize the mechanisms that drive these changes, as failure to manage the impacts of anchoring near ports is likely to result in reductions of seabed biodiversity (Broad et al., 2023) as well as compromise highly valuable fisheries resources and associated ecosystem services (Gaylard et al., 2020). Maintaining marine biota remains one of the crucial focal areas in the environmental management of the planet as a whole (Smith, 2000; McCauley et al., 2015; Intergovernmental Oceanographic Commission [IOC], 2020); this requires strong regulatory support across all levels of government if we are to sustain marine life (UN [United Nations], 2015 [SDG14: Life below water]) and the economic benefits derived from them (Bennett et al., 2016; UN [United Nations], 2015 [SDG 9: Industries, innovation and infrastructure]).

Data availability statement

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

Ethics statement

The animal study was approved by University of Wollongong Animal Ethics (Permit AE12/07r15). The study was conducted in accordance with the local legislation and institutional requirements.

Author contributions

AB: Conceptualization, Methodology, Investigation, Formal analysis, Validation, Data curation, Visualization, Resources, Funding acquisition, Project administration, Writing – original draft, Writing – review & editing. MR: Conceptualization, Methodology, Investigation, Formal analysis, Validation, Visualization, Resources, Funding acquisition, Supervision, Project administration, Writing – review & editing. TI: Conceptualization, Methodology, Investigation, Validation, Visualization, Resources, Funding acquisition, Supervision, Project administration, Writing – review & editing. BM: Investigation, Formal analysis, Data curation, Visualization, Resources, Writing – review & editing. AD: Conceptualization, Investigation, Validation, Resources, Funding acquisition, Supervision, Project administration, Writing – review & editing.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fcosc.2025.1650920/full#supplementary-material

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