Fisheries New Zealand

Tini a Tangaroa

Spatially explicit benthic impact assessments for bottom trawling in New Zealand

New Zealand Aquatic Environment and Biodiversity Report No. 329

A.A. Rowden, O.F. Anderson, P. Neubauer, J. Hamill,
D.A. Bowden, L. Tremblay-Boyer, A. Charsley, D. MacGibbon

ISSN 1179-6480 (online)
ISBN 978-1-991285-28-7 (online)

April 2024

Te Kāwanatanga o Aotearoa
New Zealand Government

Disclaimer

Requests for further copies should be directed to:

Fisheries Science Editor
Fisheries New Zealand
Ministry for Primary Industries
PO Box 2526
Wellington 6140
NEW ZEALAND

Email: Fisheries-Science.Editor@mpi.govt.nz
Telephone: 0800 00 83 33

This publication is also available on the Ministry for Primary Industries websites at:
http://www.mpi.govt.nz/news-and-resources/publications
http://fs.fish.govt.nz go to Document library/Research reports

Please cite this report as:

Rowden, A.A.; Anderson, O.F.; Neubauer, P.; Hamill, J.; Bowden, D.A.; Tremblay-Boyer, L.; Charsley, A.; MacGibbon, D. (2024). Spatially explicit benthic impact assessments for bottom trawling in New Zealand. *New Zealand Aquatic Environment and Biodiversity Report No. 329.* 118 p.

TABLE OF CONTENTS

Plain language summary

The trawl footprint describes how much seabed area has been contacted by trawling gear in New Zealand's territorial sea (TS) and exclusive economic zone (EEZ), but it does not provide a measure of the effect of fishing on seabed communities.

This project used the trawl footprint information, in addition to other sources of information on impacts of contact by trawl gear on seabed fauna, to quantify the potential impacts to seabed communities and habitats.

Fishing gear types were first described and categorised, and footprints for each category of gear were produced. Two published impact assessment methods were applied to the TS and EEZ. The methods had different strengths and weaknesses and the outputs of the two methods were found to be complementary to one another.

The first method applied, the MRSP approach, combines information on gear categories, expert opinion on the vulnerability of seabed fauna to trawl gear, and the bottom contact footprint of trawl fishing. This approach does not consider how the fauna recover over time.

The second method, the relative benthic status (RBS) approach, uses information on the proportion of the seabed area swept by trawls and published information for depletion and recovery rates for seabed fauna considered to be particularly vulnerable to trawling. This method predicts a future state for the seabed fauna assuming no change to fishing effort.

This project provides outputs for both methods that can be used in conjunction with distribution data for seabed fauna to assess impacts of trawling and inform spatial planning processes.

Recognising the shortcomings of the MRSP and RBS approaches, two further approaches were explored and developed using data from the Chatham Rise. One approach aimed to enhance the RBS method by making this more relevant to local seabed fauna by using bycatch data from the Chatham Rise instead of relying on information from international sources. The results were encouraging but indicated that further method development is required.

The second approach expanded a previously applied spatio-temporal modelling approach to assess impacts to fauna thought to be useful indicators of potential trawling effects. It was found that this approach, as with the others, was limited by the available data, and further development is required to improve the utility of this approach in the future.

EXECUTIVE SUMMARY

Rowden, A.A.[1]; Anderson, O.F.[1]; Neubauer, P.[2]; Hamill, J.[1]; Bowden, D.A.[1]; Tremblay-Boyer, L.[2]; Charsley, A.[1]; MacGibbon, D.[1] (2024). Spatially explicit benthic impact assessments for bottom trawling in New Zealand.

***New Zealand Aquatic Environment and Biodiversity Report No. 329.* 118 p.**

The spatial extent of bottom contact by mobile fishing gear by New Zealand's inshore and deepwater fisheries has been mapped in detail. However, understanding the effect of bottom fishing on benthic communities over this area is far more complex and is dependent on the actual impact of fishing to a given taxon or community. The aim of this project was to apply, assess, and further develop methods to quantify impacts of fishing on benthic taxa or communities within New Zealand's Territorial Sea (TS) and Exclusive Economic Zone (EEZ).

A variety of methods for spatially explicit benthic impact assessment have been developed, and implemented, to quantify the likely bottom impacts of fishing. These methods combine information about parameters including: historical fishing effort; gear type; gear deployment; substratum/habitat type; vulnerability of taxa or communities; and the distributions of benthic taxa, communities, or ecosystems. In data-poor situations, particularly in the absence of reliable data or models for the distribution of benthic taxa or communities, proxy data have been used in these impact assessments.

This project used two published methods to assess benthic impact at the scale of the TS and EEZ and explored the development and application of two additional methods at the scale of the Chatham Rise. Before these methods could be applied, the types of mobile fishing gear used in the New Zealand inshore (since 2007/8) and deepwater (since 1989/90) fisheries were characterised and categorised, and the spatial and temporal extent of bottom contact was determined for the different fishing gear configurations.

The first benthic impact assessment method (here referred to as the MSRP method using the first initials of its authors) combines information on gear categories, expert opinion on the vulnerability of three benthic functional groups to trawl gear, and the bottom contact footprint of trawl fishing. The second method, relative benthic status (RBS), relied on the determination of the swept area ratio of trawling in the region; RBS was calculated for different vulnerable marine ecosystem (VME) indicator taxa. These methods, as applied here, do not consider spatial distributions of any specific benthic taxa, but such distributional layers can be incorporated in a subsequent step—as long as the taxon fits within one of the defined functional groups/VME indicator taxa used by these methods. The availability of the distributional layers, along with estimates of vulnerability parameters, for other benthic taxa or communities is critical for the wider application of this approach.

The third method used in the present study was a modification of the RBS approach which used trawl and benthic bycatch data from the Chatham Rise in simulations to explore the underlying assumptions of RBS, as well as statistical models to directly estimate depletion and recovery parameters for four benthic taxa from these local data. The results of this work were encouraging, but further development is required before this enhanced approach can be implemented more widely and with confidence. The fourth method updated a previous application of a spatial-temporal modelling approach which assessed the impact to a soft-sediment habitat indicator taxon on the Chatham Rise and extended its application to a hard substratum indicator taxon. This approach is also reliant on estimates of benthic taxa depletion, and further development of this approach is required to improve its utility.

[1] National Institute of Water and Atmospheric Research (NIWA), New Zealand.
[2] Dragonfly Data Science, New Zealand.

While further development of the second two methods is required, outputs from the application of the complementary MSRP and RBS methods can now be used to inform marine spatial planning processes in New Zealand and can support inshore and deepwater Fisheries Management in meeting objectives in their respective management plans.

1. INTRODUCTION

A wide variety of species are harvested on or near the seabed in New Zealand's Territorial Sea (TS) and Exclusive Economic Zone (EEZ). Mobile bottom fishing methods that target these species can significantly impact benthic communities (see global reviews by Jennings & Kaiser 1998, Clark et al. 2016a for inshore and offshore fishing impacts, respectively). Numerous studies have been conducted in New Zealand to understand the effects of fishing on different benthic habitats (e.g., Thrush et al. 1995, Cryer et al. 2002, Clark & Rowden 2009).

Our understanding of the impacts to benthic habitats at the scale of the EEZ needs to be developed (Thrush et al. 1998). The spatial extent of bottom contact by mobile fishing gear by New Zealand's inshore and deepwater fisheries has been mapped in detail (Baird et al. 2015, Baird & Wood 2018, Baird & Mules 2021). However, understanding the effect of bottom fishing on benthic communities over this area is far more complex and is dependent on knowledge of both the spatial distributions of taxa or communities and the actual impact of fishing to a given taxon or community.

Risk assessment approaches are being used to evaluate the effects of fishing on various components of the marine ecosystem in New Zealand and contribute to spatial planning processes (Clark et al. 2014, Ford et al. 2015, Richard & Abraham 2015). A necessary step in the risk assessment process is the assessment of impacts, and ideally such assessments should be quantitative (Hobday et al. 2011).

The aim of this project is to apply, assess, and further develop methods to quantify impacts of fishing on benthic taxa or communities within New Zealand's TS and EEZ. The outputs of the project will inform marine spatial planning processes in New Zealand, contribute to the development and application of benthic risk assessment approaches, and will support inshore and deepwater fisheries management to meet objectives of Fisheries New Zealand's respective management plans.

A variety of methods for spatially explicit benthic impact assessment have been developed, and implemented, to quantify the likely bottom impacts of fishing (e.g., Rijnsdorp et al. 2016, Eigaard et al. 2017). These methods combine information about parameters including: historical fishing effort; gear type; gear deployment; substratum/habitat type; vulnerability/sensitivity/mortality of taxa or communities; and the distributions of benthic taxa, communities, or ecosystems. In data-poor situations, particularly in the absence of reliable data or models for the distribution of benthic taxa or communities, proxy data have been used in these impact assessments.

In 2015 a group of 'experts' met in New Zealand to address the question "What is the best scientific approach to assessing trawl and dredge impacts on benthic fauna and habitats in New Zealand in the short, medium and long-term?". This MPI-initiated and -sponsored 'expert' workshop, concluded that population-based modelling approaches (termed "fishing impact/productivity approaches" in the report), were a "useful starting point" for benthic risk assessment of fishing in New Zealand waters (Ford et al. 2016). However, data to support these sorts of assessments are not always available and the workshop recommended that less data-intense methods should be considered in the interim.

Sharp et al. (2009) developed a method to assess the impact of longline fishing gear in the Convention of the Conservation of Marine Living Resources (CCAMLR) area. This simple method considers gear types, gear deployment, vulnerability of taxa, historical fishing effort, and, although it does not directly incorporate knowledge about the distribution of benthic taxa or communities (making it useful for application in data-poor cases), it can be overlaid on/incorporated with distributional maps/data for relevant taxa. The utility of this method has been demonstrated for areas in the South Pacific Regional Fisheries Management Organisation (SPRFMO) Convention Area, where benthic impacts of bottom trawling have been estimated (Mormede et al. 2017). This method, here referred to as the MSRP method using the first initials of its authors (Mormede/Sharp/Roux/Parker), is similar to those used by Rijnsdorp et al. (2016) and Eigaard et al. (2017) in European waters.

Pitcher et al. (2017) developed a quantitative method for assessing the risks to benthic habitats by towed bottom-fishing gears. This method is based on a simple equation for relative benthic status (RBS), derived by solving the logistic population growth equation for equilibrium state. Estimating RBS relies only on maps of fishing intensity and habitat type, and parameters for impact and recovery rates for benthic fauna, which are taken from meta-analyses of multiple experimental studies of towed-gear impacts for different gear/habitat types. The RBS method has been used to assess the impact of bottom trawl fishing on the world's continental shelves and deepwater, including those of New Zealand (Amoroso et al. 2018, Pitcher et al. 2022). Such quantitative risk methods can be further improved (and made relevant for local benthic fauna) by directly estimating vulnerability parameters (and, potentially, recovery rates, depending on the time and spatial scales of observations) from observer data (Zhou et al. 2014, Neubauer et al. 2019), as well as by incorporating taxon-specific population dynamics parameters into the models.

Mormede et al. (2021) developed an approach to benthic risk assessment based on spatial population models (SPM) and vector-autoregressive spatio-temporal (VAST) models and using modeled data for the distribution of benthic indicator species. Assessing fishing impacts on indicator species may serve to evaluate the fundamental condition of the environment without having to capture the full complexity of the system. Benthic indicator taxa were defined by Mormede et al. (2021) as those taxa that are most responsive (negatively) to habitat disturbance, so that changes in their individual abundance and distribution, and estimates of impact and risk due to fishing, will reflect changes at the scale of the habitat. The approach of Mormede et al. (2021), which they demonstrated using only one candidate indicator taxon on Chatham Rise, was built on previous work carried out in New Zealand (Mormede & Dunn 2013).

For this project, we conduct spatially explicit benthic impact assessments for the New Zealand TS and EEZ using the MSRP and RBS methods of Mormede et al. (2017) and Pitcher et al. (2017), as well as use data from Chatham Rise to develop and evaluate a modified version of RBS and to update and progress the initial work of Mormede et al. (2021). This multi-method approach provides for a comparative assessment of the practicality and effectiveness of each method to estimate the impact of fishing given current data limitations. This evaluation of multiple methods follows the conclusion of Ford et al. (2016), that given the uncertainty associated with benthic impact assessments, "Using a number of fundamentally different modelling approaches [is] favoured to understand the true uncertainty, and the influence of alternative modelling assumptions on the resulting estimates of relative risk".

The first stage of this project requires the characterisation of the bottom gears used in New Zealand fisheries, because their design and dimensions will influence the nature and extent of bottom contact and thereby the severity of the impact on the seabed and its fauna (Eigaard et al. 2016). Therefore, the key parameters required from such a characterisation are those that define the width of the gear being towed and the physical characteristics of the gear actually in contact with the seafloor. Mean or representative values for these parameters are sought for application to broad categories of fishing defined by the parameters recorded in the fisheries catch and effort data collection system, including (among others) vessel length, nationality, fishing method, wingspread, headline height, and target species. Here we take an existing categorisation based on such parameters (that of Baird & Mules 2021), then refine and expand it based on a review of mobile bottom fishing methods used in New Zealand fisheries.

In New Zealand waters, there are two main mobile fishing gear groups that can impact benthic fauna and habitats: trawls and dredges. Trawls are used to target the main commercial mobile species such as hoki, orange roughy, squid, jack mackerel, scampi, and snapper. Dredges are used principally to target benthic bivalve species such as scallops, oysters, and clams. The rigging and components of these two main gear types are highly variable with configurations determined by vessel size, the specific target species, and type of seabed habitat. However, common components of the kinds of trawl gear used in New Zealand include a pair of otter boards (trawl doors), sweeps, bridles, and one or more trawl nets (Figure 1a), while the components of a dredge usually consist of a rigid rectangular frame with a cutting bar (with or without teeth) which dislodges shellfish into a collecting basket as the dredge is towed over the seafloor (Figure 1b).

Some general descriptions of trawl gear used in New Zealand are given by Baird et al. (2002, 2011) and Clement & Associates Limited (2008), as well as by Eayrs et al. (2020) in a detailed report on mitigation techniques to reduce benthic impacts of trawling. An overview of shellfish dredge gear types and gear size regulations can be found in Beentjes & Baird (2004) and Michael (2009). However, although details of specific gear types are sometimes recorded in voyage reports or papers (e.g., Clark et al. 2016b), typically descriptions and technical specifications for the range of mobile bottom gear used is largely unavailable, partly because it is considered the intellectual property of fishers. Thus, we provide in this report, for context, descriptions of typical trawl and dredge gear types and configurations used across these New Zealand fisheries.

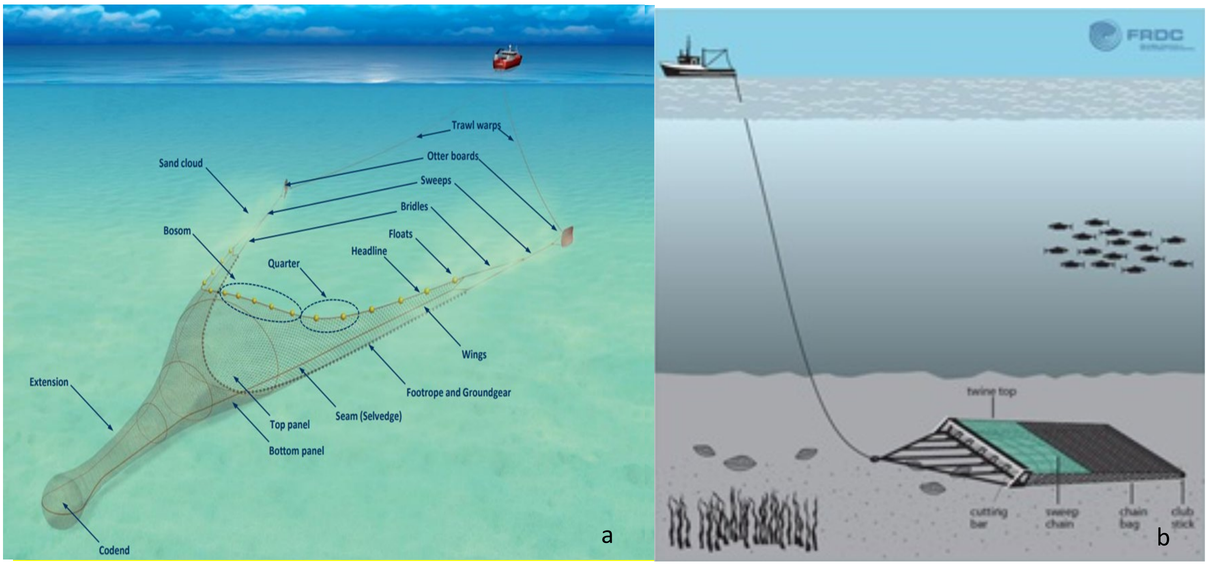


Figure 1: (a) Basic components of bottom trawl gear (image source: with permission from Seafish), and (b) a shellfish dredge (image source: modified from Fisheries Research and Development Corporation, Australia).

Overall Objective:

The aim of this Fisheries New Zealand project (BEN2019-04B) is to apply, assess, and further develop methods to quantify impacts of fishing on benthic taxa or communities within New Zealand's TS and EEZ.

Specific Objectives:

1. Characterise all mobile bottom fishing gear configurations used since 2007/8 for inshore fisheries, and since 1989/90 for deepwater fisheries.

2. Determine the spatial and temporal extent of bottom contact by different fishing gear configurations.

3. Characterise the impacts of different gear configurations on key benthic taxa and/or communities.

2. METHODS

2.1 Characterisation and categorisation of mobile bottom fishing gear configurations

To characterise mobile bottom gear for deepwater vessels since October 1989 and for inshore fisheries since October 2007, data were extracted from Fisheries New Zealand Central Observer Database (*cod*). Data were not available in *cod* for Danish seine or shellfish dredge fisheries. Data for observed historical fishing effort, prior to these dates, are either not available in any useful quantity or their spatial precision is low and therefore they are not considered in any of the analyses.

All trawl related data for the time period were extracted and underwent basic grooming procedures to remove or correct obvious errors. These data consisted of two datasets.

The first was a 'trip level' dataset. This dataset listed the bottom-contacting trawl gear components for each unique gear configuration used on a given vessel's trip. There could be multiple configurations for a given vessel on a given trip. These unique configurations are identified in *cod* with the variable gear_equipment_code (gear code). Typical gear components listed under a single gear code could be for example: rubber discs/cookies, bobbins, rockhopper discs, chain, rubber spacers, etc.

For each unique combination of vessel-trip-gear code, we determined a single ground gear 'type' based on the component(s) listed for that vessel-trip-gear code combination. For example, if the given components of a particular gear code were rubber spacers, bobbins, and chain backbone, then this gear code would be defined as a 'bobbin rig'. Five broad categories of ground gear were determined based on the type of components typically found together and assigned to each gear code for each vessel and trip:

1. Bobbin rig
2. Rubber discs/cookies
3. Rockhopper
4. Chain
5. Wire

The second dataset that was extracted from *cod* was for all trawl trips at the level of the individual fishing-event, i.e., a single trawl tow. These individual fishing events are recorded in *cod* with the variables vessel, trip, and gear code. Therefore, a gear type (as listed above) can be linked with each fishing event using the combination of the variables vessel, trip, and gear code.

The fishing-event level data also contained information on vessel size, flag nationality, and target species. Various vessel type and target species groups were then investigated to see which ground gear types were used in those groups in line with previous work (Baird et al. 2015) and based on regulations and knowledge of fishing gear types used in New Zealand fisheries.

The following categories were used:

Category A: Domestic vessel under 28 m, largely inshore trawlers, and those smaller vessels >20 m that may use smaller midwater trawls at times.

Category B: Vessels 28–46 m represent New Zealand domestic vessels; this fleet is highly diverse with respect to gear set-up.

Category A and B vessels targeting scampi. These vessels require a specific bottom trawl gear set-up. There are around ten vessels that target scampi year-round.

Category C: 46–82 m domestic freezer and foreign vessels. These vessels use a mixture of midwater and bottom trawl gear.

Category D: Big Autonomous Trawler Reefer vessels 104 m in length. These are large Soviet-era fishing vessels known by their Russian acronym—BATM. These are the largest fishing vessels operating in the deepwater fleet and generally use midwater trawl gear.

Category E: Domestic vessels targeting orange roughy and oreos (ORH/OEO) with bottom trawls.

The above categories were further separated by fishing method, target species, habitat type, and where applicable, door spread averages. Habitat was defined, for orange roughy and oreo fishing only, as either UTF (underwater topographic feature) or 'slope' (i.e. non-UTF) tows. Tows were considered a UTF tow if the tow duration was less than 30 minutes and the start position was within 3 nm of a known hill (elevation < 500 m), 5 nm of a knoll (elevation 500–1000 m), or 8 nm of a seamount (elevation > 1000 m), following established methods and known UTF locations (respectively, Roux et al. 2017, Rowden et al. 2008).

The most common gear type for each of the above categories was considered to be representative of the given vessel category where other gear types were either relatively rare, or clearly errors (e.g., bobbin rigs on a midwater trawl). Note that there is an implicit assumption in this categorisation, and in the characterisation in general, that the observed vessels and their gear are representative of the fleet. Observer coverage has been variable over time and among target fisheries but has generally been at least 10% or greater in each year for most of the deepwater fisheries (see e.g., Finucci et al. 2022, Anderson & Finucci 2022, Anderson et al. 2023).

Published gear plans and literature were used, as well as interviews with vessel skippers, vessel managers, fishers, net makers, and fishery stakeholders (see Acknowledgements) to confirm the gear specifications and deployment characteristics.

These sources of information were combined to first provide basic descriptions of typical trawl and dredge gear types and configurations used across New Zealand fisheries, because such descriptions are not available all together anywhere else and provide useful context for later objectives of the overall project.

2.2 Spatial and temporal extent of bottom contact by different fishing gear configurations

2.2.1 Commercial trawl data

Detailed and comprehensively error-checked trawl catch-effort data compiled under Fisheries New Zealand project BEN2019-01 (mapping of the trawl footprint) were used as the basis for building a dataset of trawl polygons representing the bottom contact from all inshore bottom trawl fisheries from 2007–08 to 2018–19 and all deepwater fisheries from 1989–90 to 2018–19. Trawl start positions were available for all trawls, but it was necessary for the authors of that work to generate finish positions for a proportion of the data using sequential fishing locations, and randomly jitter start and finish positions for records where reported latitude and longitude data were truncated to the nearest minute of arc. Total trawl widths (door spread) were provided for a range of fishery/gear categories in this dataset including all midwater trawls towed within 1 m of the seafloor. Door spread is not recorded on commercial data forms and so was estimated based on vessel size, target species, and known gear parameters (see Baird & Mules 2021 for full details of data processing and error-checking procedures).

To account for the offset between the position of the trawl gear on the bottom and the vessel location (from which the start and finish positions are recorded), all start and finish positions were adjusted based on the direction of travel, by a value calculated from $Depth \times \sqrt{3}$, assuming a trawl warp length of two times the depth. Values for missing depths were estimated using a 1 km resolution bathymetry raster for the region. Note that this procedure assumes the offset is in line with the direction of travel of the vessel and does not account for any lateral deviation of the gear due to currents or other factors acting on the gear in the water.

Data for dredge fisheries (mainly those for scallops, oysters, and clams) were not included because of the lack of historical fine-scale location data.

2.2.2 Total trawl widths

A total trawl width was assigned to each tow according to the gear categorisation (see Section 2.1). The categories are an extension of those defined by Baird & Mules 2021 which associate a total door spread value to a range of gear categories based on analysis of available data and as agreed on by the Fisheries New Zealand Aquatic Environment Working Group (AEWG) (Appendix 1). The extensions to these categories were made to differentiate orange roughy/oreo fishing (on and off UTFs) from other types of fishing, based on parameters defined by Mormede et al. (2017) for these fisheries, and to separate midwater trawling near the bottom (within 1 m, as defined by Baird & Mules 2021) from bottom trawling, resulting in a lower effective trawl width.

2.2.3 Effective trawl widths

The effective trawl width is defined here as the width of the trawl in actual contact with the seafloor, a value less than the total door spread due to some components of the trawl (doors, sweeps, and bridles) sometimes being above the seafloor. Effective trawl widths were calculated for bottom trawls from the outputs of an expert gear workshop in 2017 (as reported by Mormede et al. 2017) and from values derived from a Delphi method (i.e., fully independent) survey of NIWA staff with a knowledge of trawl gear operation (Appendix 2). Participants were asked to estimate the percent bottom contact time for different parts of trawls (including midwater trawls fished within 1 m of the seafloor), including the trawl doors/wing-end weights, sweeps/bridles, and ground gear. Responses were compiled and average values used to adjust the door spread values assigned under BEN2019–01 to estimate effective trawl widths. A drawback of this method is that it relies on expert opinion alone, with no empirical data to support estimates, and results are likely to differ depending on which experts are included.

For example, for non-UTF orange roughy trawls, doors, and ground gear are expected to be in contact with the seafloor at all times during a tow but sweeps/bridles are estimated to be in contact only 80% of the time (from Mormede et al. 2017). Based on ratios for a typical trawl with component widths as follows: doors, 2 m; sweeps/bridles 113.5 m; ground gear 18.5 m (from Mormede et al. 2017), a fraction of 0.831 can be calculated to apply to a given door spread, assuming an approximately constant ratio of component widths, to estimate an effective trawl width for each category. Note that although Mormede et al. (2017) reported a value of 3 m for the combined width of two typical door furrows, this was amended to 2 m after initial presentations of this analysis to the AEWG, and subsequent literature research (e.g., Ivanovic et al. 2011, Eayrs et al. 2020).

2.2.4 Determination of the spatial and temporal extent of bottom contact by different fishing gear configurations

Procedures from Mormede et al. (2017) were again followed to produce representations of the degree, spatial extent, and temporal variability of bottom contact by each of the 24 categories of fishing gear, both separately and combined.

Firstly, to better allocate effort into grid cells from tows that spanned more than a single 1×1 km cell, each tow polygon (a straight line defined by the start/finish position and the effective trawl width) in the revised footprint dataset was split into 150 m segments, then each segment assigned to a cell of a 1×1 km grid covering the extent of the New Zealand TS/EEZ, based on the midpoint of the segment. This is a larger segment length than applied by Mormede et al. (2017) (100 m) but was necessary due to the larger dataset in this case (about 150 M records after splitting cells into segments) leading to processing power limitations.

The area of these segments was then used to produce spatial grids of bottom contact in two ways: firstly, by calculating the cumulative proportional footprint in each cell (whereby each successive trawl overlaps with previous trawls in the same cell in the simplified (i.e., not using spatial GIS operations) random manner described by Mormede et al. (2017) to represent the proportion of the cell contacted (values range from 0–1); and secondly, by summing the area of the segments in each cell to represent the total area contacted, regardless of any overlapping of segments. The first method follows that used by Mormede et al. (2017) (Table 1) for subsequent calculation impact layers by faunal group, and the second method produces output that is equivalent to the Swept Area Ratio (SAR) applied in the method for calculating taxon-specific impact layers (Pitcher et al. 2017). These calculations require the area of each "1×1 km" cell as input, and this differs by latitude due to the use of a Mercator projection centred near the mid-latitude of the study area (41° S). For the occupied cells of the footprint the mean cell area is 0.92 km^2, with an interquartile range of 0.86–0.98 km^2 (a similar range to that shown by Mormede et al. 2017 for fishing effort outside the EEZ). The mean cell area (0.92) was used in all calculations. Use of an equal area projection (where each cell then has by definition the same area) is a logical future development of this method.

Table 1: Summary of the procedure for calculating the cumulative proportional footprint and swept area ratio in each grid cell (adapted from Mormede et al. 2017).

Step	Description
1	Each 150 m tow segment was assigned to a cell.
2	Each segment length was multiplied by the effective trawl width (Appendix 1) to calculate the footprint area of each segment, then divided by the mean cell area to estimate segment contribution to proportional footprint.
3	The proportional footprint of each cell (*F*) was calculated assuming random overlap between trawl segments, whereby: $F=F1+F2-(F1*F2)$ for two segments *1* and *2*, then looped over all segments *x*: $F=F+Fx-(F*Fx)$.
5	The swept area ratio of each cell was calculated as the sum of the segment areas in each cell.

Spatial grids of bottom contact for each method and gear category were prepared as a set of maps (in both *jpg and *asc format) at a resolution of 1 km and in the Mercator 41 projection (ESPG:3994). This format is the most suitable for input into the resource management modelling (Zonation) for which these grids are intended under a related Fisheries New Zealand project (BEN2019-05: *Towards the development of a spatial decision support tool for managing the impacts of bottom fishing on in-zone, particularly vulnerable or sensitive habitats*). Grids and maps can readily be converted into the Fisheries New Zealand Albers equal area projection (EPSG:9191), as required.

Maps were initially produced to represent bottom contact only for all years combined, but maps for individual years can be produced as required. This is due to the potential number of maps that would need to be made if annual representations were required (24 categories, 29 years, 2 formats, 2 impact assessment methods = 2874), doubled when maps for impact are also produced, and the storage space this would require.

2.3 Impacts of different gear configurations on key benthic taxa

2.3.1 MSRP for TS and EEZ

The MSRP method (Mormede et al. 2017) was used to estimate the impact from bottom trawls for each of three categories of benthic fauna, for each gear category—separately and combined.

Impact values are defined by Mormede et al. (2017) as "the proportion of vulnerable benthic taxa damaged or destroyed in a single passage of a bottom trawl". Separate estimates of impact values were made for each gear category and each of three functional groups of vulnerable benthic organisms, based on expert-derived values given by Mormede et al. (2017) for UTF and slope ORH/OEO fishing. For other fishing gear categories, impact values were derived from a Delphi survey of NIWA experts, who were asked to estimate the percent impact (mortality) to the same three functional groups of benthic fauna from different ground gear types (Appendix 2). The three faunal groups were: Large, erect, hard, sessile (LEHS); Small, fragile, encrusting (SFE); Deep, burrowing infauna (DBI) (after Mormede et al. 2017).

This method is applied in a similar way to (i.e., is effectively an extension of) the calculations of the extent of bottom contact. Here the effective trawl widths are further reduced by applying to them the expert workshop/Delphi survey impact values for each combination of faunal and gear category. The total impact per cell is then calculated in the same way as described for total bottom contact (i.e., with random overlap of individual segments, see Table 1). Impact is therefore represented on a scale of 0–1, with 0 being a completely unimpacted state and 1 being completely impacted.

The MSRP method applies a faunal category-specific mortality to the adjusted trawl widths and does not allow for recovery over time; therefore it will underestimate current impact to some degree. Spatial grids of MSRP impact were prepared as sets of maps in the same format as for bottom contact estimates, i.e., 1 km resolution and Mercator 41 projection.

2.3.2 RBS for TS and EEZ

The method of Pitcher et al. (2017) was followed to generate measures of Relative Benthic Status (RBS) for benthic taxa considered to be indicators for vulnerable marine ecosystems (VMEs) in the South Pacific (Parker et al. 2009). Although the method can be broadly applied, we limited its use here to VME indicator taxa for which spatial predictions of suitable habitat in the EEZ are available (200–3000 m; Stephenson et al. 2021) and are most widely recognised as being vulnerable to the effects of fishing. RBS is a quantitative method for assessing risk to benthic habitats by mobile bottom fishing methods, using a modification of the Schaefer (1954) logistic population growth equation, with an additional term to describe the direct impacts on the benthos, to calculate the equilibrium state. RBS estimates the long-term relative abundance of biota as a fraction of its unimpacted level. For this method, fishing intensity data were utilised in the form of Swept Area Ratios (SAR), as described above, and published data on depletion and recovery rates for the different VME indicator taxa (Table 2). There are similarities between the MSRP and RBS methods, but key differences are that RBS accounts for future trawling as well as the potential for recovery, and MSRP incorporates bottom contact as a proportion of the cell area whereas RBS sums all contacted area. In both methods the calculated impacts are assigned to the cell as a unit, whereas in reality there will be variability of impact within each cell, with some locations not impacted at all.

The equation for RBS estimates the long-term relative abundance of biota (*B*) as a fraction of the carrying capacity (*K*) as follows:

$$B/K = 1 - F\, d/R \text{ (where } F<R/d\text{, otherwise } B/K=0)$$

In this equation, relative benthic status = B/K (range 0–1), R is the taxon-specific proportional recovery rate per year, d is the gear/taxon-specific depletion rate per trawl, and F is trawling intensity (SAR). F was implemented in the equation as the long-term (30-year) mean SAR in each cell. In the equation, the ratio d/R represents sensitivity to trawling, specifically the time interval (in years) between trawls that would lead to an RBS of zero, i.e., local extinction. Absolute status in each cell can be determined for a specific taxon by multiplying the calculated RBS (B/K) by its predicted abundance/habitat suitability, from species distribution models, i.e., $B=K*B/K$.

Depletion rates are dependent on the type of trawl used, as well as the habitat type (e.g., gravel, sand, mud). No published depletion rate estimates are available that can be individually associated with the range of gear categories defined in this study, or with any specific habitat type. This study therefore differs from that of Pitcher et al. (2017) in the lack of specific depletion values for separate fishing methods and habitat types. Instead depletion rate estimates were utilised for a range of taxa based on a meta-analysis of published studies of otter trawl impacts (Welsford et al. 2014, Mormede et al. 2017, Pitcher et al. 2017); values for recovery were obtained from the same set of studies. Uncertainty around these estimates was available from examinations of the original data used by Pitcher et al. (2017) and was presented by SPRFMO (2020) as low and high sensitivities around a central estimate, shown here in Table 2.

This approach therefore takes into account the relative widths of the 24 separate gear categories (as these are incorporated directly into the SAR calculations), but it is not able to account for the varying impacts on the benthos from the range of ground gear types in use (i.e., rubber disks, rockhopper, bobbins, chain, wire/chain).

Using the formula above, RBS for each taxon was calculated in each cell of the study area using a SAR constructed from the combined footprint from all 24 gear categories, and uncertainty estimated by calculating a worst case (highest *d*, lowest *R*) and best case (lowest *d*, highest *R*) for comparison with the base case (best estimates of *d* and *R*) using the values in Table 2. It would be possible to also calculate RBS separately for each gear category, using gear category specific SARs, but this was not done here.

Table 2: Trawl fishing depletion (*d*) and recovery (*R*) rates by individual taxa, with sensitivities for the uncertainties in these values (low and high) as used in the calculation of RBS. The taxon codes are those used by Fisheries New Zealand.

		Depletion			Recovery		
Taxon	Code	*d*	*d* (low)	d (high)	*R*	*R* (low)	*R* (high)
Solenosmilia variabilis	SVA	0.67	0.52	0.82	0.20	0.15	0.25
Madrepora oculata	MOC	0.67	0.52	0.82	0.20	0.15	0.25
Goniocorella dumosa	GDU	0.67	0.52	0.82	0.20	0.15	0.25
Enallopsammia rostrata	ERO	0.67	0.52	0.82	0.20	0.15	0.25
Antipatharia	COB	0.50	0.39	0.61	0.33	0.25	0.41
Other Alcyonacea (soft corals)	SOC	0.35	0.27	0.43	0.24	0.18	0.30
Gorgonian Alcyonacea	GOC	0.50	0.39	0.61	0.27	0.20	0.34
Stylasteridae	COR	0.41	0.32	0.50	0.33	0.25	0.41
Demospongiae	DEM	0.38	0.30	0.46	0.24	0.18	0.30
Hexactinellida	HEX	0.38	0.30	0.46	0.24	0.18	0.30
Pennatulacea	PTU	0.34	0.26	0.42	0.39	0.29	0.49

2.3.3 Modified RBS-type approach for Chatham Rise

2.3.3.1 Rationale

The RBS method is a straightforward approach to estimate the impacts of trawling on benthic communities in data-limited situations (Pitcher et al. 2017). It follows initial work by Ellis et al. (2014), who provided a modelling framework to scale up trawling impacts at very fine 'pixel' scales (such as those used in trawl Before-After-Control-Impact-type experiments) to management 'grid' scales (e.g., the resolution used for effort reporting), but assumes equilibrium conditions. Beyond an estimate of trawl effort (as the swept area ratio, SAR), the RBS approach relies on the availability of two key parameters for the communities of interest (recovery, *R*, and depletion, *d*). Previously, these were estimated from small-scale experiments (Pitcher et al. 2017), large-scale comparative studies scaled to larger scale impacts (Hiddink et al. 2017, Pitcher et al. 2022), or on expert-elicited parameters for species and gear combinations (e.g., SPRFMO 2020).

The majority of experimental impact studies concern particular benthic assemblages in specific geographic areas (mostly Northern Hemisphere locations; Hiddink et al. 2017). It is therefore often unclear how well these experimentally derived values transfer to particular taxa and/or location and gear combinations in New Zealand. An additional concern is that experimental recovery rates in study areas might not be representative of recovery rates in actively trawled areas if there is less biomass in areas adjacent to the trawled site to boost local recruitment. Large-scale comparative studies suffer from similar issues as they typically collate the results of small-scale studies (Hiddink et al. 2017).

Expert-elicited values for key parameters depend strongly on knowledge of the biology of key VME indicator taxa as well as their response to fishing; for many of these species, biological parameters such as reproductive output, growth rates, and dispersal potential remain highly uncertain, especially for Southern Hemisphere taxa.

In addition to assumptions about key parameters, the RBS method assumes that taxa or communities are at equilibrium with spatial fishing effort averaged over some period of time. This assumption may hold for long established trawling grounds but is unlikely to be respected in areas where trawling effort varies over time (but see SPRFMO 2020, for an alternative formulation). As such, deriving the status of benthic communities from observations resulting from the commercial trawl effort would provide an important advance in measuring bottom trawl impacts over the data-limited RBS approach.

Although data on bycatch of benthic taxa in trawl fisheries are steadily improving in terms of taxonomic and geographical resolution, existing data for most taxa are unlikely to be sufficient to provide a time series that would make traditional, integrated stock assessment methods feasible. Nevertheless, spatial assessments can make it possible to consistently estimate key demographic (i.e., productivity) and fishery (depletion levels) parameters given differential fishing pressure in space and time: while in traditional stock assessments information about productivity comes from the response of a population to changes in fishing pressure over time (i.e., contrast in the time series of relative abundance as a consequence of changing levels of catch), a similar signal can be obtained when considering the response of populations in space, even with short time series of exploitation. This premise was used to derive community recovery parameters in recent large-scale meta-analyses, albeit under the assumption of known gear depletion rates, relative depletion status, and equilibrium conditions (Hiddink et al. 2017, Pitcher et al. 2022). Whether the full surplus-production formulation underlying the RBS can be fitted from non-equilibrium (i.e., spatially and temporally variable) catch and effort remains an open question.

Spatial surplus production models were fitted to simulated and observed data for benthic taxa on the Chatham Rise, adapting the model developed by Ellis et al. (2014) for use within a Bayesian statistical framework to estimate key parameters. This approach does not require existing values for R and d as equivalent pixel-scale parameters are estimated as part of the model fitting procedure. This is advantageous for benthic communities in New Zealand, given the paucity of biological information for many taxa.

To understand the behaviour of the models, a simple simulator of spatial trawl tracks was first constructed on a very fine spatial grid, accounting for 'pixel scale' impacts (e.g., 10s to 100s metre resolution, Ellis et al. 2014), such as those used to calibrate the original RBS formulation. Pixel-scale impacts are then scaled to larger 'management grid' units for use in a spatially discrete surplus production model. The simulation model was used to highlight the potential and some shortfalls of the spatial estimation approach and characterise the reliability of model estimates under various scenarios of observation error. Lastly, this framework was illustrated using fisheries observer data for four groups of bycaught benthic taxa on the Chatham Rise, but we caution that this preliminary application should be further refined before applying it more widely. The groups were chosen based on data availability and span taxa classified as VME indicator taxa (Figures 2 and 3).

2.3.3.2 Simulation setup: benthic population and trawl simulator

To understand the potential for spatial differences in fishing effort to inform model-based estimates of RBS, a population of an arbitrary benthic taxon was simulated across an arbitrary grid of 100×100 pixels. This approach assumes that the main spatial difference between pixels is their carrying capacity K, simulated using a random field with a spatial decorrelation distance of 0.2 per unit. This results in a spatial distribution of the taxon as variable through space but clustered in specific areas of the grid, similar to what would be expected in a real population. The population growth rate r for the taxon (taken to be 0.1 annually or 0.008 monthly) was assumed to be constant in space and time and that the distribution of the focal taxon is uniform at the pixel scale (i.e., within each pixel).

A trawl simulator was set up that places random trawl tracks of random length and direction either 1) proportionally to, 2) inversely proportionally to, or 3) randomly with respect to a simulated spatial map of the bycatch taxon in question. The proportional depletion from trawl effort for gear G was defined to be $d_G = 0.1$ for the focal taxon, and constant across space and time, that is, each trawl pass over the pixel removes 10% of the existing biomass irrespective of local population size.

Simulations were set up at a monthly timescale with the dynamics of the focal taxon within each pixel assumed to follow a Schaefer surplus production (i.e., logistic) model. Analogous to the original RBS formulation, we assumed that movement and dispersal between pixels play a negligible role in local population dynamics, and that recruitment is entirely local. Therefore, the dynamics for pixel s at time t,

$$B_{t,s} \;=\; rB_{t-1,s}\left(1-\frac{B_{t-1,s}}{K_s}\right)-\;C_{t-1,s},$$

are a function of the growth rate r for the focal taxon, the biomass in pixel s at the previous time step $(t-1)$, the carrying capacity K_s for the pixel and the pixel catch $C_{t-1,s}$ in the previous time step.

To generate trawl catches at the pixel scale, trawl tracks were randomly placed by successively drawing from a Poisson point process with spatial intensity f(*K),* with a random direction, fixed tow width and log-normally distributed length. Each track intersects a portion w of pixel *s,* and therefore only a proportion w of the pixel's biomass is vulnerable to a given tow track. Pixel catch at time t is thus:

$$C_{t,s} \;=\; \sum_{\tau=1}^{\mathrm{T}} dw_{\tau,s}\prod_{i=1}^{\tau-1}\left(1-dw_{i,s}\right))\,B_{t,s}$$

with the set of 1 to τ tows leading to sequential depletion at the pixel scale (i.e., catch at time t accounts for removal of biomass by previous tows in the current time step by first computing how much biomass should be left given previous trawl effort in the cell in the current time step). This formulation is more realistic than assuming instantaneous removals, given the monthly resolution of the model.

Accounting for the proportion of the pixel intersected by the tow track (w) represents a slight modification from the pixel-level model developed by Ellis et al. (2014), which forms the basis of the RBS method (cf. scaling from experimental, pixel-level impacts to aerial impacts on a larger 'management grid'). The introduction of term w in the current formulation could lead to some bias in our application of the scaling relationships developed by Ellis et al. (2014), as the original formulation assumed that any pixel that is touched by a tow is fully impacted. However, the version used here represents a more realistic process since tow track data cannot be gridded on infinitesimally small pixels given GPS and reporting precision.

The model was run from a simulated, randomly generated equilibrium state. All scenarios were simulated for 20 years at a monthly time step, starting from unfinished equilibrium distributions.

2.3.3.3 Estimating RBS parameters from spatial data: statistical model

Estimating parameters of the logistic population model is most conveniently done on a spatially aggregated grid; fitting on the pixel scale (the scale at which the gear interacts with benthic fauna, i.e., 10s to 100s of metres) would be ideal, but computationally prohibitive, especially when considering applications beyond the simulated examples. Note that spatially continuous population models (e.g., Thorson et al. 2016) could also be fitted to this type of data. However, given the spatial scale of trawl impacts and the uncertainty about precise locations of removals along tow tracks, these may not add precision, are more difficult to set up, and are computationally more demanding.

The model provided by Ellis et al (2014) is a convenient starting point to scale from pixel-scale impacts to management grid level impacts. Impacts at the latter scale are effectively the output of the RBS method and were based on the initial Ellis et al. (2014) approach. Given a level of aggregation of fishing effort at the management grid scale, the model provides a relationship between *r,* the rate of population increase at the pixel scale, *d*, the proportional depletion per tow at the pixel scale, and their respective corresponding values (R_g and D_g) scaled-up to the grid ('management') scale. The relationship depends on the distribution of effort in space, that is, whether tow tracks are evenly spread, completely random, or clustered. This is measured via an 'effort concentration' parameter (β_g), which causes both D and R to be potentially variable in space (and hence indexed by the grid cell g).

The methods and procedures outlined by Ellis et al. (2014) were followed to estimate the effort concentration parameter (β_g) for each grid cell as the variance relative to the mean of the number of tows per pixel within each grid cell. Estimation of population dynamics parameters was carried out in a Bayesian framework using the general purpose Bayesian inference software Stan (Carpenter et al. 2015). Two model setups were used for estimates based on simulated data: the model was first run across the full time series to ensure the model could successfully estimate the depletion status of simulated populations when given sufficient data with reasonable spatial and temporal contrast. Then the estimation was repeated with the second half of the time series only, to reflect more realistic applications where observer data on particular species may only be available more recently (as is the case in New Zealand, cf. introduction of the Benthic Bycatch Form in January 2008). For this scenario, the initial population was estimated based on the RBS assumption (i.e., population at equilibrium with average fishing effort prior to the start of the time series) and a process error term. The estimation model equations at the grid ('management') scale for grid g are then:

$$
\begin{aligned}
B_{g,t=t_{init}} &= \left(1 - \bar{F}_{t\le t_{init}} \frac{D_g}{R_g}\right) \times K_g \times e^{z} \\
B_{g,t>t_{init}} &= B_{g,t-1} + R_g B_{g,t-1} \left(1 - B_{g,t-1}/K_g\right) e^{-F_{g,t-1} D_g} \\
C_{g,t} &= B_{g,t}\left(1 - e^{-F_{g,t} D_g}\right) \\
D_g &= \frac{\log\left(1 + \beta_g d\right)}{\beta_g} \\
\frac{R_g}{-\beta_g \log(1-d)} &= \frac{r \times \log\left(1 + \beta_g d\right)}{-\beta_g \log(1-d)}
\end{aligned}
$$

with $\underline{F}_{t\le t_{init}}$ the average swept area ratio for the period preceding t_{init}, F the swept area ratio, β_g an empirically derived estimate of effort concentration per grid cell, and D_g and R_g defined following Ellis et al. (2014).

The model was fitted on simulated observed catches. Observed catches were generated in one of two ways. First an average of 30% observer coverage was assumed, distributed randomly in the fleet, such that the observed catch can be straightforwardly scaled to the expected total catch. A key source of observation error in observer records comes from catches not being attributable to the exact location in a tow. To represent this source of uncertainty, the total catch for observed tows was averaged across the tow track ('smearing') to generate observed catch. Second, an additional source of observation error was considered in the form of a detection probability, since observations can be biased low by the potential non-detection of small quantities. While the smearing introduces observation error, the non-detection introduces bias—i.e., a non-linearity between observed catch-per-unit-effort and abundance. The latter was induced by setting the observation probability to $P\left(C_{g,t} > 0\right) = 1 - e^{-2*B_{g,t}}$. Parameter, ξ, was also introduced to represent detection rates.

The observation likelihood was written as a hurdle model of the form:

$$
C_{g,t}^{Obs_{pred}} = B_{g,t}\left(1 - e^{-F^{Obs}{}_g D_g}\right)
$$

$$
p(C_{g,t}^{Obs} \mid \theta, C_{g,t}^{Obs_{pred}}, \sigma_{Obs}) = (1-\theta) \times Lognormal\,(\log(C_{g,t}^{Obs_{pred}}), \sigma_{Obs}) \text{ if } C_{g,t}^{Obs} > 0,
$$

with θ the detection probability such that $P\left(C_{g,t}^{Obs} = 0\right) = \theta = 1 - e^{-\xi C_{g,t}^{Obs_{pred}}}$ estimates the probability of observing zero catch given predicted catch $C_{g,t}^{Obs_{pred}}$, with high ξ corresponding to perfect detection rates. σ_{Obs}captures measurement error in the observed catch.

Priors were vaguely informative for most parameters, reflecting a semi-informed scenario where priors may be elucidated from expert input, but large uncertainties remain about key parameters (the 0.025, 0.5, and 0.975 quantiles for key priors are given in square brackets; note r is on monthly scale for simulations):

$r \quad \sim Lognormal(\log(0.01), 1)$ [0.001,0.01,0.07]
$d \quad \sim Beta(3,20)$ [0.03, 0.12, 0.3]
$K_g \sim Lognormal\ (\mu_K, sd_K)$
$\mu_K \sim Normal[0](10, 10)$
$sd_K \sim Student\ t$ (3, 0, 25)
$z \quad \sim Normal\ (0, sd_z)$
$sd_z \sim Normal[0](0, sd_z)$

The model was estimated using Markov chain Monte Carlo (MCMC), running 8 chains for each of 3 effort scenarios, 2 observation error/bias scenarios, and using data from the start (equilibrium, 'Eq') of the fishery or from year 10 (of 20) (non-equilibrium, 'No eq').

2.3.3.4 Application to trawl bycatch from the Chatham Rise

We applied the above statistical model to estimate r and d from recorded catch of key taxa on observed trawls conducted on the Chatham Rise since 1989. We selected four sample groups spanning different biology and data availability in the observer dataset (Figure 2): Anthozoans (class Anthozoa), demosponges (class Demospongiae), sea pens (order Pennatulacea), glass sponges (class Hexactinellida). Note that only coarser taxonomic aggregations could be included as few records are identified at the species scale especially earlier in the time series (Figure 3), but many of the taxa are included in the coarser taxonomic aggregation used here (e.g., class Anthozoa includes *Enallopsammia rostrata*, *Goniocorella dumosa*, *Madrepora oculata*, *Solenosmilia variabilis*, Actiniaria, Alcyonacea, and Antipatharia (orders), etc.).

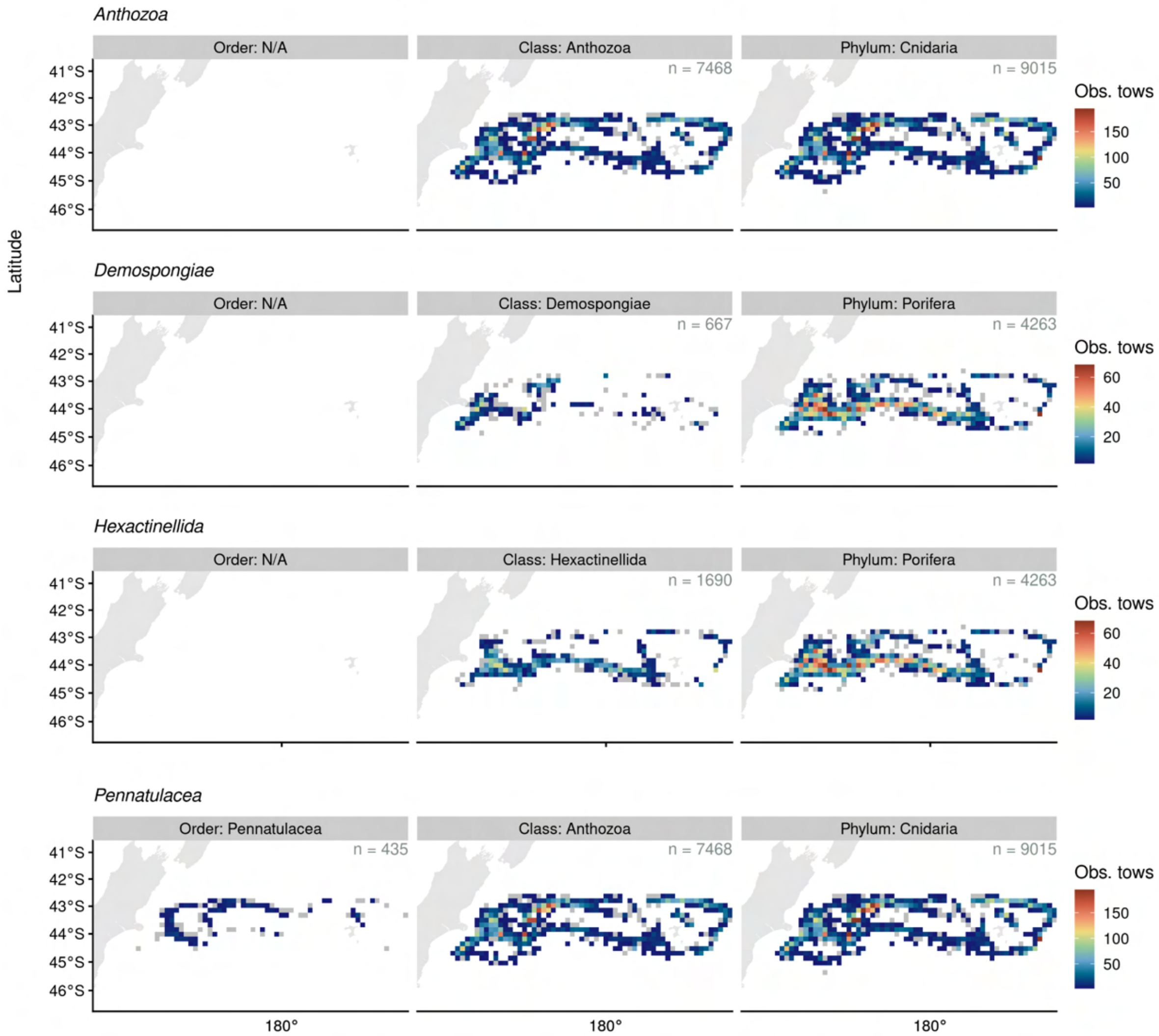


Figure 2: Location of records (all years) for the groups used for the application of the benthic status assessment model and corresponding higher taxonomic ranks (where applicable). The number of records in the observer dataset is shown in the top right corner, with locations with a single observation in grey.

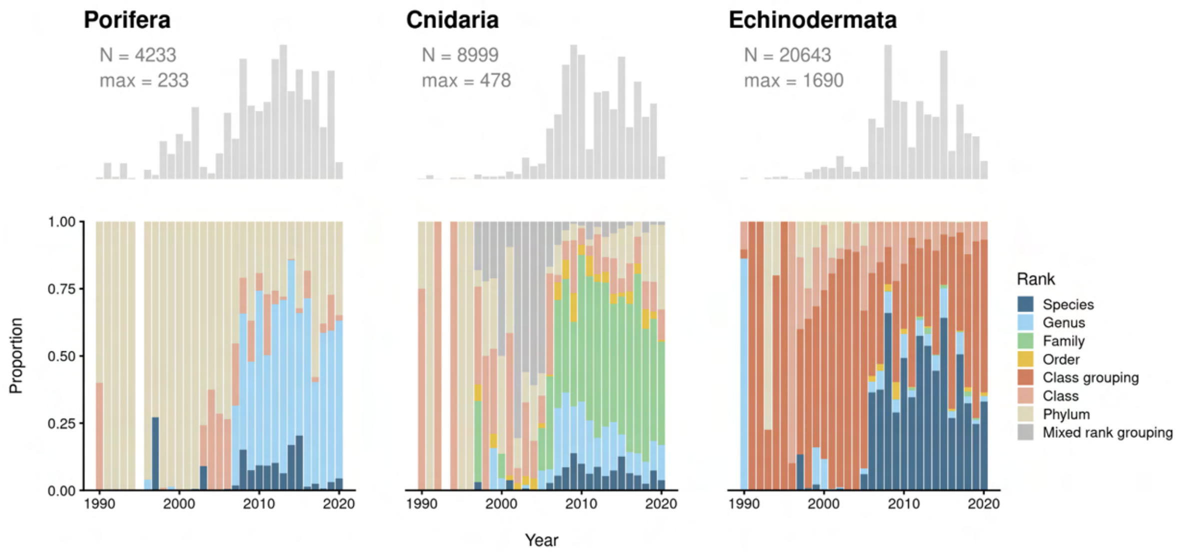


Figure 3: Number of observer records per year, with key phyla in panels (top) and proportion of annual observer records by taxonomic rank (bottom). For the top panel, the total number of observations and the maximum number of annual observations is shown in the top-left corner.

There was a remarkable increase in the number of benthic bycatch records by observers since 2008 following the introduction of the Benthic Bycatch Form. Therefore the model was started in 2008, and used effort prior to 2008 to estimate the initial distribution across the model grid ($B_{g,t=t_{init}}$) at the start of the model. A 20 km grid was used, which required 840 grid cells to cover the Chatham Rise area. All datasets were restricted to within the area of the hoki survey core strata in order to cover an area of relatively consistent depth. Gear types were further subsetted for modelling to retain only those that had catch of focal taxa, leaving a total of nine gear types and 456 grid cells with effort associated with these gear types, and a total of 4475 observed grid cell, gear, and year combinations.

For each grid cell, year, and gear category (as defined above), the SAR was calculated by month for commercial and observer data, as well as aggregation metric β. The latter was calculated using 500 m × 500 m pixels and counting the number of tows $N_{s \in g,G}$ intersecting each pixel in grid g, giving $\beta_{g,G} = \frac{\mathrm{SD}(N_{s \in g,G})^2}{\overline{N_{s \in g,G}}} - 1$ for each 20 km grid cell.

The model described in the simulation setting above was adapted to span multiple gear categories. Notably, the equations for R_g and D_g pertain to each grid cell, but the scaling relationships only hold for constant values: f d with variable d and β, R_g is not analytically determined as it depends on a mixture of d and β. The simplifying assumption was made, that the resulting R_g is the effort-weighted mean of R_g calculated across gear types—meaning the observed R_g value will be most affected by the gear configurations that dominate the effort in any grid cell. All other formulae extend straightforwardly to catch from multiple gears.

Priors were adjusted to reflect values derived for VME indicator taxa given by SPRFMO (2020) (Table 3). The prior for μ_K (on the log scale) is notoriously difficult to specify in Bayesian assessment models as high bounds effectively lead to *a priori* optimistic settings (Thorson & Cope 2017, Kim & Neubauer in prep.). Here a pragmatic approach was used, that specifies this prior rather narrowly but allows for data driven deviation from K_g via the relatively wide student-t prior on sd_K. Sensitivity to these prior assumptions should be more thoroughly explored in future applications of this method. In summary, Chatham Rise specific priors were defined as given in Table 3.

Table 3: Priors used for key parameters in the statistical model estimating relative benthic status of each taxonomic group. Values in square brackets show the 5th and 95th percentiles.

Anthozoa (based on priors for coral VME indicator taxa in SPRFMO 2020)
$r \quad \sim Lognormal(\log(0.2), 0.1)$ [0.13, 0.20, 0.3] $d \quad \sim Beta(30,15)$ | [0.52, 0.67, 0.80]
Demospongiae
$r \quad \sim Lognormal(\log(0.24), 0.1)$ $d \quad \sim Beta(49.2,80)$
Hexactinellida
$r \quad \sim Lognormal(\log(0.24), 0.1)$ [0.13, 0.20, 0.3] $d \quad \sim Beta(49.2,80)$
Pennatulacea
$r \quad \sim Lognormal(\log(0.39), 0.15)$ [0.13, 0.20, 0.3] $d \quad \sim Beta(41.3,80)$
All groups
$K_g \quad \sim Lognormal\ (\mu_K, sd_K)$ $\mu_K \quad \sim Normal[0](\log(400), 3)$ $sd_K \sim Student\ t$ (3, 0, 25)

All remaining priors remained identical to the initial simulation-estimation set-up. All models were run using 5–8 MCMC chains, with convergence assessed visually and by inspection of multivariate (adjusted) Rhat statistic.

2.3.4 Application of Mormede et al. (2021) methods for Chatham Rise

The quantitative risk assessment method developed by Mormede et al. (2021) comprises two parts. In the first, a spatial population layer is built for a given taxon and, in the second, this layer is used in combination with a trawl footprint estimate to calculate the biomass trajectory for the taxon using a Bayesian spatial population model. Here we update and extend the estimation of spatial population layers for selected benthic taxa, using models based on biomass survey data from Chatham Rise trawl surveys (Stevens et al. 2021). Trawl surveys were conducted on the Chatham Rise yearly from 2007 to 2016, and then again in 2018 and 2020.

Bayesian spatial population models were not first produced in this study. Following the methodology of Mormede et al. (2021), vector autoregressive spatio-temporal (VAST) models were built to produce relative biomass estimates for Chatham Rise across space and time. VAST is a flexible modelling framework that allows for different data types (including presence/absence, density, and biomass data) and enables spatial, temporal, spatio-temporal, and vessel effects to be estimated, as well as habitat and catchability information to be incorporated (Thorson & Barnett 2017, Thorson 2019). VAST also allows various, easily interpretable, outputs including maps and plots to be produced. Following the recommended model structure of Mormede et al. (2021), habitat and environmental covariates were not considered in this study. Additionally, the analyses did not include vessel effects and catchability covariates because surveys followed a standardised sampling procedure (Hurst et al. 1992). The VAST framework also allows for joint-species distribution modelling where multiple categories (Thorson & Barnett 2017) can be modelled simultaneously (i.e., multiple species, length, or ages, etc.). However, categories were not considered in the present study. VAST models are spatial-delta generalised linear mixed models consisting of two linear predictors for probability of encounter $p_1(i)$ and positive catch rates $p_2(i)$). The first linear predictor is given as:

$$p_1(i) = \left(\beta_1^* + \sum_{f=1}^{n_{\beta 1}} L_{\beta 1}(f)\beta_1(t_i, f) + \sum_{f=1}^{n_{\omega 1}} L_{\omega 1}(f)\omega_1^*(s_i, f) + \sum_{f=1}^{n_{\epsilon 1}} L_{\epsilon 1}(f)\epsilon_1^*(s_i, f, t_i) \right).$$

The second linear predictor is given as:

$$p_2(i) = \left(\beta_2^* + \sum_{f=1}^{n_{\beta 2}} L_{\beta 2}(f)\beta_2(t_i, f) + \sum_{f=1}^{n_{\omega 2}} L_{\omega 2}(f)\omega_2^*(s_i, f) + \sum_{f=1}^{n_{\epsilon 2}} L_{\epsilon 2}(f)\epsilon_2^*(s_i, f, t_i) \right)$$

where $\left(\beta^* + \sum_{f=1}^{n_\beta} L_\beta(f)\beta(t_i, f)\right)$ accounts for temporal variability, $\left(\sum_{f=1}^{n_\omega} L_\omega(f)\omega^*(s_i, f)\right)$ accounts for spatial variability, and $\left(\sum_{f=1}^{n_\epsilon} L_\epsilon(f)\epsilon^*(s_i, f, t_i)\right)$ accounts for spatio-temporal variability in each of the predictors. The VAST framework allows flexible specification of the temporal and spatio-temporal correlation structure. Temporal and spatio-temporal correlation terms were set as random effects. Temporal effects were estimated as a random walk between years or as independent between years if it was unable to be estimated. The temporal terms of the spatio-temporal effects were estimated as first-order autoregressive processes. If spatio-temporal effects were unable to be estimated, they were dropped from the model. The present study used a Poisson-link function (Thorson 2018) with an encounter probability component (r_1(i)), and a positive catch rates component ($r_2(i)$). These are derived from the two linear predictors so that:

$$r_1(i-) = 1 - (-a_i \times \exp(p_1(i)))$$

and,

$$r_2(i) = \frac{(a_i \times \exp(p_1(i)))}{r_1(i)} \times \exp\big(p_2(i)\big).$$

The area swept by each survey tow (i) is given by a_i. Biomass estimates $d(g, t)$ are given as the product of $r_1(g, t)$ and $r_2(g, t)$ which were estimated at each grid cell g and time t. Table 4 gives definitions of each of the VAST model terms.

Two VAST models were constructed. The first model used sea cucumber or holothurian (class Holothuroidea) biomass data from the Chatham Rise trawl survey data to assess the impact of bottom-contacting fishing to soft-sediment habitats. The second model used stony coral *Goniocorella dumos*a biomass data to assess the impact of bottom-contacting fishing to hard substratum habitats. However, there were relatively very few observations of this stony coral throughout the Chatham Rise (number of data points greater than zero between 3 and 8 percent per year) and a model could not be fitted to the data. Instead, a model was fitted at the class level of Anthozoa. This meant that there were more data available per year (number of data points greater than zero between 46 and 78 percent per year). These data were given at each sampling station (Figure 4) on the Chatham Rise as weight in kilograms standardised to the area swept by the trawl gear. The standardised trawling procedure is described in detail by Hurst et al. (1992).

The impact of fishing was calculated using the methodology outlined by Mormede et al. (2021). The tow-by-tow footprint, f_{tgz} is the area towed (footprint) of a tow in a given time *t* (2007–2020), grid cell g, and fishery z. This study explored the fish/squid fishery and the scampi fishery, as per Mormede et al. (2021). The vulnerability V_z (also known as depletion) of holothurians due to fish/squid trawling and scampi trawling were set to 0.1 and 0.8, respectively (Mormede et al. 2021). Whereas the vulnerability values for Anthozoa due to fish/squid trawling and scampi trawling were set to 0.18 and 0.36, respectively. The vulnerability of Anthozoa was determined by taking an average of the mean depletion values by gear category (GC) derived from the modified RBS analysis (see Section 3.3.3.3, Table 10). The vulnerability of Anthozoa due to fish/squid trawling was taken from GC3, GC8, GC9, GC13, GC14, and GC15. Note that GC2 was not used because there was very little trawling by this gear type on the Chatham Rise. The vulnerability of Anthozoa due to scampi trawling was taken from GC4, GC5, GC6, and GC7.

The bottom fishing impact mortalities I_{tgz} at each time, grid cell, and fishery were calculated as:

$$I_{tgz} = 1 - \prod_{tows} \left(1 - f_{tgz} \times V_z\right).$$

The fishing impact overlap metric $I_{tgz}^{(0)}$ gives the proportion of biomass at a time and grid cell impacted due to trawling for a fishery. This is calculated as:

$$I_{tgz}^{(0)} = I_{tgz} \times \frac{B_{tg}}{\sum_{g=1}^{n_g} B_{tg}}$$

Where, B_{tg} is the biomass of either holothurians or Anthozoa at time *t* and grid cell *g* estimated using VAST models. The models used $n_g = 2000$ grid cells. VAST models were assessed using dharma residuals (Hartig 2020). Dharma residuals are designed for mixed effects regression models. Dharma residuals are generated using the *dharma* R-package, where a simulation-based approach is used to standardise residuals between zero and one. The fitted model is used to simulate response values for each observation and the cumulative distribution function (CDF) is calculated from these simulated values. Residuals are calculated by taking the value of the CDF corresponding to the value of the observed value. This process is repeated 250 times. An observed value with a residual value of zero indicates that all simulated values are greater than that observed, whereas an observed value with a simulated value of 0.5 indicates that only 50% of the simulated values are greater than that observed.

Table 4: Definitions of the terms used in the VAST models. See Thorson (2019) for a full account of model parameters. The predictor number (1 or 2) was removed from a term if the definition was equivalent in each model predictor.

Term	Type	Definition
i	Index	The observation number.
f	Index	The factor number.
t	Index	The year.
g	Index	A grid cell.
z	Index	The fishery. Either fish/squid or scampi.
t_i	Data	The year for observation *i*.
s_i	Data	The spatial location for observation *i*.
a_i	Data	The area swept for observation *i*.
V_z	Data	The vulnerability/depletion value for a species/group.
n_g	Dimension	Number of grid cells.
n_β	Dimension	Number of factors for temporal effect.
n_ω	Dimension	Number of spatial factors.
n_ϵ	Dimension	Number of spatio-temporal factors.
β^*	Fixed effect	Linear predictor time-average intercept.
$L_\beta(f)$	Fixed effect	Scalar for temporal covariation in factor f.
$\beta(t_i, f)$	Random effect	Temporal effect for time t_i and factor f.
$L_\omega(f)$	Fixed effect	Scalar for spatial covariation factor f.
$\omega^*(s_i, f)$	Random effect	Spatial factors for location s_i and factor *f*.
$L_\epsilon(f)$	Fixed effect	Loadings matrix for spatio-temporal covariation in factor f.
$\epsilon^*(s_i, f, t_i)$	Random effect	Spatio-temporal factors for location s_i, factor *f*, and time t_i.
$p_1(i)$	Derived quantity	The first model linear predictor.
$p_2(i)$	Derived quantity	The second model linear predictor.
$r_1(i)$	Derived quantity	Probability of encounter.
$r_2(i)$	Derived quantity	Positive catch rates.
$r_1(g, t)$	Derived quantity	Probability of capture estimated at grid cell g and time t.
$r_2(g, t)$	Derived quantity	Positive catch rates estimated at grid cell g and time t.
$d(g, t)$	Derived quantity	Predicted density for grid cell g and time t.
I_{tgz}	Derived quantity	The bottom fishing impact mortality.
$I_{tgz}^{(0)}$	Derived quantity	The fishing impact overlap metric.
f_{tgz}	Derived quantity	The tow-by-tow fishery footprint.

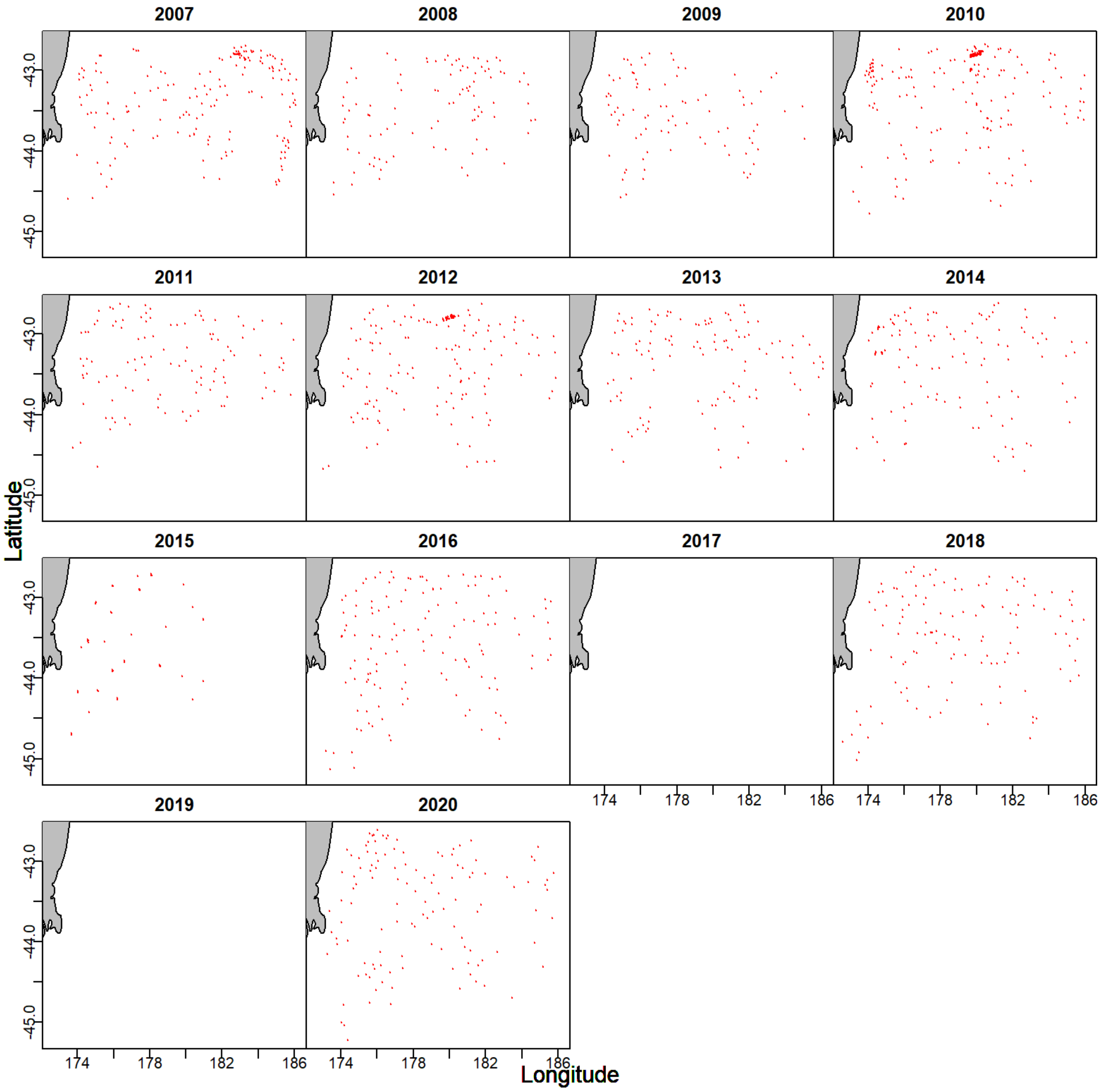


Figure 4: Locations of stations in the trawl surveys conducted between 2007 and 2020. See Stevens et al. (2021) for more detail.

3. RESULTS

3.1 Characterisation and categorisation of mobile bottom gear types and configurations

3.1.1 Trawl and dredge gear types

3.1.1.1 Two or multiple panel bottom trawl

Two-panel net: Nets having only two major parts, upper and lower, and these panels are attached to each other laterally to form the two seams. The upper part invariably includes an overhang. The cross section of the net is elliptical in shape (Figure 5).

Four-panel net: Nets having four parts, the upper, the lower, and the two lateral side panels. Such trawl nets may comprise up to eight panels, particularly those where high vertical opening is required.

Ground gear configurations vary depending on bottom type and target species.

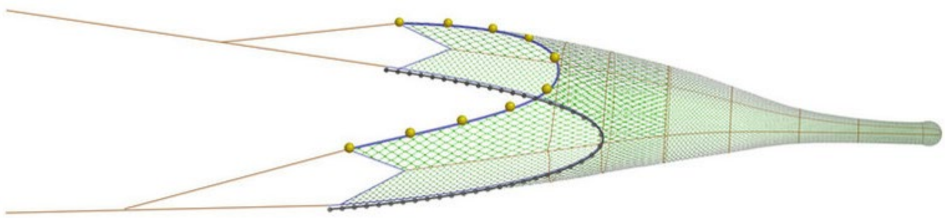


Figure 5: Typical multipurpose Two-panel bottom trawl (image source: with permission from Seafish).

3.1.1.2 Alfredo-style bottom trawl (e.g., orange roughy, oreo hill fisheries)

The Alfredo-style trawl is characterised by cut away lower wings and a short ground rope (10–40 m) (Figure 6). Ground gear configurations vary depending on bottom type and target species.

The lower wings on these trawls are either absent or significantly reduced to minimise the risk of net damage, especially on rough terrain. The sweeping gear and ground gear may be short to enable more rapid and predictable gear response to facilitate accurate targeting of fish aggregations or for use on rough terrain.

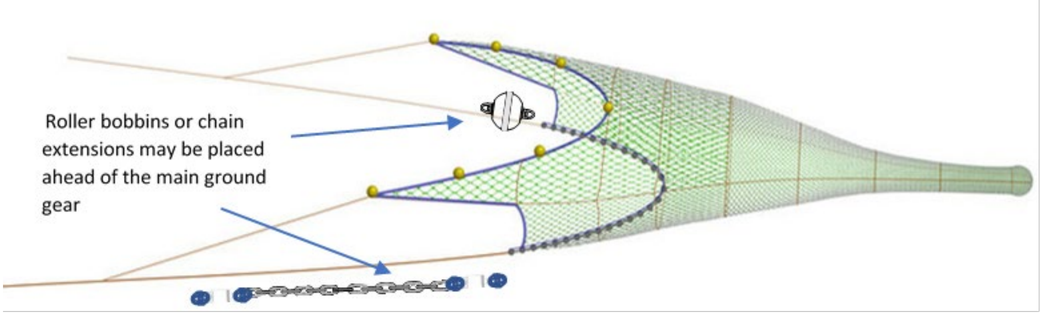


Figure 6: Alfredo-style trawl with cut away lower wings and short ground rope (image source: used with permission and modified from Seafish).

3.1.1.3 Scraper trawl

Scraper trawls are two-panel trawls with a low headline height, extended wings, and ground rope (Figure 7). The ground gear is made up of small tightly packed components typically made up of rubber cookies (60–150 mm) with weights added. This trawl design is used to target bottom dwelling fish species or when reduced headline height is needed to avoid non target species higher in the water column.

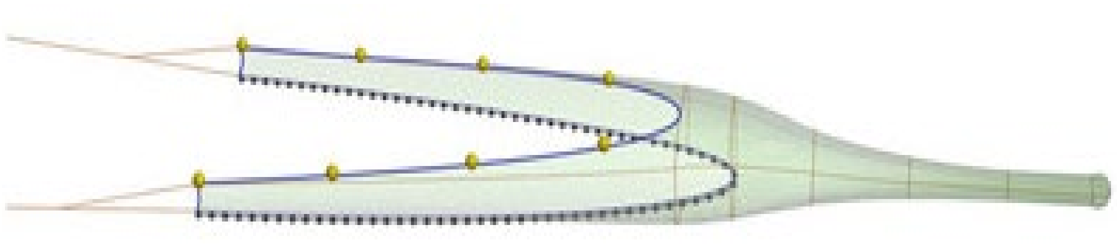


Figure 7: Scraper trawl (image source: with permission from Seafish).

3.1.1.4 Twin-rig trawl & demersal pair trawl

When a single twin-rig trawl is used, a heavy weight known as a clump or roller maintains the two nets relative to each other and has continual contact with the seafloor (Figure 8a). Ground gear may consist of rubber or steel bobbins, rubber discs, or chain. Pair trawls may use longer sweeping wires than a single trawl to achieve a wide effective wing and door spread (Figure 8b).

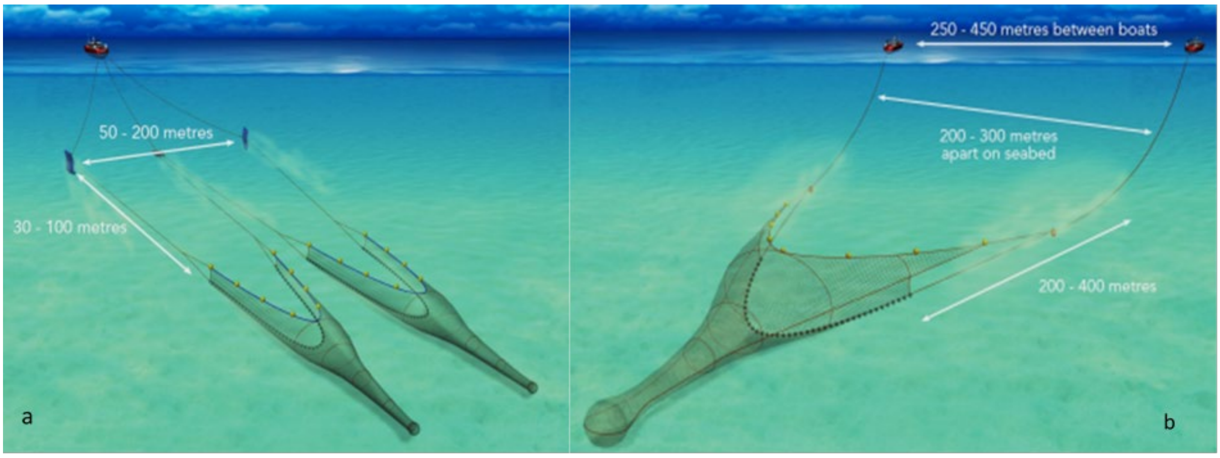


Figure 8 (a): Twin rig demersal trawling (two trawls – one boat); (b) Demersal pair trawling (two boats – one trawl) (image source: with permission from Seafish).

3.1.1.5 Scampi trawl

Vessels targeting scampi use a two-panel trawl with a low headline height. Ground gear is of small diameter, tightly packed, and suitable for soft substrates, typically comprising rubber cookies (30–100 mm), with weights or chain added (Figure 9a).

Trawls are set up in a double- or triple-rig configuration (Figure 9 b&c), with each net having a wingspread of 25–30 m. The number of nets used depends on fishing and weather conditions. Nets can be deployed by separate warps from blocks outside the line of the hull or can be towed by a single warp deployed from the centreline of the vessel over the transom (two nets only).

Trawl doors are towed from a bridled arrangement from the main warps. Sweeping gear is reduced with the top and bottom bridles linking directly from the top and bottom of the trawl doors that are typically rectangular with curved foils, which due to design and proximity to the trawl, travel the same path as the net.

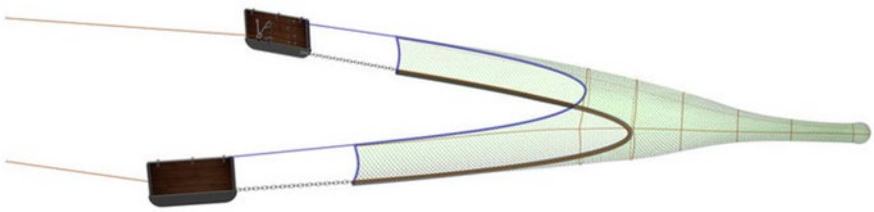


Figure 9a: Single scampi trawl (image source: with permission from Seafish).

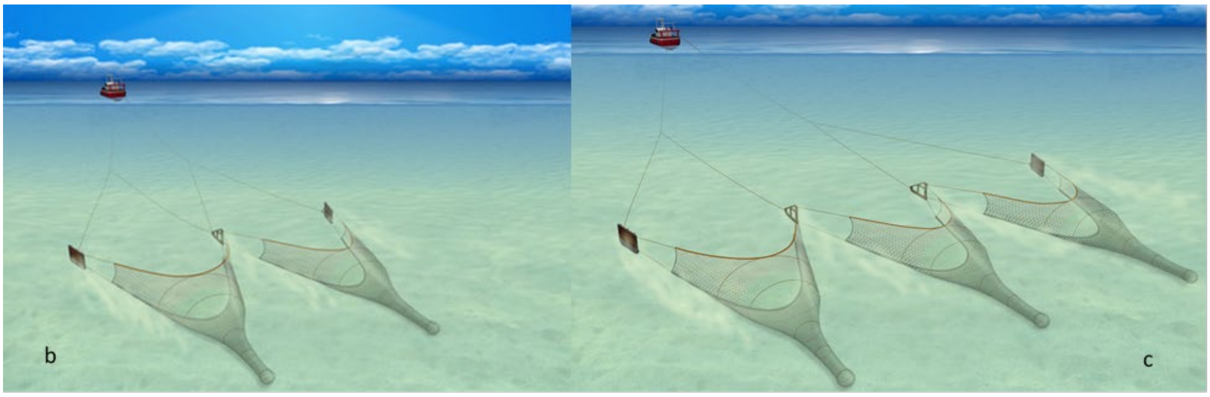


Figure 9: (b) Twin or (c) triple rig trawl arrangements as used by the New Zealand scampi fleet (image source: with permission from Seafish).

3.1.1.6 Midwater or pelagic trawl

As its name suggests, in normal use, midwater trawl gear would not be expected to contact the seabed. However, Tingley (2014) noted midwater trawl gear can be, and is, effectively operated as demersal trawl gear, designed to fish on the seabed in some fisheries.

The pelagic-style trawl doors used with these nets are designed to remain above the seabed. Midwater trawls have minimal ground gear components; however, chain or chain wrapped with rope may be used in the centre section, in place of wire. Unlike bottom trawls, pelagic trawls are towed on the headline and weights on the bridles or wing ends provide the vertical force to open the net in a downward vertical direction (Figure 10).

The points of bottom contact when fished hard down are from the wing-end weights backwards, possibly to the cod end. Variations in the extent of bottom contact will result from differences in the way the gear is configured (Baird et al. 2011). As a rule-of-thumb, the weight of wing-end weights used (in kilograms) are approximately half the vessel's horsepower per side (O. Hoggard, Motueka Nets, pers comm.).

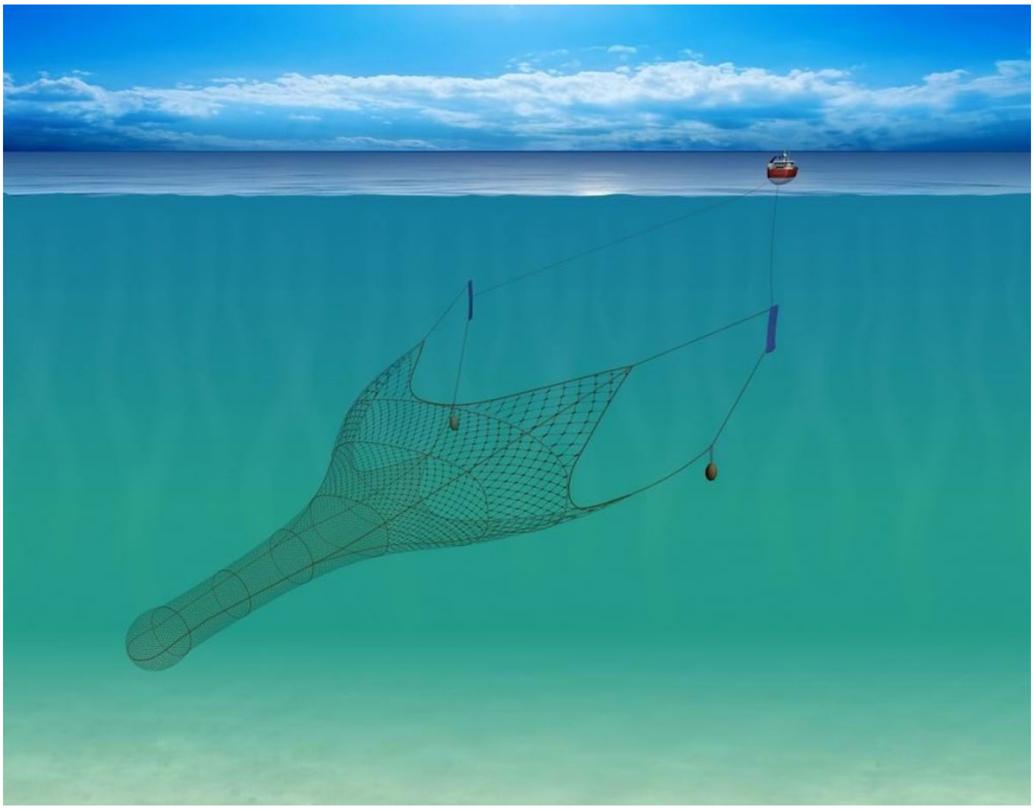


Figure 10: Midwater trawl (image source: with permission from Seafish).

3.1.1.7 Scallop dredge

Scallops are harvested using either box or ring-bag dredges.

A type of box dredge is used in the Northland and Coromandel scallop fishery. The box or collection part of the dredge is constructed of steel framing and steel mesh and is dragged along the seafloor on narrow (typically19 mm) steel runners that keep the box off the seafloor (Figure 11a). The specifications of these box dredges vary between fishers, reflecting individual preferences and the substrate type dredged, but a typical box dredge has a width of 2.44 m (the maximum allowable width is 2.5 m), length of 2.4 m, height of 400–500 mm and weight of about 150–200 kg (Beentjes & Baird 2004). Fisheries (Commercial Fishing) Regulations (1986) allow two dredges to be towed at once. If a single dredge is towed, the maximum width is 2.5 m, or if two dredges are used, width must not exceed 1.4 m each.

Specifications of ring-bag dredges vary among vessels, but dredges are generally about 2.4 m wide, 2.7 m long, and weights range from about 250 kg to 450 kg; they are designed to maintain a vertical mouth opening of about 0.5 m. The headframe is raised on short skids keeping it clear of the seafloor. Ring-bag dredges used in the Nelson area do not have cutting bars or tines and are not designed to dig into the substrate; tickler chains are used as the ground tending gear and only these and the ring-bag are in contact with the substrate (Figure 11b). Challenger Area Commercial Fishing Regulations (1986) restrict the width of this type of dredge to a maximum to 2.5 m, and only two dredges at a time may be towed (Beentjes & Baird 2004). The construction of the ring dredge varies between the Northern and Challenger areas (Williams et al. 2014). Vessels in the Chatham Islands scallop fishery use a ring-bag dredge like that of the Nelson scallop fishery, except that it has a bar fitted with spring-loaded tines (Figure 12).

Figure 11: (a) Box dredge and (b) ring-bag dredge used to collect scallops (image sources: J. Williams, NIWA).

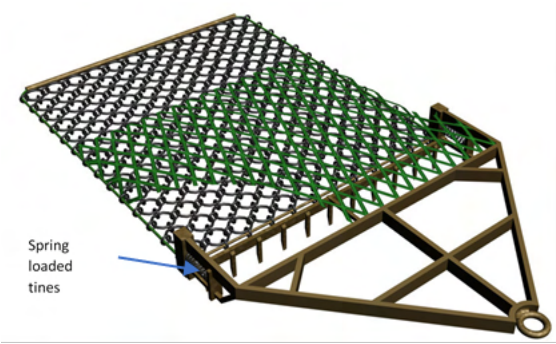


Figure 12: Example of scallop ring-bag dredge used in the Chatham Islands (image source: used with permission and modified from Seafish).

3.1.1.8 Oyster dredge

Dredging for oysters occurs in Foveaux Strait using a double-bit dredge, consisting of a ring-bag attached to a heavy steel frame (Figure 13). A cutting bar, known in the oyster industry as a bit bar, provides the gear substrate interface. Commercial Fishing regulations (1986) restrict fishers to two dredges per vessel and each dredge may have a bar or bit not exceeding 3.35 m in length. Specifications for a Foveaux Strait double bit dredge currently in use are: weight 550 kg, overall length 1.65 m, overall width 3.32 m, mouth height 0.44 m, bit bar depth 0.08 m (Keith Michael, pers. comm.). The internal diameter of the metal rings that make up the bag of most dredges is 62 mm, with either 20 mm or 45 mm connector rings.

Figure 13: Foveaux Strait oyster dredge (image source: Tony Smith).

3.1.1.9 Danish seine

Trawls and dredges are the only strictly mobile gears used in New Zealand fisheries. However, Danish seining, while using an essentially static net, does include a degree of net and ground gear movement across the seabed which could impact benthic fauna. Although this method is not being incorporated into the benthic impact assessment component of this project, a brief description of this form of fishing is included here for potential consideration in future assessments. Historically, this type of fishing largely occurred off the east coast of the North Island. When fishing with a Danish (anchored) seine, the gear is set out from an anchor point in a roughly triangular area on the seabed using very long ropes (Figure 14). As the two ropes are winched in from the vessel, the area between the ropes diminishes and the seine gradually closes to herd the fish and, towards the end of the haul, moves forwards in the same way as a bottom trawl.

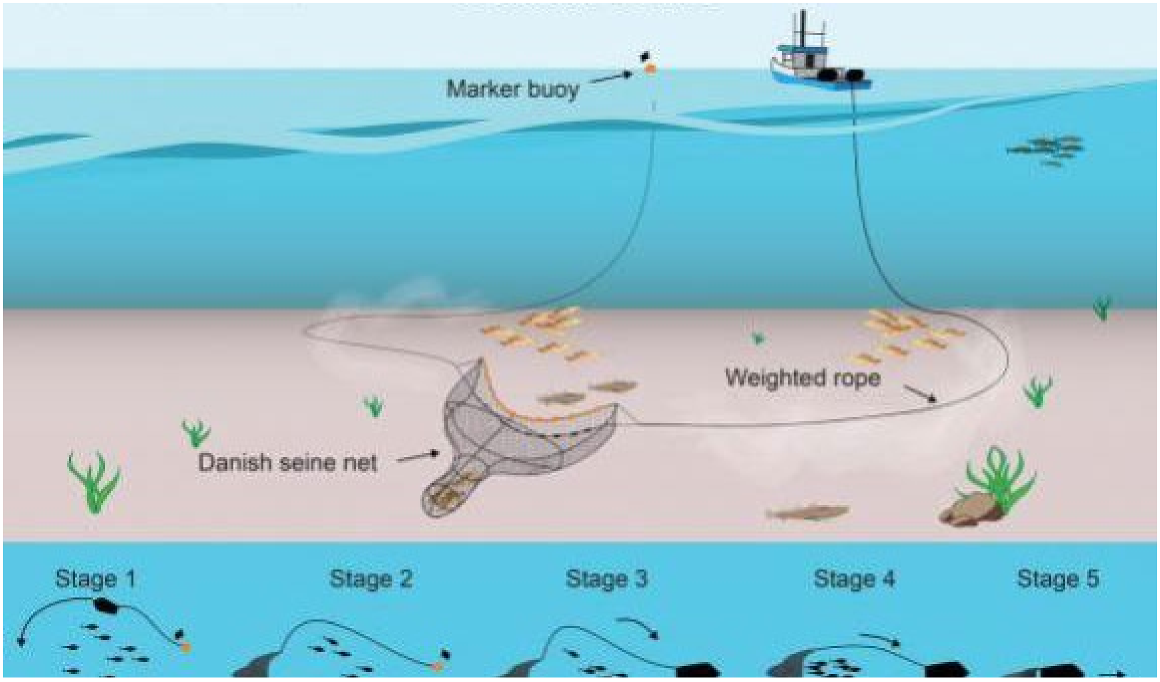


Figure 14: Danish seining (image source: used with permission and modified from Australian Fisheries Management Authority).

3.1.2 Trawl ground gear types

3.1.2.1 Wire

The simplest type of ground gear used on trawls is a wire ground rope, which is attached directly to the net (Figure 15). The wire ground rope may be wrapped or 'served' with rope. Wire ground rope is suitable for trawling on 'clean' or 'soft' bottoms when targeting demersal fish.

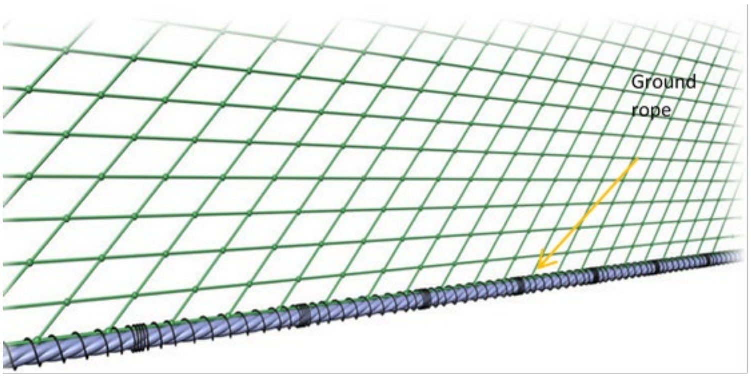


Figure 15: Wire or rope-covered wire ground rope (image source: used with permission and modified from Seafish).

3.1.2.2 Chain

Chain is sometimes attached to the wire ground rope. Chain adds weight to improve bottom contact or to 'tickle' the bottom, disturbing demersal species and making them flee upward into the mouth of the net. How the chain is arranged on the ground rope may vary depending on bottom type and target species (Figure 16).

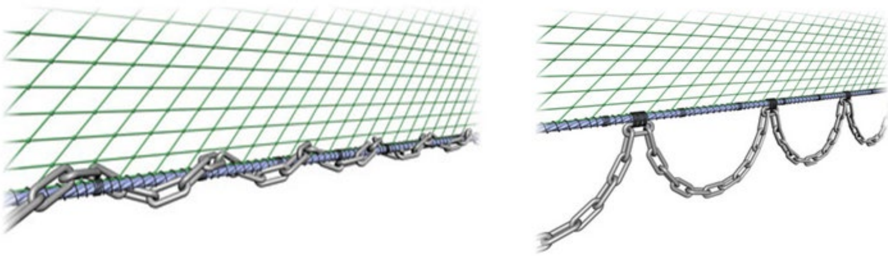


Figure 16: Examples of chain attached to the ground rope (image source: with permission from Seafish).

3.1.2.3 Rubber disks or cookies

Rubber disks or 'cookies' are ~50–300 mm diameter disks of rubber, with a central hole, that are packed tightly and threaded onto wire or chain, and linked to the ground rope via chain droppers (Figure 17). This ground gear is used where good bottom contact is needed, typically on mixed species inshore and scampi trawls.

The rubber disks are cut out from old tyres; they are of low density and can lose up to 95% of their weight in water. Weights (or chain) are commonly added to the centre section of this type of ground gear to improve bottom contact.

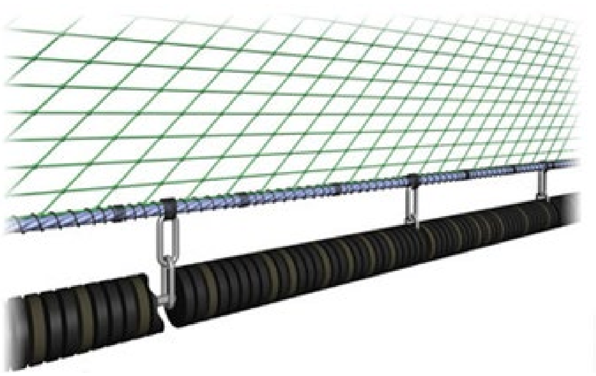


Figure 17: Rubber disks or cookies (image source: with permission from Seafish).

3.1.2.4 Bobbin rigs

Bobbin rigs are used where hard/rough seabed may occur. Made from hollow steel or solid rubber, they range in size between 15 cm and 65 cm. Steel bobbins have holes to allow water to enter and prevent implosion at depth and may have a raised central wear-band. Bobbins are used in combination with smaller rubber disks (see above), and they rotate as the gear moves over the seabed (Figure 18a&b). Modified chain droppers, known also as Lancasters, are used to attach the bobbin rig to the ground rope/net. They may attach directly to the ground rope or may include a traveller wire which allows for better alignment of the ground gear components (Figure 18c).

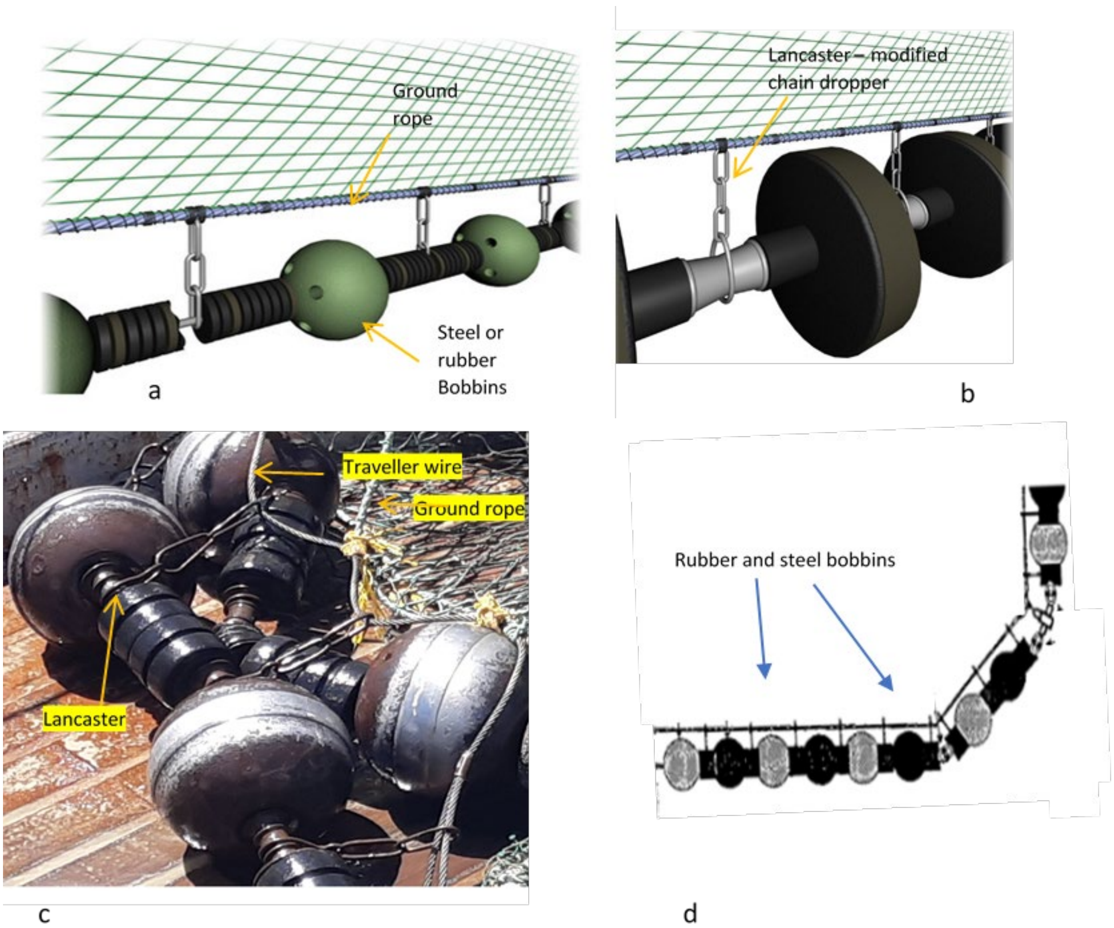


Figure 18: (a) Steel or rubber spherical bobbins attached via chain droppers. (b) Large rubber disks attached via Lancasters – modified chain droppers (image source: used with permission and modified from Seafish). (c) Bobbin rig attached to ground gear with a traveller wire arrangement. (d) Rubber and steel bobbins from an orange roughy trawl (image source: NIWA).

3.1.2.5 Rockhopper rig

A rockhopper rig is made up of rubber disks (60–800 mm) which may be joined directly to the ground rope (Figure 19a) or may include a wire to align the components (Figure 19b). Rockhopper disks do not rotate; they are tensioned and designed to ride or spring over rocks or objects on the seabed. These rigs are used where rocks or other objects may occur on the seabed.

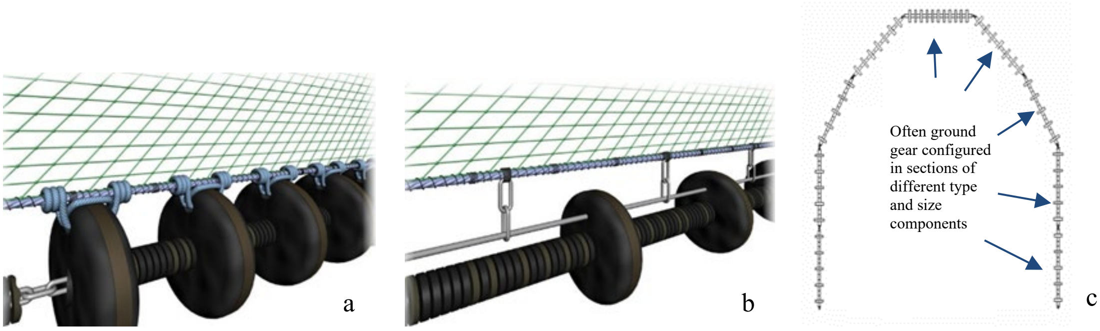


Figure 19: (a) Rockhopper disks connected directly to ground rope, and (b) via droppers (image source: with permission from Seafish). (c) Typical rockhopper rig arrangement with larger disks in centre and reducing diameter disks toward the wing ends (image source: used with permission and modified from Harley & Ellis 2007).

3.1.3 Sweeping gear

Sweeps and bridles are cables that connect the trawl doors to the trawl net and may be in contact with the seabed for part of that distance. The selection of length of these cables and their ‘angle of attack’ will determine the area of seabed they sweep (Grieve et al. 2015). Sweeps and bridles are constructed of fibre or steel core dye-form wire, PVC coated wire, Spectra™, Dyneema™, synthetic or combination rope (strands of wire rope covered in polypropylene fibre twisted together) (Figure 20). Vessels trawling on soft substrate, like those targeting scampi, may use lower density material such as Malleta or plastic covered wire to reduce the chance of the sweeping gear ‘bogging’ in fine sediments.

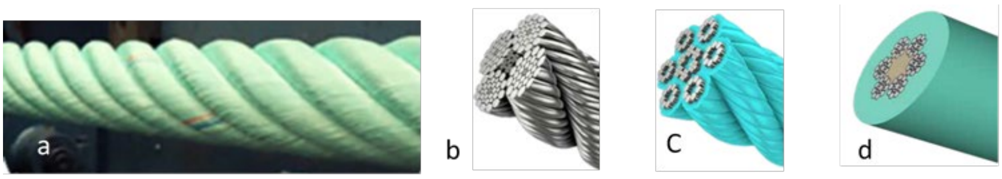


Figure 20: Materials used to construct sweeps and bridles: (a) Malleta, (b) wire rope, (c) combination wire rope, and (d) plastic coated wire rope (image source: with permission from Bridon Cookes).

Bridles are cables split from the sweep wire and can be adjusted to tune the trawl to suit the bottom type, or to target specific species. It is quite common for fishers to make small alterations to the rigging of the bridles, such as when changing from daylight to night-time fishing or from soft to harder ground. Shortening the top bridle typically has the effect of reducing the headline height of the trawl which in turn can increase bottom contact of the ground gear; conversely adding length to the top bridle will increase the headline height of the trawl and reduce bottom contact.

Steel hardware such as hammerlocks (for joining sections of trawl gear) and swivels (to prevent tangling of cables) form part of the componentry of the sweeping gear. Additions to the sweeping gear can include rubber disks or bobbins which may serve to protect the cables and reduce bottom contact or increase their fish herding effectiveness.

3.1.4 Trawl doors/otter boards

Trawl doors or otter boards are used to spread the mouth of the trawl net open and herd fish (through physical presence, noise, or creation of a sediment cloud) towards the centre of the net. The doors are towed over the seabed at an oblique angle to the direction of the fishing gear. Rectangular flat trawl doors are commonly operated at an angle of 30°, while more efficient designs can be operated at 20° or less, reducing potential contact area with the seabed. The operation and seabed contact of the doors can be influenced by factors such as towing speed and movement, warp to depth ratio, type and weight of the doors, bottom type, and trawl and door rigging and geometry. Doors vary in design from the traditional flat, rectangular boards that create great turbulence in their wake to modern slotted or foil boards that reduce turbulent drag by maximising hydrodynamic efficiency. Commonly used trawl doors in New Zealand fisheries include those shown in Figure 21. Within the inshore fleet, Clement & Associates (2008) found ~75 % of doors were steel and 25% wooden, and that door area and weight was proportional to vessel horsepower. They also reported few fishers knew the 'angle of attack' for their doors which effects the area and intensity of contact with the seabed. The trawl doors used by the New Zealand deepwater bottom trawlers typically range from ~1200 kg to 6000 kg in weight and from ~4 m² to 10 m² in area (Parker et al. 2008).

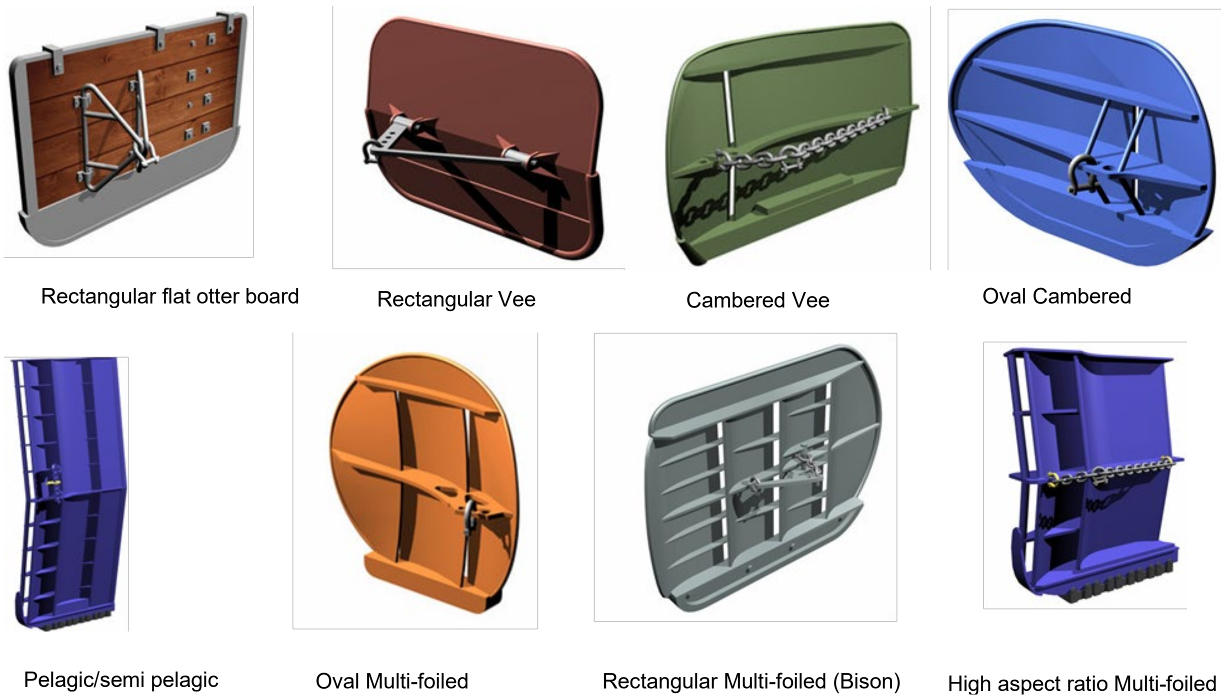


Figure 21: Commonly used trawl doors within the New Zealand trawling fleet (image sources: used with permission and modified from Seafish). For image of Super-V style door, see https://www.nichimo-marine.jp/products/trawls/trawl_doors/002_super_v_type_door.html.

3.1.5 Other bottom trawl components

The following trawl gear components may form part of the main ground gear or sweeping gear. They may be used for the transition from bottom bridles to ground rigs, to reduce bottom contact, reduce wear and tear, herd fish, or to extend the contact area of the ground gear.

3.1.5.1 Wing-end extension components

Wing-end extensions increase bottom contact between the bottom bridle and the lower wing of the trawl particularly where the trawl design features cut away lower wing panels. These extensions may be a length of chain between the bottom bridle and the main ground rig. Other extension components include Dan Lenos or rolling bobbins (Figure 22a), or rubber disks (Figure 22b), which maintain contact with the seabed and serve to provide protection to the attachment point of bridle and trawl.

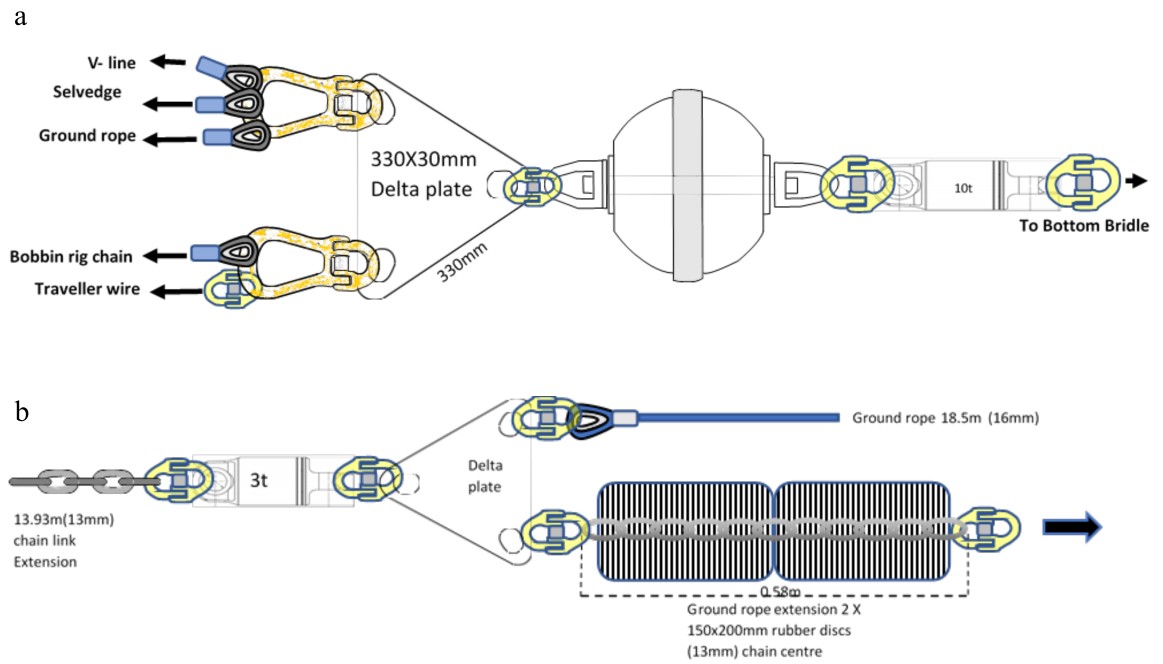


Figure 22: Example of ground gear extension: (a) Dan Leno or roller bobbin arrangement ahead of the main ground gear, and (b) on a high opening bottom trawl using rubber disks and chain (image source: NIWA).

3.1.5.2 Traveller/false fishing wires

Traveller or false fishing wires allow for lateral movement of the ground gear components resulting in improved overall alignment of the ground rig and trawl net (Figure 18c & 23).

Figure 23: False fishing or traveller wires allow better alignment of ground gear components (image source: NIWA).

3.1.5.3 Twin-rig trawl clump

When a twin/double trawl rig (twin rigging) is used for bottom fishing, a weight is used to achieve bottom contact of the front part of the inner sweeps/bridles that are located between the two trawl nets (Figure 24). This weight might be heavier than the weight of a trawl door (normally 30% heavier in twin-rig trawls). The weights differ in shape and rigging, and their effect on the bottom will vary (Grieve et al. 2015).

Figure 24: (a) Roller clump and (b) weighted clump which are used as part of bottom gear found on twin-rig trawls (see also Figure 3a. which shows twin-rig arrangement) (image source: with permission from Seafish).

3.1.6 Categorisation of bottom trawling gear types

The gear types and descriptions provided in the previous sections represent the full range and variability of mobile bottom fishing gear used in the New Zealand inshore and offshore fisheries, based on detailed examinations by observers, reports and publications, and examination of net plans. However, assignation of every reported commercial trawl in Fisheries New Zealand databases to a specific gear configuration is limited by the lack of gear details that are required to be reported. To enable contact and impact measures to be assigned to all commercial fishing trawl records, door spread values and ground gear types could only be determined from a limited set of recorded parameters: vessel length, vessel type (nationality), and target species.

Therefore, ground gear types were assigned to commercial fishing events based on the most observed type within existing categories (Baird & Mules 2021) defined by recorded parameters. Generally, bottom trawl ground ropes on vessels less than 28 m, scampi vessels (which can be up to 46 m), and larger domestic vessels (up to 82 m) not targeting the main middle-depth fish species are composed of rubber discs. Bottom trawl rockhopper rigs are mostly used on vessels 28–46 m not targeting scampi, large (>82 m) domestic vessels not targeting the main middle-depth fish species, and vessels targeting orange roughy and oreos on UTFs. Bottom trawl bobbin rigs are used by all vessels of 46–82 m and vessels of any length targeting orange roughy and oreos on slope areas. Chain is used only on bottom trawls by BATM vessels, and a combination of wire and chain is used on all midwater trawl gear.

We used a combination of the gear configurations reported here and the existing categories derived from research into trawl footprint assessment (Baird & Mules 2021) to produce a set of 24 gear categories that represent all types of trawl fishing conducted within the TS/EEZ (Table 5). These categories can be assigned to all commercial trawls and include values for door spread and ground gear type that can be used in the determination of benthic impacts. Notable extensions to the categories of Baird & Mules (2021) are the separation of tows into bottom and midwater, and the inclusion of the parameter habitat type, derived from tow positions and known UTF locations, to differentiate UTF tows from ‘slope’ (i.e., non-UTF) tows for orange roughy/oreo fisheries.

There are relatively few gear categories for smaller vessels, despite the broad range of gear configurations known to be used in inshore fisheries, partly because of the lower level of gear details, including from observers, available from these vessels, and partly because all inshore vessels are considered domestic.

As with Baird & Mules (2021), dredge fishing is not included due to the lack of sufficient spatial resolution in historical catch and effort data records.

Table 5: Gear categories (GC) and their definitions, as used to differentiate the effective width (door spread, m) of the gear in contact with the seafloor. FOV, Foreign-owned vessel; DOM, Domestic vessel; LDOM, Large Domestic vessel; BATM, Bolshoy Avtonomniy Trawler Morozilniy.

GC	Vessel length (m)	Vessel type	No. nets	Method	Target species†	Door spread	Habitat type	Ground gear type
GC1	<20	All	1	BT	All	70	SLOPE	Rubber discs
GC2	20–28	All	1	BT	All	100	SLOPE	Rubber discs
GC3	28–46	All	1	BT	All	150	SLOPE	Rockhopper
GC4	<28	All	2	BT	SCI	50	SLOPE	Rubber discs
GC5	<28	All	3	BT	SCI	70	SLOPE	Rubber discs
GC6	28–46	All	2	BT	SCI	70	SLOPE	Rubber discs
GC7	28–46	All	3	BT	SCI	90	SLOPE	Rubber discs
GC8	46–82	FOV/DOM	1	BT	All other	150	SLOPE	Bobbin rig
GC9	46–82	DOM	1	BT	HAK, HOK, LIN, SWA	200	SLOPE	Bobbin rig
GC10	>82	DOM	1	BT	All	200	SLOPE	Rockhopper
GC11	104 m	BATM	1	BT	All	150	SLOPE	Chain
GC12	<82	LDOM	2	BT	All	400	SLOPE	Rubber discs
GC13	46–82	DOM	2	BT	HAK, HOK, LIN, SWA	400	SLOPE	Bobbin rig
GC14	All	All	1	BT	ORH, OEO	135	SLOPE	Bobbins
GC15	All	All	1	BT	ORH, OEO	110	UTF	Rockhopper
GC16*	All	All	Any	BT	All other	70	All	Rubber discs
GC17	<20	All	1	MW	All	135	SLOPE	Wire/chain
GC18	20–28	All	1	MW	All	192	SLOPE	Wire/chain
GC19	28–46	All	1	MW	All	286	SLOPE	Wire/chain
GC20	46–82	FOV/DOM	1	MW	All	286	SLOPE	Wire/chain
GC21	All	DOM	1	MW	HAK, HOK, LIN, SWA	380	SLOPE	Wire/chain
GC22	>82	DOM	1	MW	All	380	SLOPE	Wire/chain
GC23	104	BATM	1	MW	All	286	SLOPE	Wire/chain
GC24*	All	All	Any	MW	All other	135	All	Wire/chain

* GC16 and GC24 are default categories for trawls that do not fit any of the listed definitions.
† SCI, scampi; HAK, hake; HOK, hoki; LIN, ling; SWA, silver warehou; ORH, orange roughy; OEO, oreo species.

3.2 Spatial and temporal extent of bottom contact by different fishing gear configurations

There were eight respondents to the Delphi survey used to estimate effective trawl widths to provide values for each gear category in Table 5. Respondents varied in their estimates of the percentage of time each component of midwater trawl (as defined by Baird & Mules 2021) gear spent in contact with the seafloor (Appendix 2). Estimates for doors/wing-end weights and sweeps/bridles were overall similar, ranging from 0 to 25% for the former and 0–35% for the latter, with an average of 10% for both. For the ground gear, estimates ranged from 5 to 35%, with an average of 26%. Consensus was that such midwater trawl components were not frequently in contact with the seafloor, especially doors/wing-end weights, and sweeps/bridles (Table 6).

Table 6: Summary of responses (mean values) from a Delphi survey of 8 NIWA experts on the percentage time three separate components of midwater trawls towed near the bottom are in contact with the seafloor.

Gear component	Percent of time in bottom contact
Doors /wing-end weights	10
Sweep/bridle	10
Ground gear	26

The effective door spread for each gear category is given in Table 7. The resulting estimated total footprint is dominated by three main bottom trawl gear categories, GC1 (all vessels < 20 m, single net, any target species), GC8 (foreign owned or domestic vessels 46–82 m, not targeting hake, hoki, ling, or silver warehou), and GC9 (domestic vessels 46–82 m, targeting hake, hoki, ling, or silver warehou) (Table 7). Together these gear categories accounted for about 56% of the total footprint. The only other gear categories contributing more than 5% of the total footprint were GC2, GC3, and GC 13 (see Table 5 for gear category details). The eight midwater gear categories combined (GC17 to GC24) accounted for only 5.6% of the total footprint.

Table 7: Door spread (m), effective door spread (m), area (km^2), and percentage of the total footprint (all years, 1989–90 to 2018–19, combined) represented by individual gear categories (GC), based on tow polygons defined by start/finish positions and effective trawl widths (see Table 5 for gear category definitions).

Gear category	Door spread	Effective Door spread	Footprint area (km^2)	Percentage of total footprint
GC1	70	58	567 604	16.519
GC2	100	83	310 947	9.050
GC3	150	125	284 691	8.286
GC4	50	42	85 744	2.495
GC5	70	58	61 095	1.778
GC6	70	58	12 529	0.365
GC7	90	75	12 764	0.371
GC8	150	125	720 188	20.960
GC9	200	166	658 051	19.152
GC10	200	166	161 980	4.714
GC11	150	125	2 444	0.071
GC12	400	332	960	0.028
GC13	400	332	226 708	6.598
GC14	135	112	127 511	3.711
GC15	110	28	2 776	0.081
GC16	70	58	6 219	0.181
GC17	135	16	134	0.004
GC18	192	23	1 185	0.034
GC19	286	35	8 924	0.260
GC20	286	35	40 460	1.178
GC21	380	46	4 362	0.127
GC22	380	46	59 519	1.732
GC23	286	35	79 143	2.303
GC24	135	16	33	0.001

Maps representing the cumulative proportional footprint and total swept area of bottom contact are shown in Figure 25 for all gear categories and all years combined.

The cumulative proportional footprint (as defined above and by Mormede et al. 2017) for the period 2007–08 to 2018–19 (inshore fisheries) and 1989–90 to 2018–19 (deepwater fisheries) shows values approaching 1 (=100% contact/cell) for almost all of the coastal regions surrounding New Zealand (with the notable exception of the Fiordland coast in the southwest of the South Island), much of the Chatham Rise, large areas of the central west including the Challenger Plateau, and all the major fishing grounds of the sub-Antarctic plateaus.

The equivalent map for total swept area shows a range in values from 0 to a maximum of over 250 (equal to 250 square km trawled per 1 km cell) and from this the regions of the very highest fishing intensity can be seen. These can be seen in Figure 25 in (depth) bands along the west coast of the South Island, the western and central Chatham Rise, and around the margins of the Stewart-Snares shelf and western Campbell Plateau.

For maps representing the cumulative proportional footprint and total swept area separately for each gear category, refer to Appendix 3 and Appendix 4, respectively.

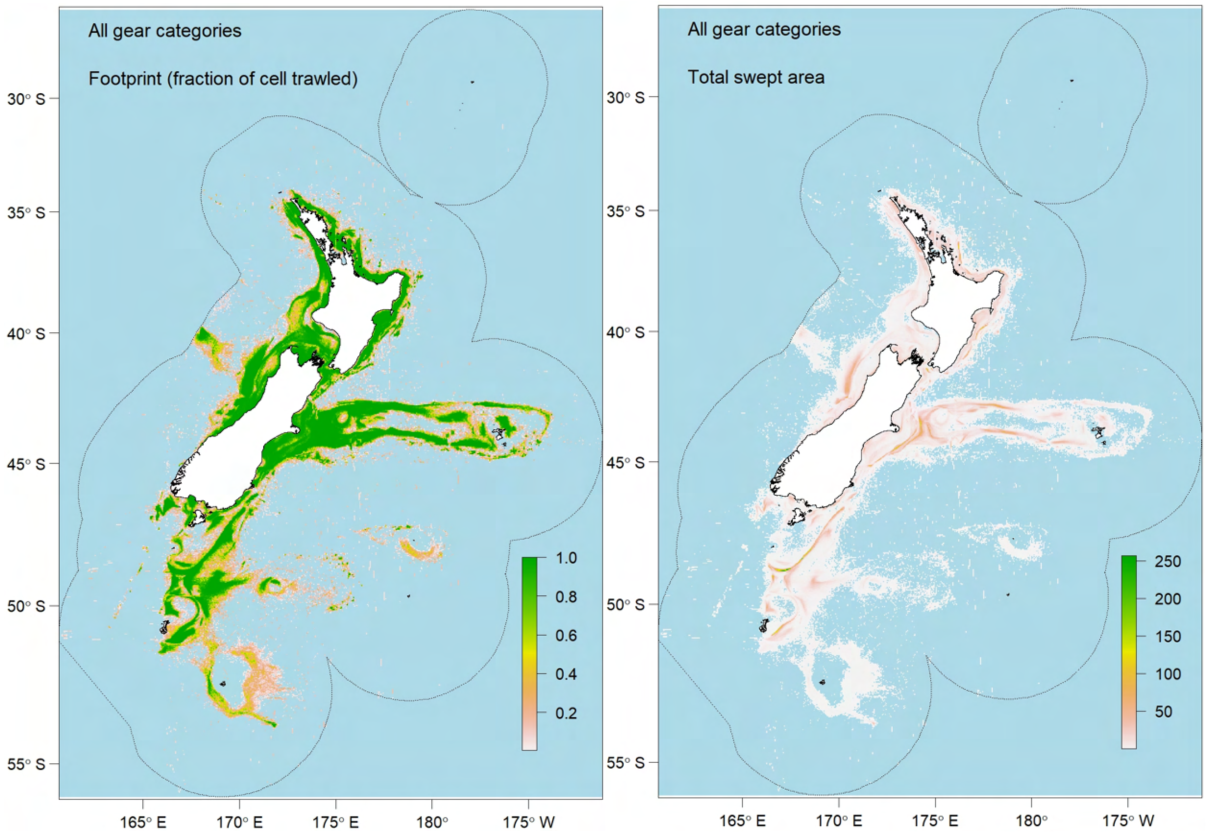


Figure 25: Spatial extent of bottom contact by all trawl gear categories for the period 2007–08 to 2018–19 (inshore fisheries) and 1989–90 to 2018–19 (deepwater fisheries), within the New Zealand TS/EEZ. Left, cumulative proportional footprint; right, total swept area. Map resolution (cell size) is 1 × 1 km.

3.3 Impacts of different gear configurations on key benthic taxa

3.3.1 MSRP for TS and EEZ

Fifteen respondents to the Delphi survey provided an estimate of the percent impact (i.e., the percent damaged or destroyed) from the passing of various ground gear configurations over the three faunal categories (large, erect, hard, sessile (LEHS); small, flexible/encrusting (SFE); deep-burrowing infauna (DBI)) (Appendix 5). The means of these estimates were then used to calculate MSRP impacts for each gear/faunal category. Estimated impacts were greatest for LEHS fauna (82–90%) and least for DBI fauna (7–9%), with intermediate values for SFE fauna (39–54%). Relative impact by gear type varied among faunal category, with bobbin and rockhopper rigs considered more damaging to LEHS fauna, but chain and wire ground gear more damaging to SFE fauna (Table 8). The impact value associated with each gear and faunal category is shown in Table 9.

Table 8: Summary of responses (mean values) from a Delphi survey of 15 NIWA experts on impact values (% mortality) from the passage of various ground gear types across three categories of benthic invertebrate fauna. LEHS, Large, erect, hard, sessile; SFE, Small, flexible/encrusting; DBI, Deep-burrowing infauna.

	Faunal category		
Ground gear	LEHS	SFE	DBI
Bobbin	90	40	9
Chain	85	54	9
Rockhopper	89	39	8
Rubber discs (Cookies)	87	44	7
Wire	82	53	9
Total	86	46	8

Table 9: Gear categories (GC) and the separate mortality percentages (the "Impact index" of Mormede et al. 2017) on three categories of benthic fauna (LEHS = Large, erect, hard, sessile; SFE = Small, fragile, encrusting; DBI = Deep, burrowing infauna (sensu Mormede et al. 2017)) caused by the passing of the various ground gear types (see Table 5 for gear category definitions).

Gear category	Impact (%)		
GGC	LEHS	SFE	DBI
GC1	87	44	7
GC2	87	44	7
GC3	89	39	8
GC4	87	44	7
GC5	87	44	7
GC6	87	44	7
GC7	87	44	7
GC8	90	40	9
GC9	90	40	9
GC10	89	39	8
GC11	85	54	9
GC12	87	44	7
GC13	90	40	9
GC14	90	40	9
GC15	89	39	8
GC16	87	44	7
GC17	82	53	9
GC18	82	53	9
GC19	82	53	9
GC20	82	53	9
GC21	82	53	9
GC22	82	53	9
GC23	82	53	9
GC24	87	44	7

The outputs of the MSRP calculations are expressed on a scale of 0–1, where 0 = completely impacted and 1 = completely unimpacted. **Note that values represent the 'potential' impact only and will only have effect in cells where the fauna are actually found, as may be estimated using species distribution models.** Considering the combined impact from all gear types, MSRP values are lowest overall for LEHS fauna, a result of the high impact values estimated for this category (Figure 26). This impact is most evident on the highly fished areas of the Chatham Rise, the Stewart-Snares shelf, north-western Campbell Plateau, and much of the near coast regions of both main islands. A similar spatial pattern of impact is shown for SFE fauna but with slightly higher MSRP values overall. MSRP for DBI is high in most areas, due to the protection afforded by their infaunal habitat, with lower levels (less than about 0.5) occurring in relatively small areas of the Chatham Rise, southern plateaus, the west coast of the South Island, and inshore parts of the North Island. A complete set of maps for MSRP impact, by gear and faunal category, is given in Appendix 6.

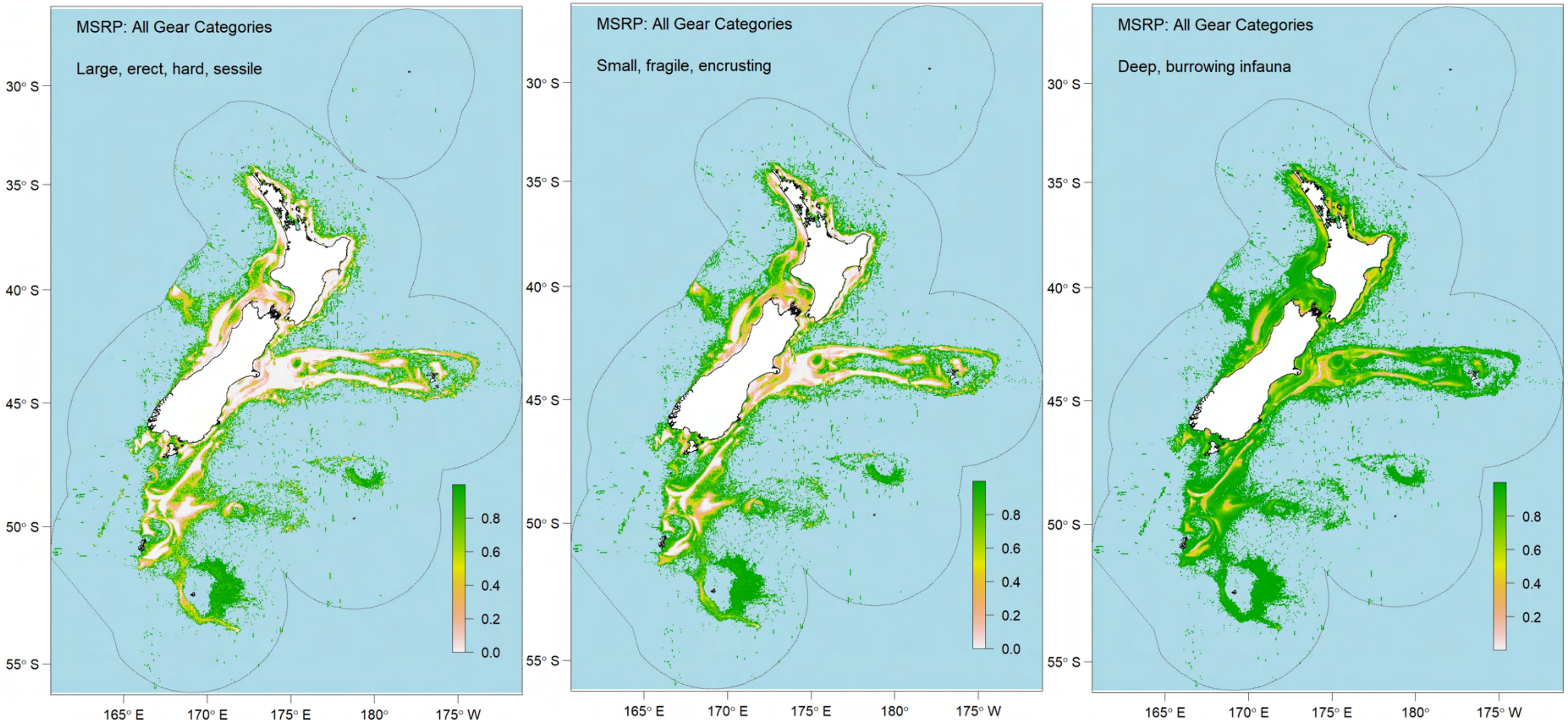


Figure 26: MSRP benthic impact estimates, as relevant to separate benthic faunal categories. Note that these maps do not include any information on the actual distribution of the benthic faunal categories. The range in values is 0–1, where 0=completely impacted and 1=unimpacted, for three categories on benthic fauna within the New Zealand TS/EEZ, based on the MSRP method (Mormede et al. 2017). See Table 8 for the mortality values behind the differences in these plots.

3.3.2 RBS for TS and EEZ

The outputs of the RBS calculations are expressed on a scale of 0–1, where a value of 1 represents the untrawled status and a value of 0 represents fully depleted status.

Depletion and recovery values are identical for the four stony corals assessed, and similarly for the two sponge groups, therefore single maps were produced to illustrate RBS for these taxa (Figure 27). Differences in RBS among the taxa represented by the seven unique sets of d and R values are subtle, but a general graduation can be seen in overall status from most depleted (stony corals, with the highest d and lowest R values) to least depleted (Pennatulacea, with the lowest d and highest R values) (see Table 2 for d and R values). The spatial patterns of status across the study area are strongly driven by fishing intensity which, although represented differently to that used in the MSRP method, results in a strong similarity of benthic status between the two methods. The areas showing the most depleted state for RBS are again on the Chatham Rise, off the north and east coasts of the North Island, the west coast of the South Island, and various regions of the southern plateaus.

The differences in status estimated using the best case and worst case sensitivities are illustrated for the most depleted taxa (stony corals) and least depleted taxa (Pennatulacea) in Figure 28. Strong differences are evident between best- and worst-case status, for both taxa, most clearly shown at the western end of the Chatham Rise and east coast South Island for stony corals, and off the east coast of the North Island for sea pens. Maps of RBS status for the remaining taxa, with sensitivities, are shown in Appendix 6.

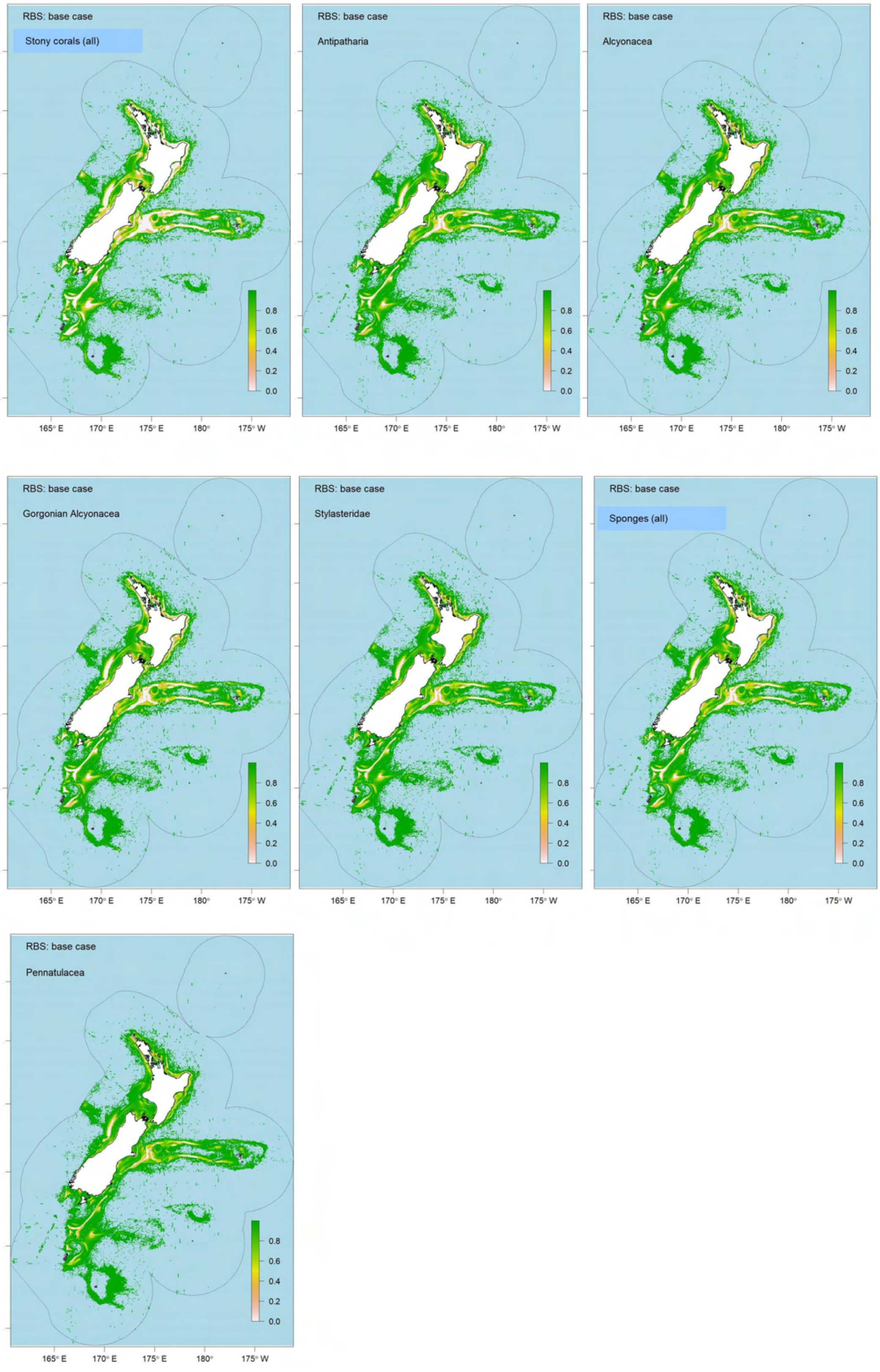


Figure 27: Benthic impact estimates, expressed as Relative Benthic Status (range 0–1, where 0=fully depleted status and 1=untrawled status) for vulnerable marine ecosystem indicator taxa, within the New Zealand TS/EEZ, based on the RBS method (Pitcher et al. 2017). Note that these maps do not include any information on the actual distribution of the VME indicator taxa. Single figures are shown to represent each of the four stony coral species and the two sponge taxa, as *d* and *R* values do not vary within these groups.

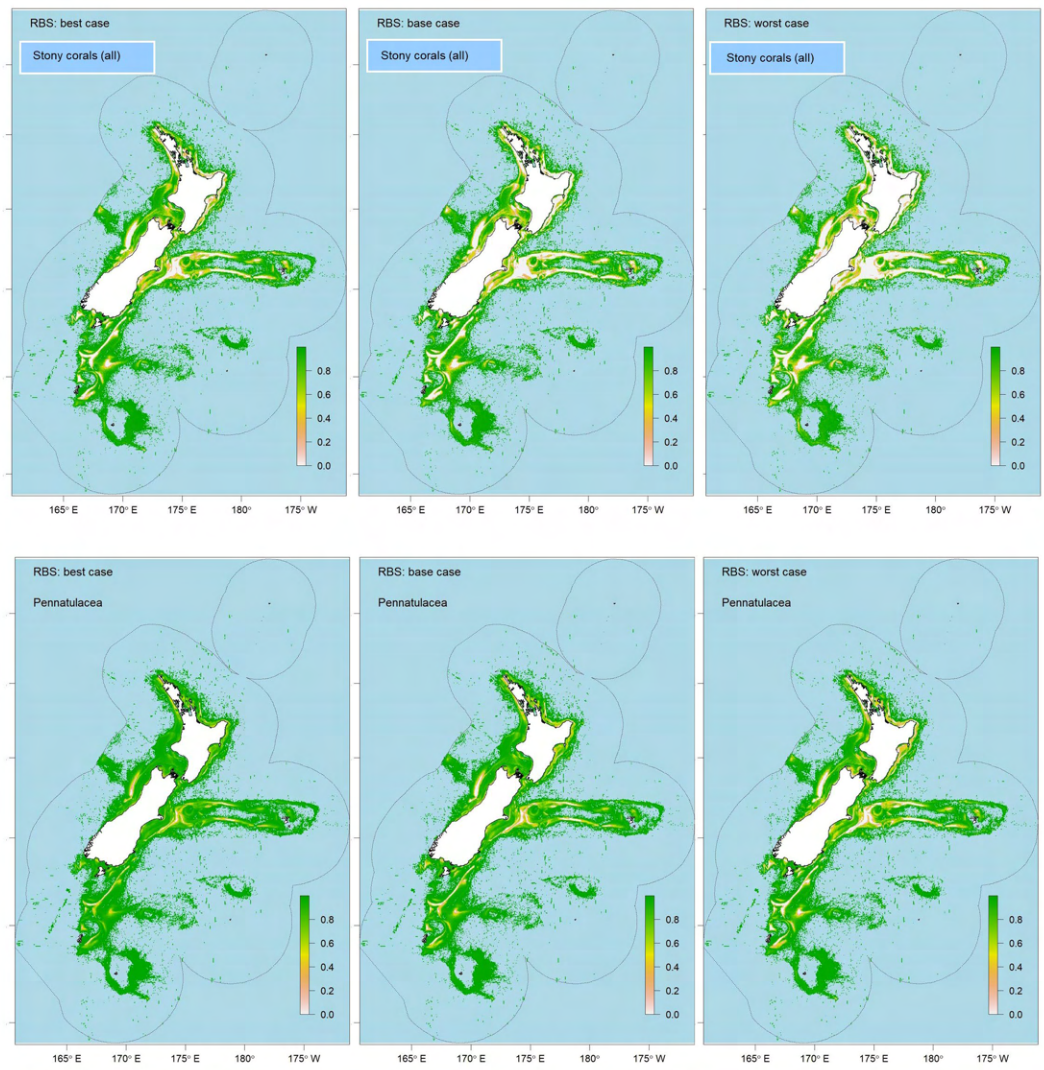


Figure 28: Relative Benthic Status. Sensitivities (best, base, and worst cases as determined from values in Table 2) for the most depleted vulnerable marine ecosystem taxa (stony corals, top) and least depleted taxa (sea pens, bottom), within the New Zealand TS/EEZ, based on the RBS method (Pitcher et al. 2017). Note that these maps do not include any information on the actual distribution of the VME indicator taxa.

3.3.3 Modified RBS-type approach for Chatham Rise

3.3.3.1 Simulation results

When fishing is proportional or inversely proportional to the biomass of the focal taxa, the resulting fishing pattern is patchy and includes areas that receive high fishing effort (and where the corresponding taxa will be more strongly depleted), as well as areas of low or no fishing (Figure 29). Simulated observed catch either reflected observation error assumptions only, or observation error and bias from non-detection of small interactions (i.e., not all interactions result in observed catch; Figure 30). Tracks were simulated for all three effort distribution patterns (proportional, inversely proportional, or randomly with respect to the carrying capacity *K*), and corresponding depletion patterns are shown in Figure 31. Effort distributions that are patchy leave large areas untouched, whereas the completely random distribution leaves no refuges and causes the spatial domain to be nearly depleted by the end of the simulation (Figure 32).

Corresponding biomass trends confirm the differential depletion of certain pixels depending on the relative effort they receive (Figure 33). Given the stochastic nature of the trawl tracks, individual pixels show occasional signs of slow recovery during periods of relatively low effort. A challenge with the aggregated records for estimating depletion and recovery rates becomes obvious at the aggregated grid scale: trends are far smoother and periods of evident recovery are relatively rare (Figure 34).

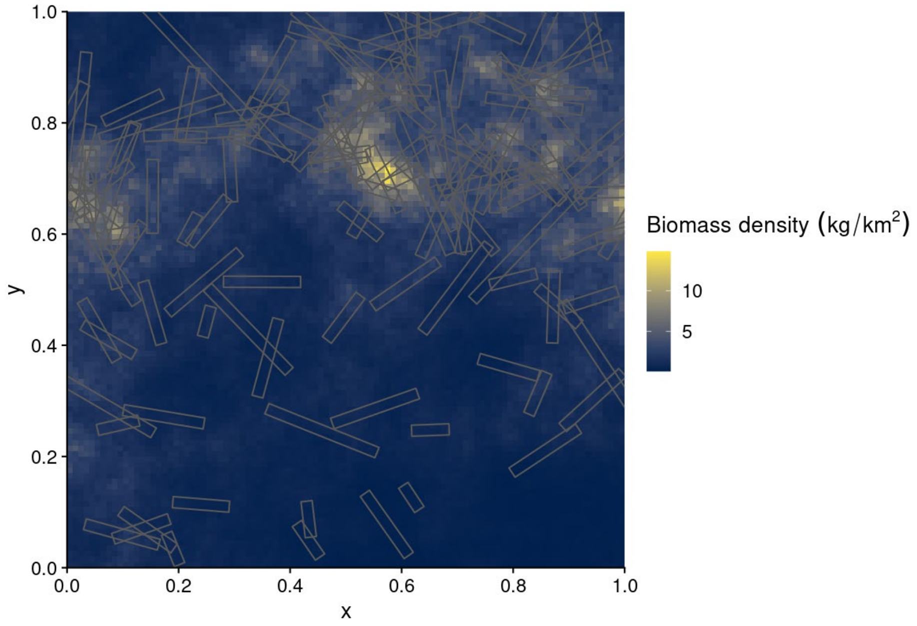


Figure 29: Simulated biomass density of the focal bycatch taxon, and random tow tracks, drawn according to the spatial distribution of the focal taxon.

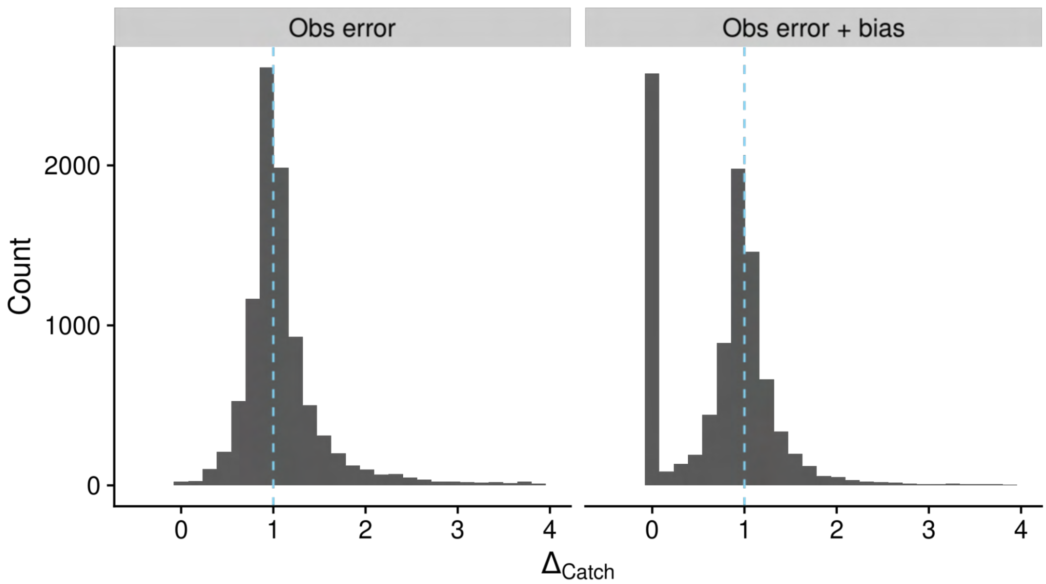


Figure 30: Simulated catch relative to true catch (vertical blue line) at the grid scale across simulations, for scenarios with observation error only, and additional bias due to non-detection of small amounts of catch.

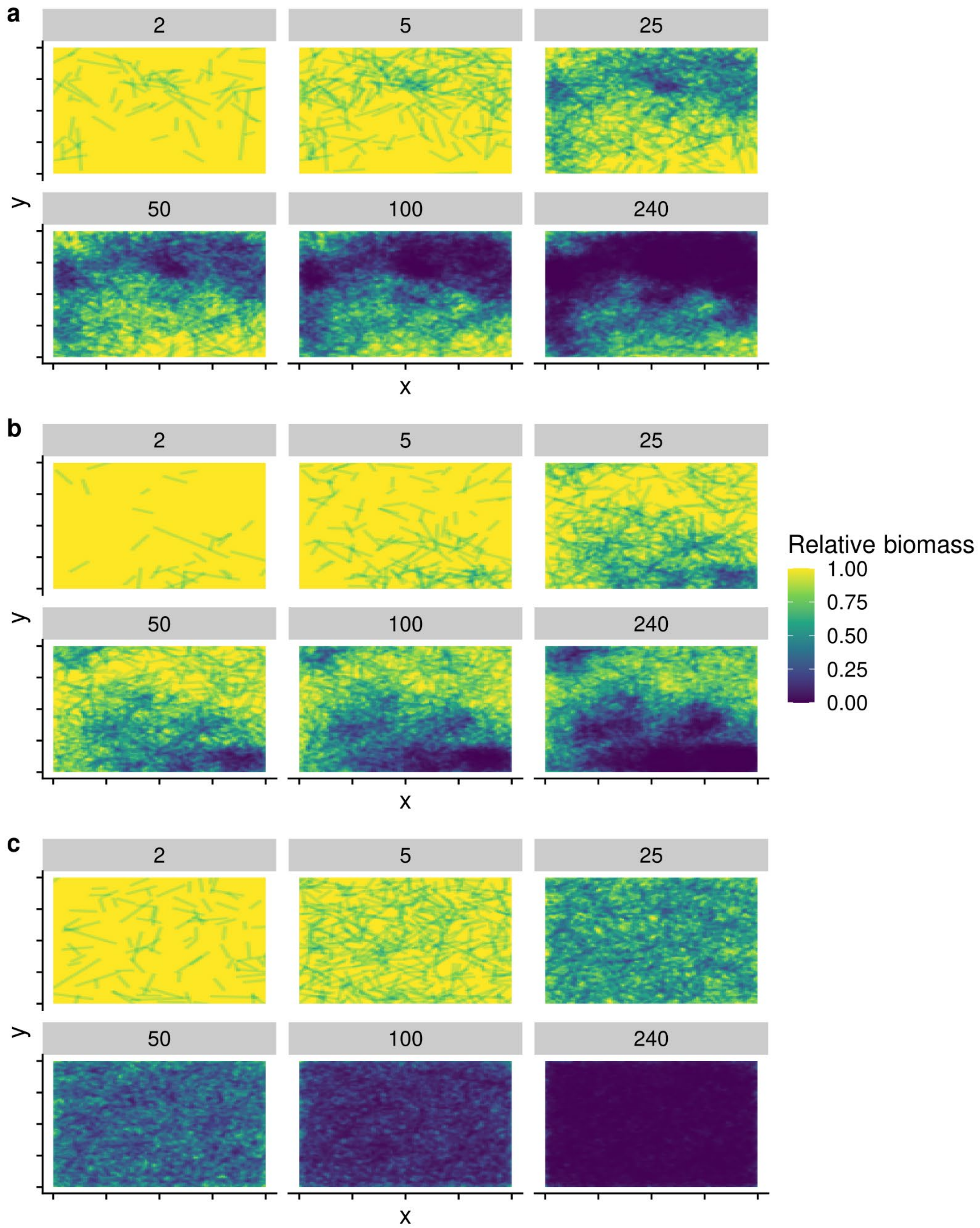


Figure 31: Simulated biomass density of the focal bycatch taxon for selected time steps (headers) after application of removals from randomly placed tows for three effort distribution scenarios: a) effort proportional to, b) inversely proportional to, and c) randomly placed with respect to equilibrium biomass density *K*. The latter is depicted in 29 and used for all effort scenarios.

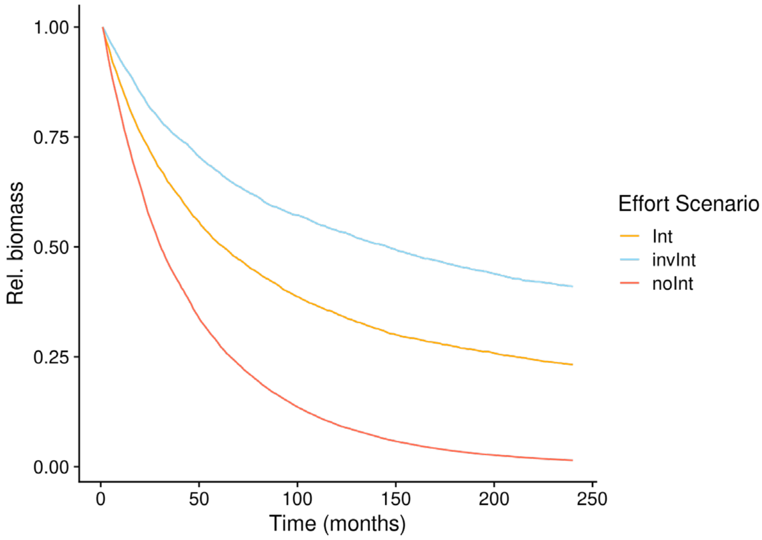


Figure 32: Simulated relative biomass trends over 240 time steps after aggregation of pixel-level trends across the complete simulation domain (10 000 pixels) for three effort distribution scenarios: a) (Int) effort proportional to, b) (InvInt) inversely proportional to, and c) (noInt) randomly placed with respect to equilibrium biomass density *K*.

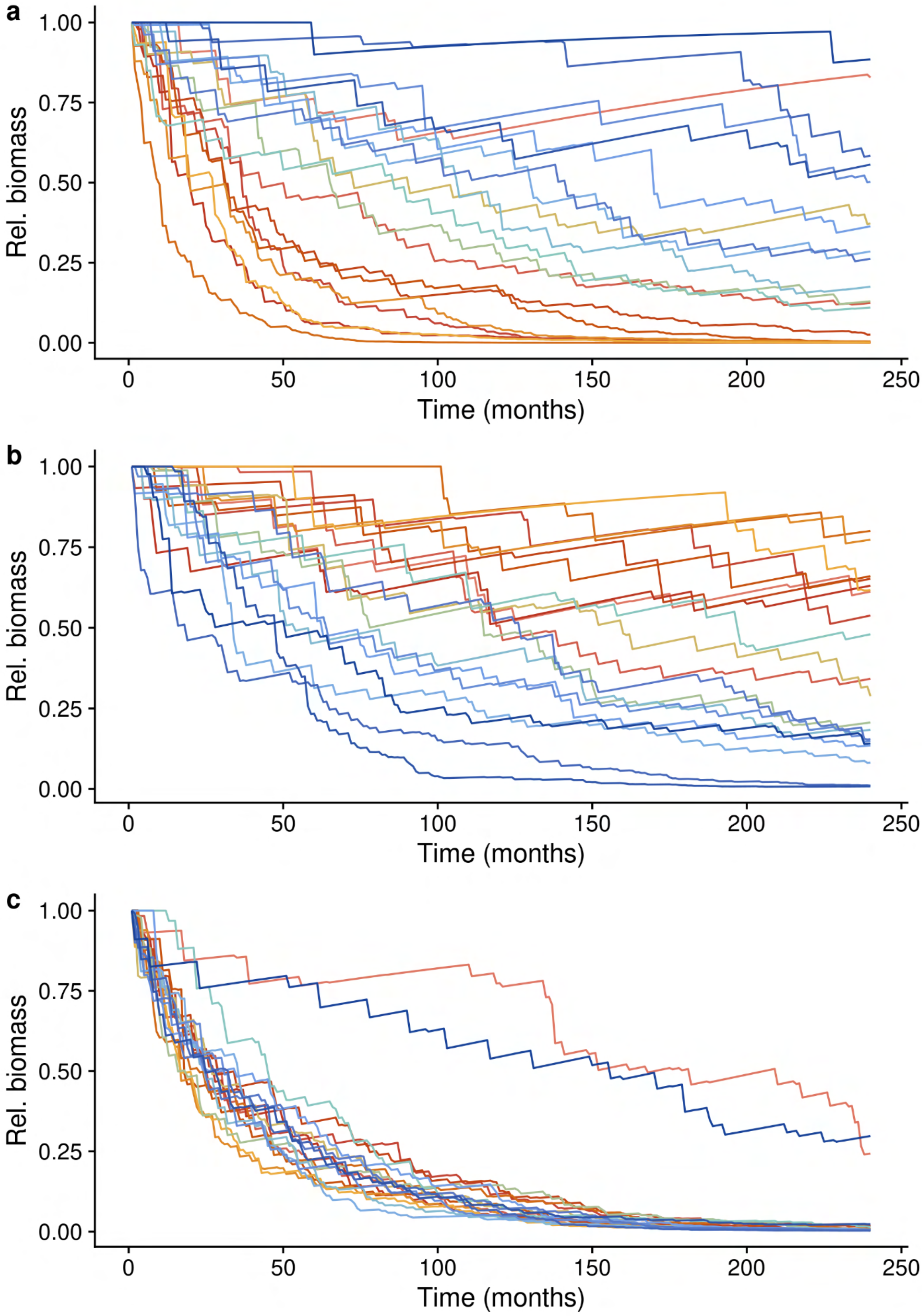


Figure 33: Simulated relative biomass trends over 240 time steps for 20 evenly spaced pixels (coloured lines) after application of removals from randomly placed tows for three effort distribution scenarios: a) effort proportional to, b) inversely proportional to, and c) randomly placed with respect to equilibrium biomass density *K*.

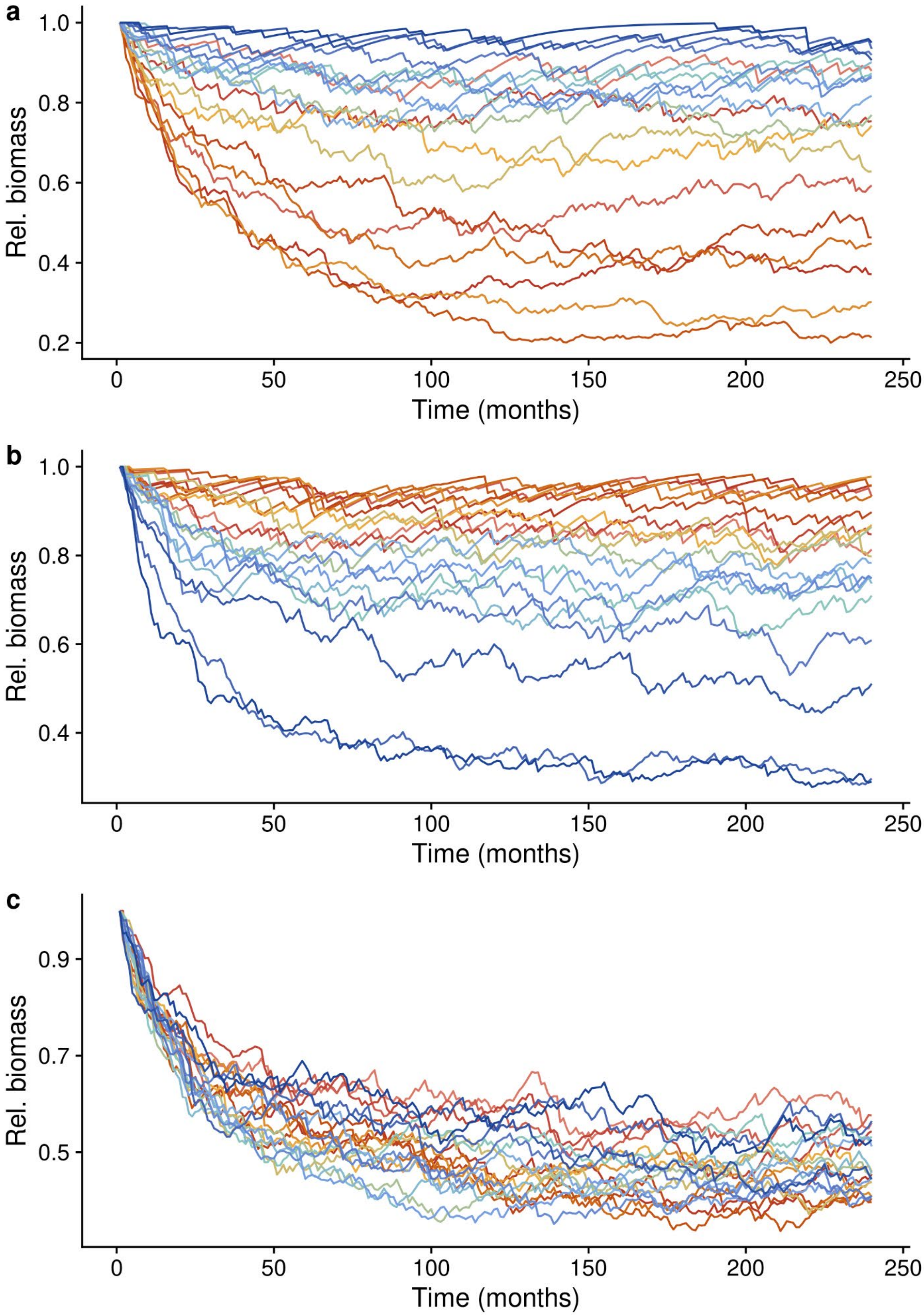


Figure 34: Simulated aggregated relative biomass trends over 240 time steps for 20 evenly spaced grid cells (coloured lines) after aggregation of pixel level trends for three effort distribution scenarios: a) effort proportional to, b) inversely proportional to, and c) randomly placed with respect to equilibrium biomass density *K*. Each grid cell contains 100 pixels.

3.3.3.2 Estimating RBS parameters from simulated data

For the statistical model run on simulated data, it was possible to compare estimated current benthic status (i.e., the relative depletion level in space) to the known value from the simulated population. While the model estimated the current benthic status relatively well (Figure 35), models fitted to partial time series only, where temporal trends were less obvious, showed bias towards overestimation of depletion levels, whereas models run over the full time series were biased slightly low (Figure 36). This bias arose from an overestimate (or underestimate) of productivity in these runs (Figure 37), which was associated with poor estimates of equilibrium biomass (Figure 38).

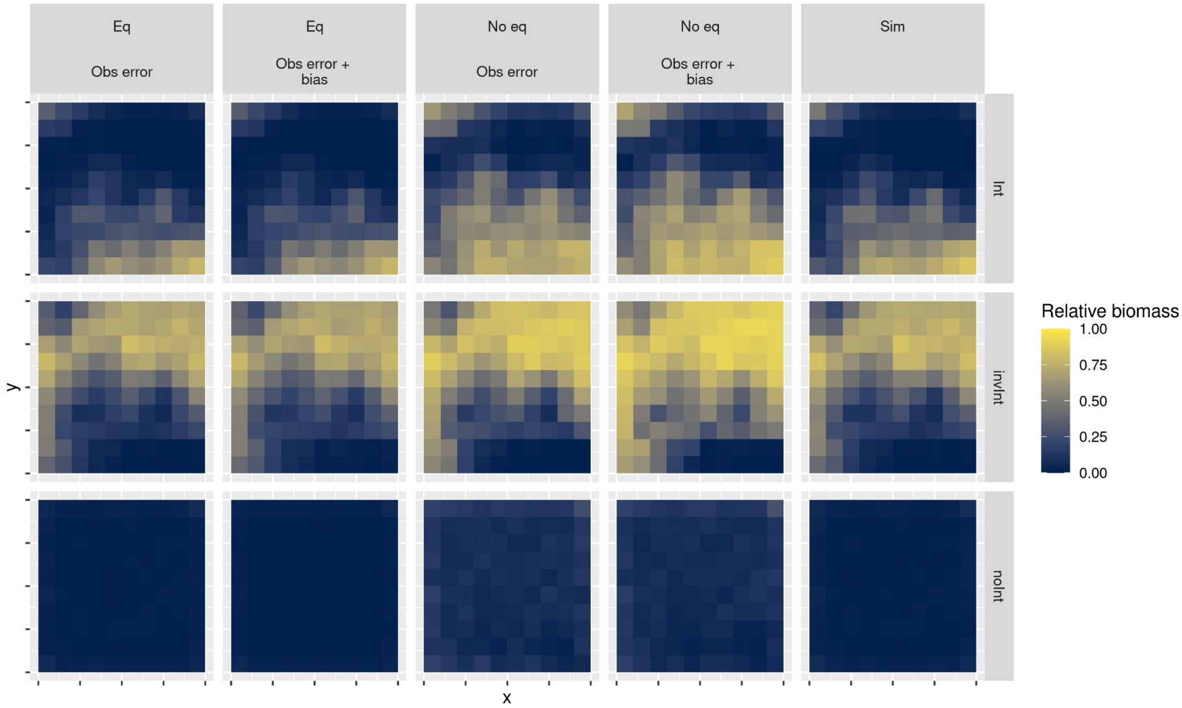


Figure 35: Estimated (posterior mean; y-axis) relative to simulated (x-axis) depletion level (biomass relative to equilibrium biomass) for models fitted to full (Eq) or partial (No eq - fitted to the second half of the time series only) datasets, with observation error only or added observation bias for small quantities, for effort scenarios (rows) where fishing effort was either proportional to (Int), inversely proportional to (Inv-Int), or independent of initial density.

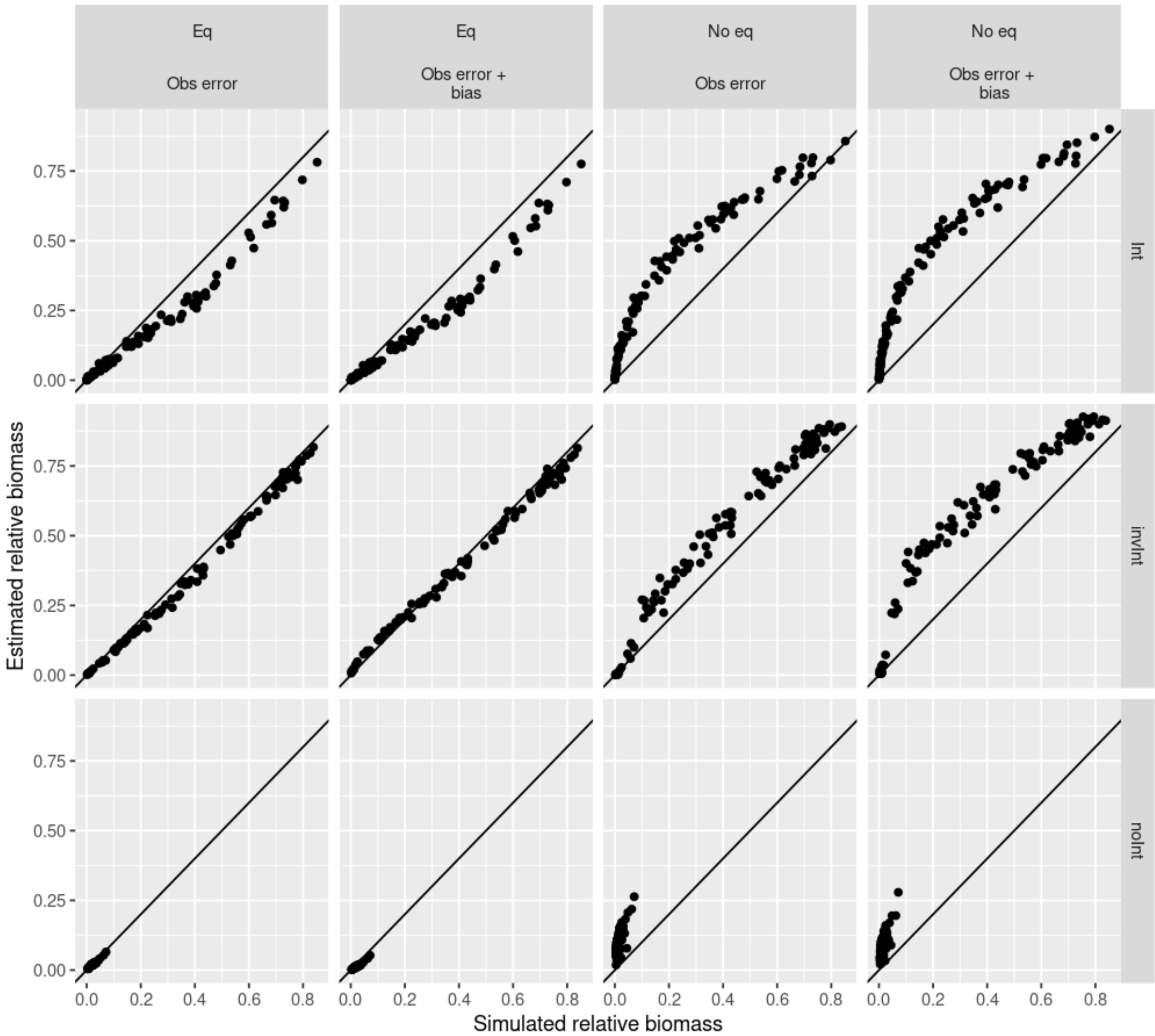


Figure 36: Estimated (posterior mean; y-axis) versus simulated (x-axis) depletion level (biomass relative to equilibrium biomass) for models fitted to full (Eq) or partial (No eq) datasets, with observation error only or added observation bias for small quantities, for effort scenarios (rows) where fishing effort was either proportional to (Int), inversely proportional to (Inv-Int), or independent of initial density.

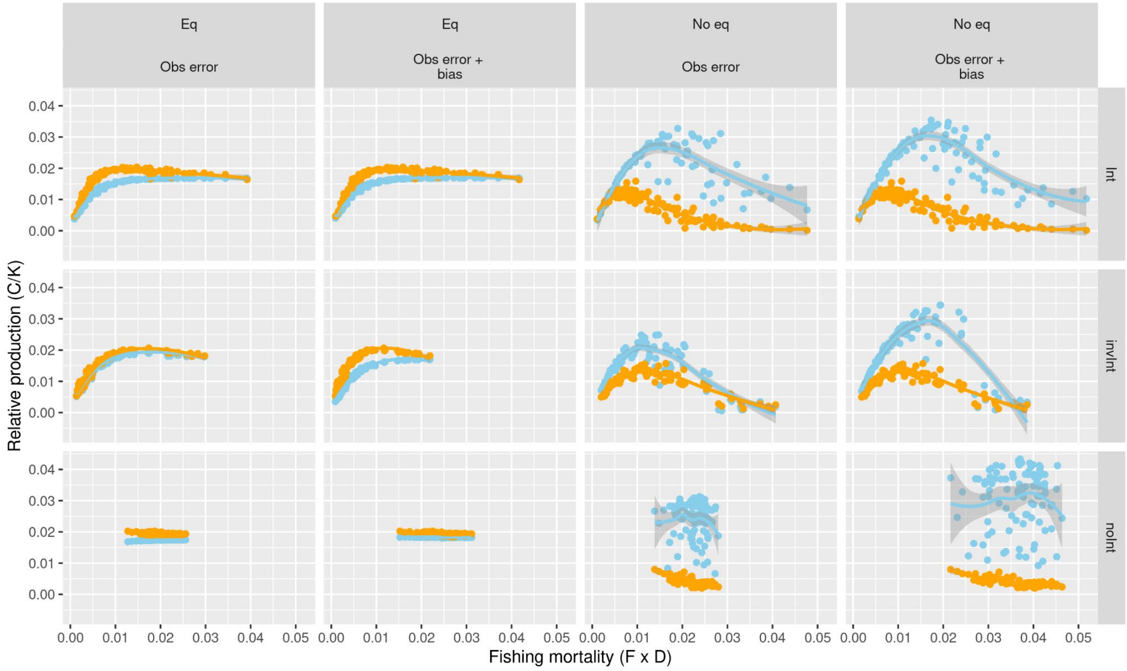


Figure 37: Estimated (posterior mean; blue) versus simulated (orange) production relative to carrying capacity as a function of fishing mortality for models fitted to full (Eq) or partial (No eq) datasets, with observation error only or added observation bias for small quantities, for effort scenarios (rows) where fishing effort was either proportional to (Int), inversely proportional to (Inv-Int), or independent of initial density.

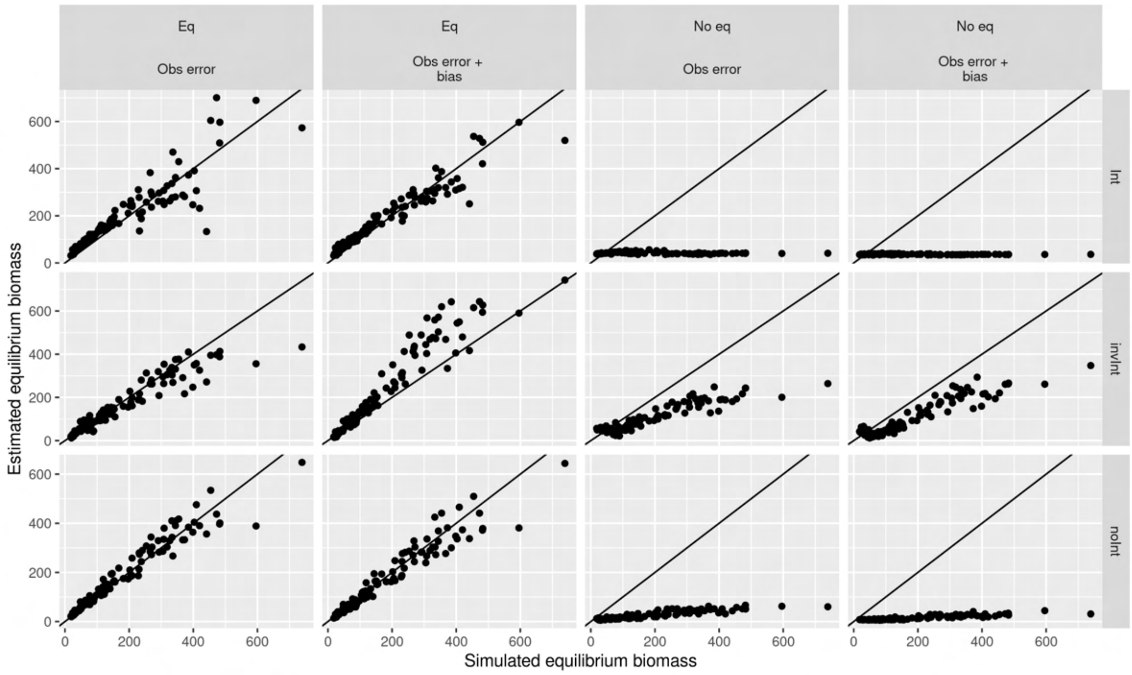


Figure 38: Estimated (posterior mean; y-axis) versus simulated (x-axis) equilibrium biomass for models fitted to full (Eq) or partial (No eq) datasets, with observation error only or added observation bias for small quantities, for effort scenarios (rows) where fishing effort was either proportional to (Int), inversely proportional to (Inv-Int), or independent of initial density.

3.3.3.3 Application to observed bycatch of key benthic taxa on the Chatham Rise

The spatial Schaefer surplus estimation model was run for input data from the hoki survey area on the Chatham Rise (Figures 39 and 40) for four key benthic taxa.

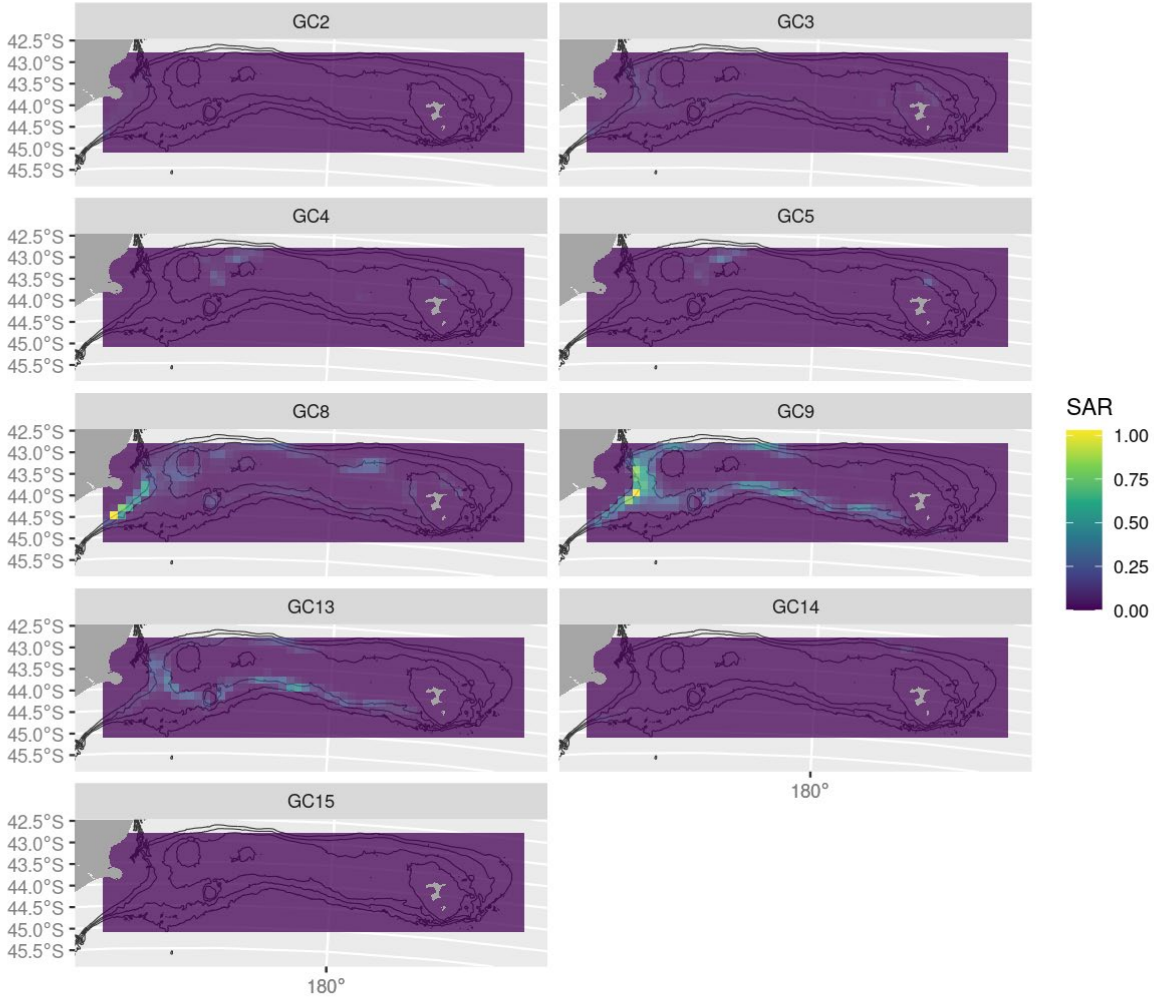


Figure 39: Swept area ratio (SAR) for retained gear categories (GC) used to estimate catches in the spatial Schaefer surplus production model. Gear categories are defined in Table 5.

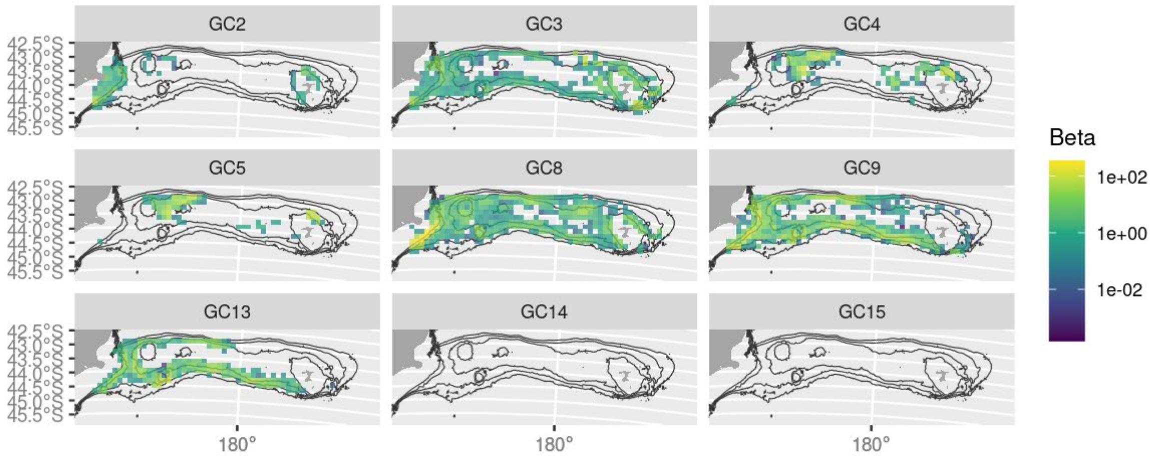


Figure 40: Spatial effort concentration within model grid cells as measured by the negative binomial dispersion parameter beta. Gear categories are defined in Table 5.

Anthozoa

Catch of Anthozoa was highest to the west of Chatham Rise, largely by gear categories 8, 9, and 13 (Figure 41). Model convergence was satisfactory (Figure 42), with parameter estimates showing substantial differences in estimated fractional depletion parameters (d_G) for the 9 different gear types considered in the analysis (Table 10). Gear categories 9 and 13 were estimated to have values of $d_G <$ 0.1, whereas other gears were closer to the prior expectation, with high d_G values estimated for fisheries with ORH/OEO targets fishing on features (GC15), as well as for scampi fisheries (GC4 and GC5). The model fitted catches reasonably well without apparent bias (Figure 43).

Estimated depletion was generally low except for areas of high effort (Figure 44a). Some areas of high effort were estimated to have been more substantially depleted, but these were also areas with higher uncertainty (Figure 44b). Estimated unfished biomass levels varied relatively smoothly across the rise, with high equilibrium biomass estimated at the shelf-break of the Canterbury Bight and Banks Peninsula (Figure 44c). RBS was estimated at or below current status (Figure 44d), especially in areas of high effort, suggesting that populations are not at equilibrium with fishing effort. Partitioning the impact into local vulnerability and depletion rates highlighted the importance of spatial effort concentration in determining local RBS values (Figure 45).

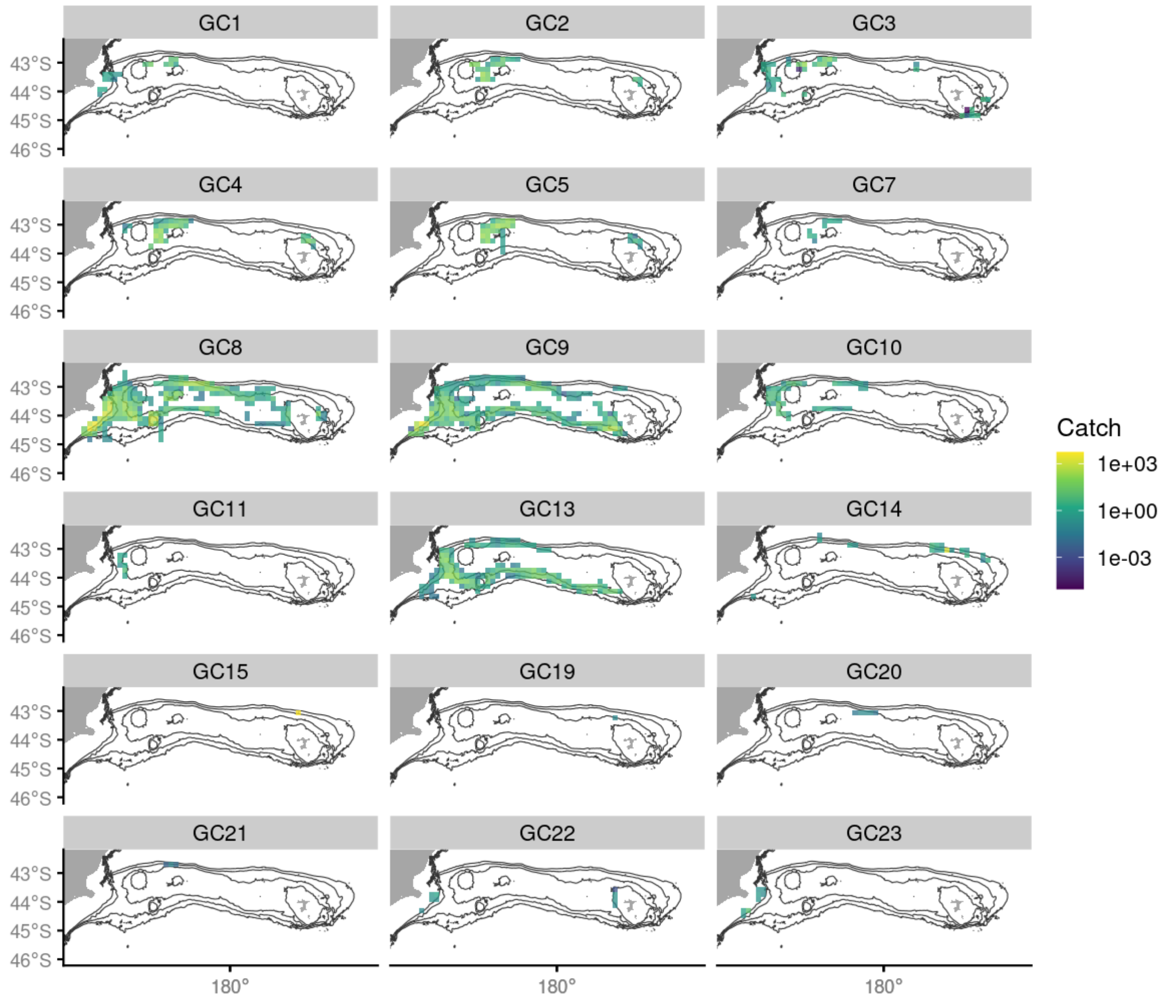


Figure 41: Spatial distribution of total catch (in log kg) of Anthozoa as extracted from the *cod* database. Gears are included if they caught the taxa in at least one instance.

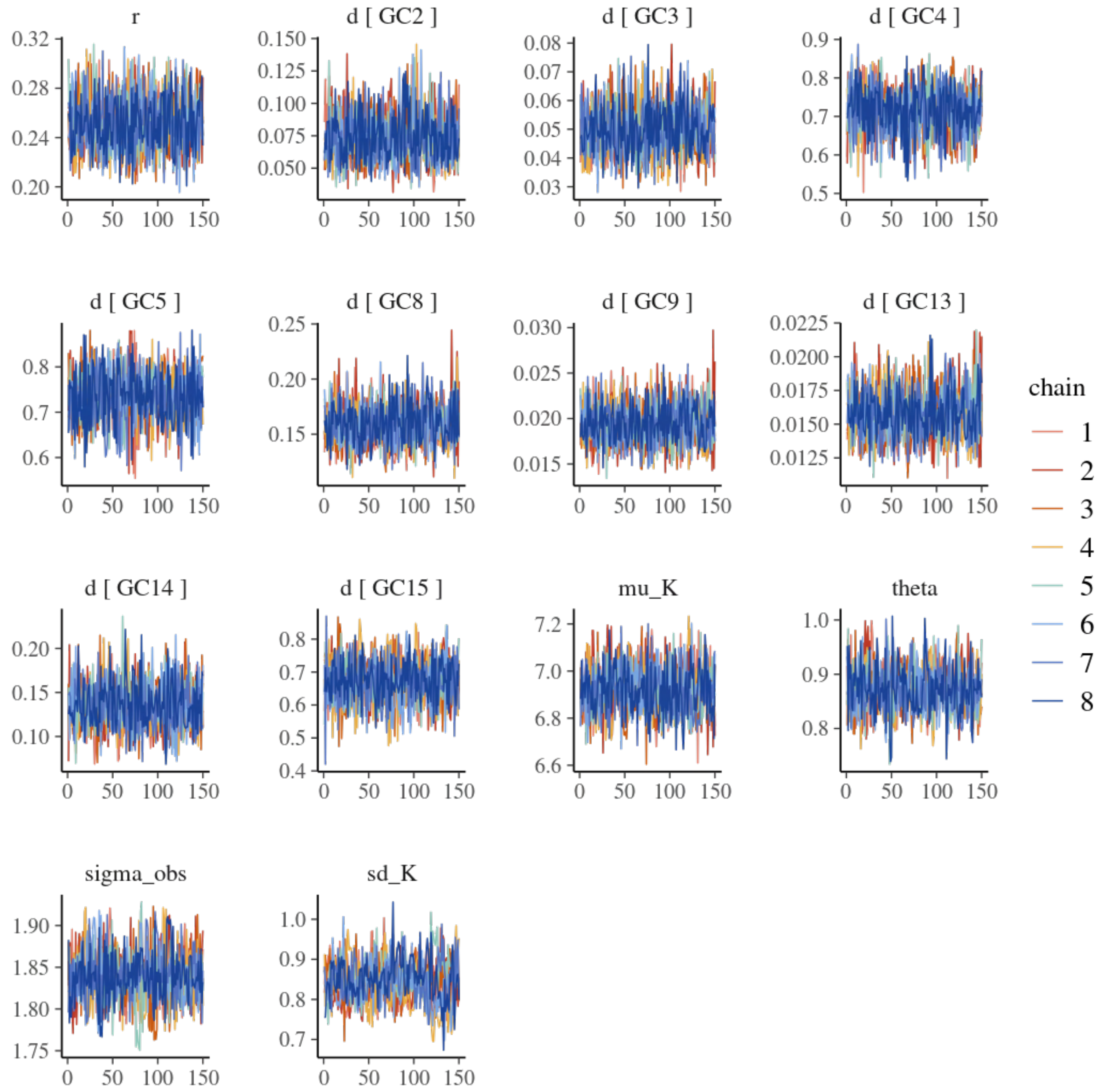


Figure 42: MCMC trace plots for key parameters from the spatial Schaefer surplus production model applied to observed catch of Anthozoa on the Chatham Rise .

Table 10: Estimated quantities for the Chatham Rise model applied to Anthozoa. Posterior mean, median, standard deviation (SD), 5th (5%), and 95th (95%) quantiles are given, alongside the Rhat statistic, which measures convergence across multiple chains (converged models will have Rhat close to 1 [<1.05]). Ess bulk and ess tail estimate the effective (decorrelated) number of samples (effective sample size [ess]) from the posterior distribution in the bulk and tail of the posterior distribution, respectively (note, that due to the properties of this estimate, ess may be greater than the number of actual samples). Gear categories (GC) refer to those retained for analysis and described in Table 5.

Variable	Mean	Median	SD	5%	95%	Rhat	ess bulk	ess tail
r	0.25	0.25	0.02	0.22	0.29	1.00	1 433	1 256
d [GC2]	0.07	0.07	0.02	0.05	0.11	1.00	1 470	910
d [GC3]	0.05	0.05	0.01	0.04	0.06	1.00	1 346	1 201
d [GC4]	0.71	0.72	0.06	0.61	0.81	1.01	1 615	994
d [GC5]	0.74	0.74	0.05	0.65	0.82	1.01	2 088	867
d [GC8]	0.16	0.16	0.02	0.13	0.19	1.00	1 098	1 110
d [GC9]	0.02	0.02	0.00	0.02	0.02	1.00	1 381	1 143
d [GC13]	0.02	0.02	0.00	0.01	0.02	1.00	1 213	1 041
d [GC14]	0.13	0.13	0.03	0.09	0.18	1.00	1 366	1 102
d [GC15]	0.67	0.68	0.07	0.56	0.78	1.01	1 863	949
μ_K	6.92	6.92	0.10	6.76	7.09	1.00	945.4	959
ξ	0.87	0.87	0.04	0.81	0.94	1.00	1 130	850
σ_{Obs}	1.84	1.84	0.03	1.79	1.89	1.00	1 141	524
σ_K	0.85	0.84	0.05	0.76	0.93	1.03	257	259

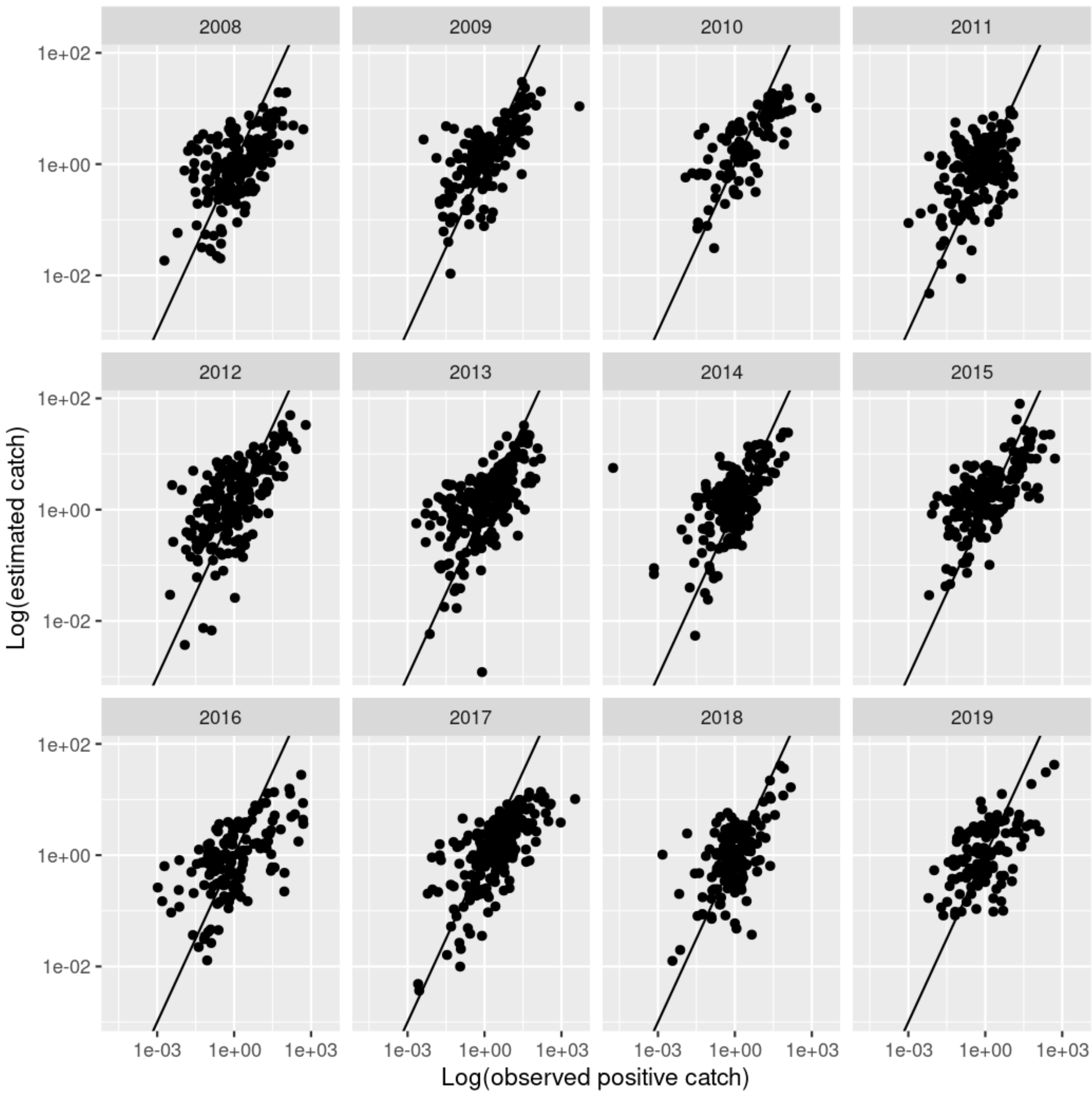


Figure 43: Plot of estimated (posterior mean) against observed Anthozoa catches (in kg) by year, for events with positive catch.

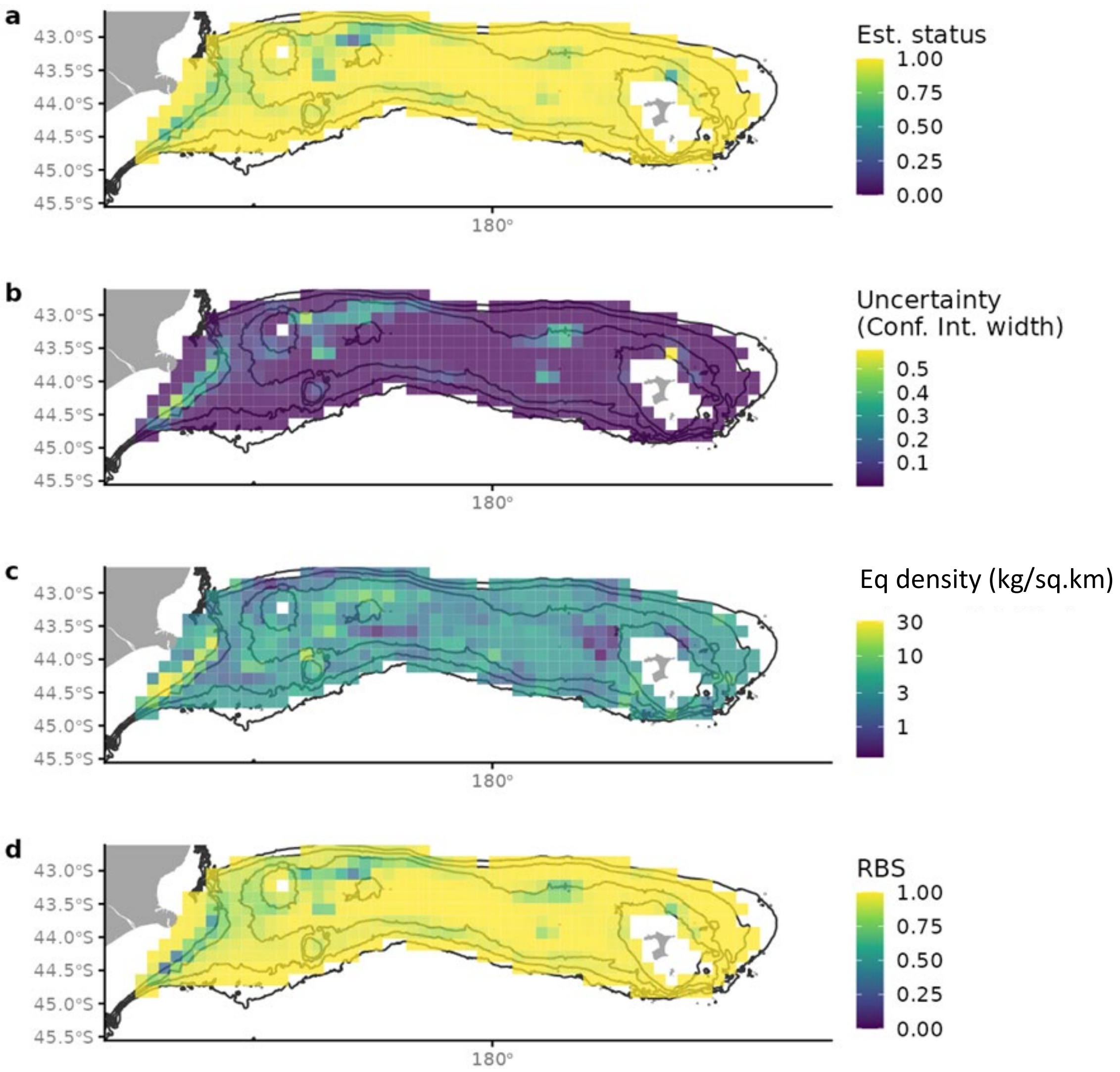


Figure 44: Estimates depletion status at 2019 (a), estimation uncertainty (on log-odds scale; b), estimated density at pre-fishing equilibrium (c), and estimated RBS (d) for Anthozoa within the extent of the hoki survey strata on the Chatham Rise. Values in panels a, c, and d are given at the mean of the posterior distribution.

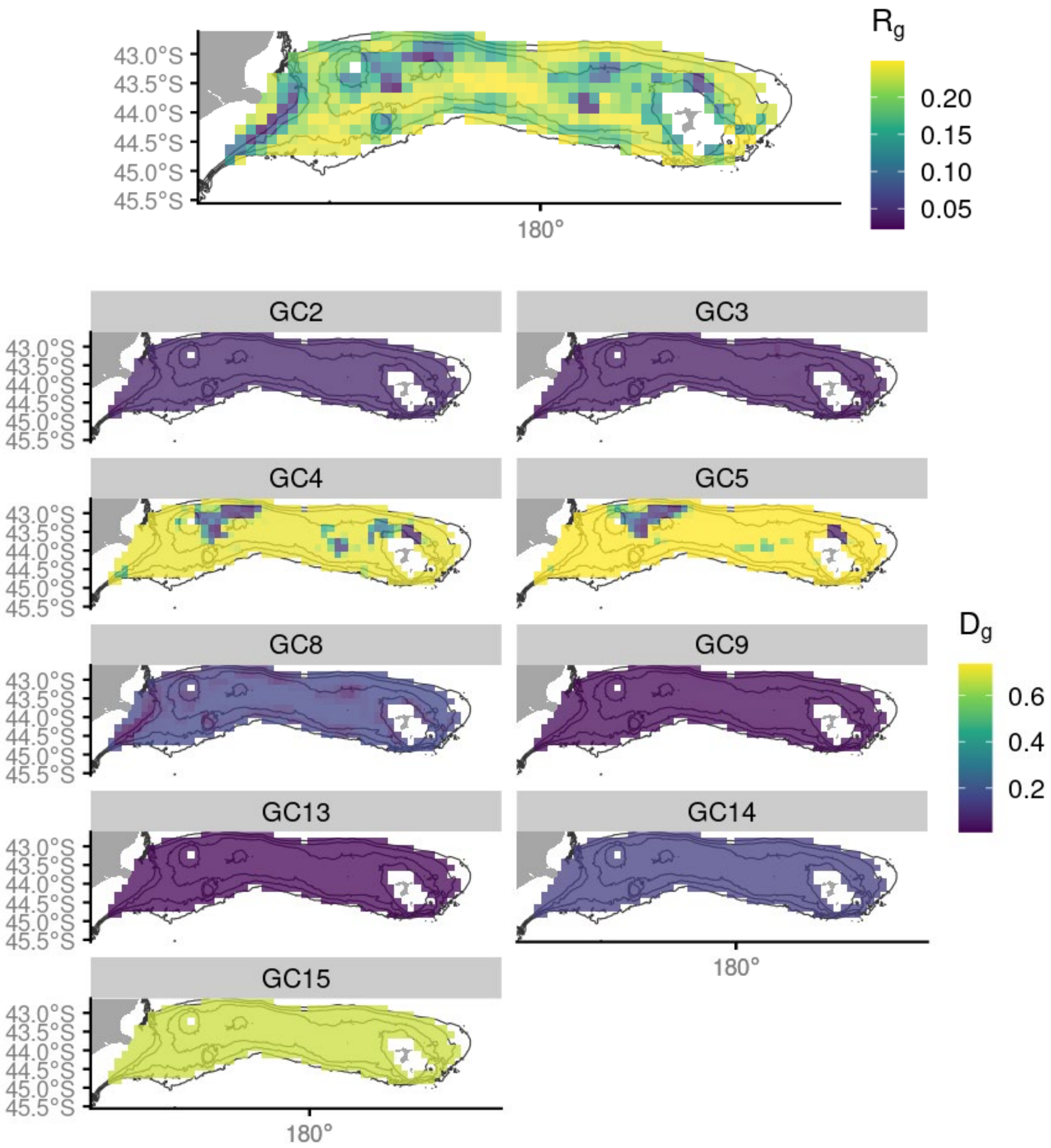


Figure 45: Decomposing RBS for Anthozoa within the extent of the hoki survey strata on the Chatham Rise into local vulnerability (R_g – top panel) and depletion rates (D_g – bottom panels) for different gear categories.

Demospongiae

As for Anthozoa, most of the observed catch of Demospongiae was attributed to trawls assigned to GC8 and GC9 (Figure 46), with GC13 showing lower bycatch of demosponges. Model convergence was mostly satisfactory, with some autocorrelation in the chains for the estimate of productivity (*r*) and initial equilibrium biomass (Figure 47). Most fleets were estimated to have gear depletion rates *d* between 25% and 40%, except for GC13 and GC22 which had estimates of *d* < 10% (Table 11). Fits to the catch were acceptable (Figure 48).

Estimated depletion for demosponges overall was low throughout the Chatham Rise, with some areas of high depletion around Mernoo Bank especially (Figure 49a). There were some differences in the location of depletion 'hotspots' between the population model and the RBS approach, with the RBS approach predicting higher depletion along the continental shelf off Banks Peninsula, corresponding to high effort from fishing fleets operating gear belonging to GC8 and GC9 (Figure 49d).

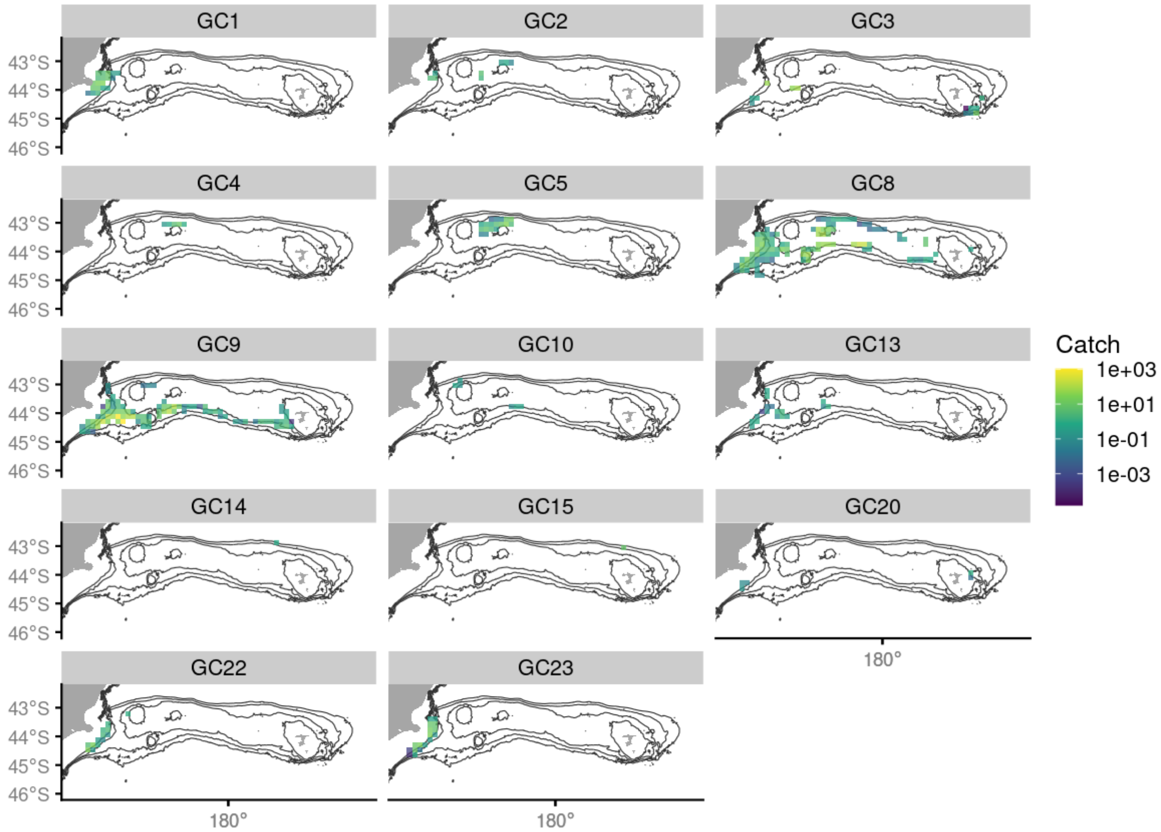


Figure 46: Spatial distribution of total catch (in kg) of demosponges as extracted from the *cod* database. Gears are included if they caught the taxa in at least one instance.

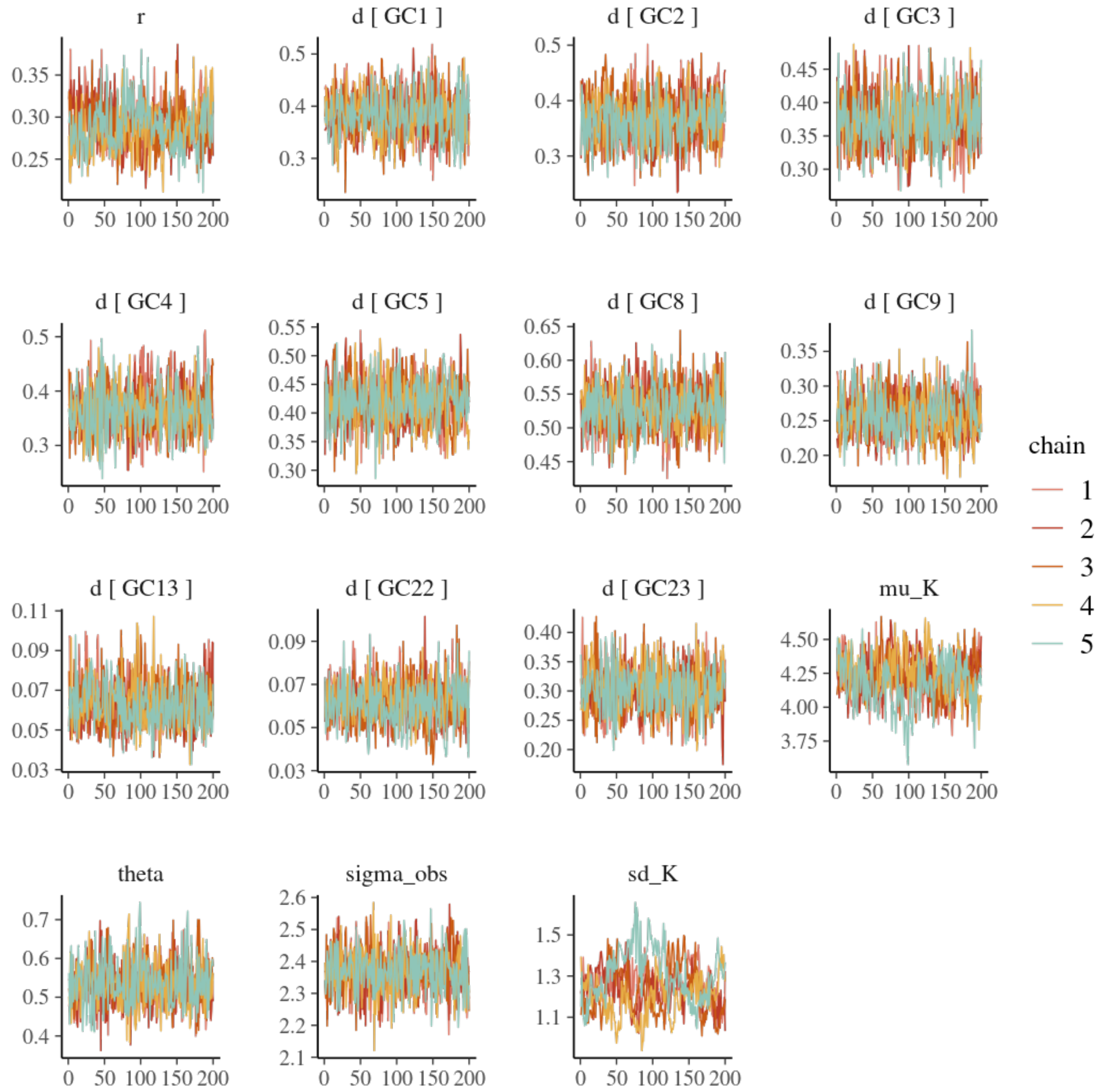


Figure 47: MCMC trace plots for key parameters from the spatial Schaefer surplus production model applied to observed catch of demosponges on the Chatham Rise.

Table 11: Estimated quantities for the Chatham Rise model applied to Demospongiae. Posterior mean, median, standard deviation (SD), 5th (5%), and 95th (95%) quantiles are given, alongside the Rhat statistic, which measures convergence across multiple chains (converged models will have Rhat close to 1 [<1.05]). Ess-bulk and ess tail estimate the effective (decorrelated) number of samples (effective sample size [ess]) from the posterior distribution in the bulk and tail of the posterior distribution, respectively (note, that due to the properties of this estimate, ess may be greater than the number of actual samples). Gear categories (GC) refer to those retained for analysis and described in Table 5.

Variable	Mean	Median	SD	5%	95%	Rhat	ess bulk	ess tail
r	0.29	0.29	0.03	0.25	0.34	1.02	272.33	555.28
d [GC1]	0.38	0.38	0.04	0.31	0.46	1.01	934.64	602.04
d [GC2]	0.37	0.37	0.04	0.30	0.44	1.01	987.74	477.33
d [GC3]	0.37	0.37	0.04	0.31	0.44	1.01	1063.77	760.36
d [GC4]	0.37	0.36	0.04	0.29	0.44	1.00	1011.81	686.77
d [GC5]	0.42	0.42	0.04	0.35	0.49	1.00	964.57	817.80
d [GC8]	0.53	0.53	0.03	0.47	0.58	1.00	1046.02	908.49
d [GC9]	0.26	0.26	0.03	0.21	0.31	1.01	593.54	853.69
d [GC13]	0.06	0.06	0.01	0.05	0.08	1.00	866.57	667.71
d [GC22]	0.06	0.06	0.01	0.05	0.08	1.01	785.91	589.10
d [GC23]	0.30	0.30	0.04	0.24	0.37	1.00	1071.65	631.06
μ_K	4.22	4.22	0.16	3.95	4.48	1.04	117.41	267.57
ξ	0.53	0.53	0.06	0.44	0.63	1.01	1072.72	642.30
σ_{Obs}	2.37	2.37	0.07	2.25	2.48	1.00	936.62	730.42
σ_K	1.25	1.24	0.11	1.07	1.43	1.10	39.71	59.03

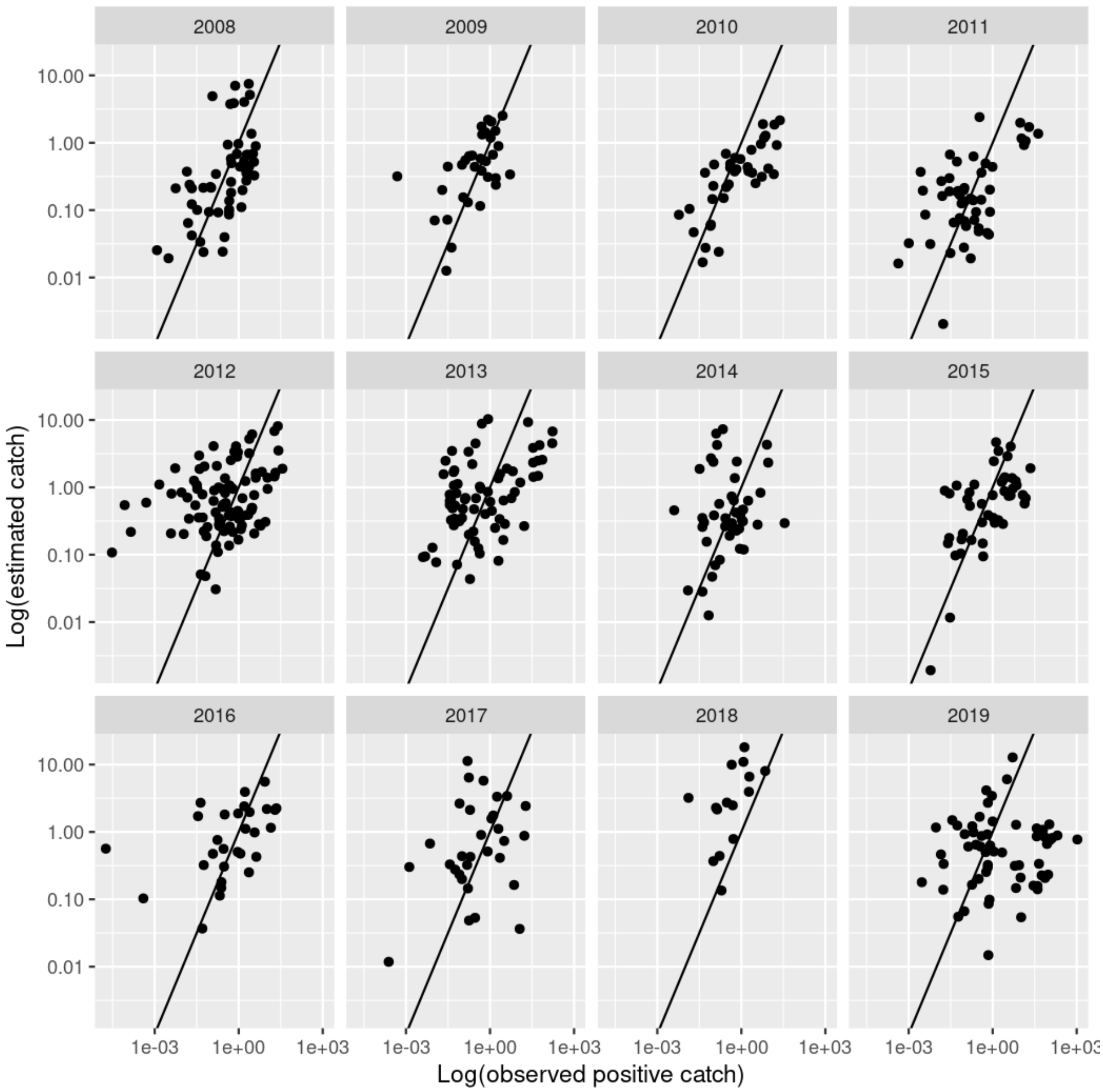


Figure 48: Plot of estimated (posterior mean) against observed demosponge catches (in kg) by year, for events with positive catch.

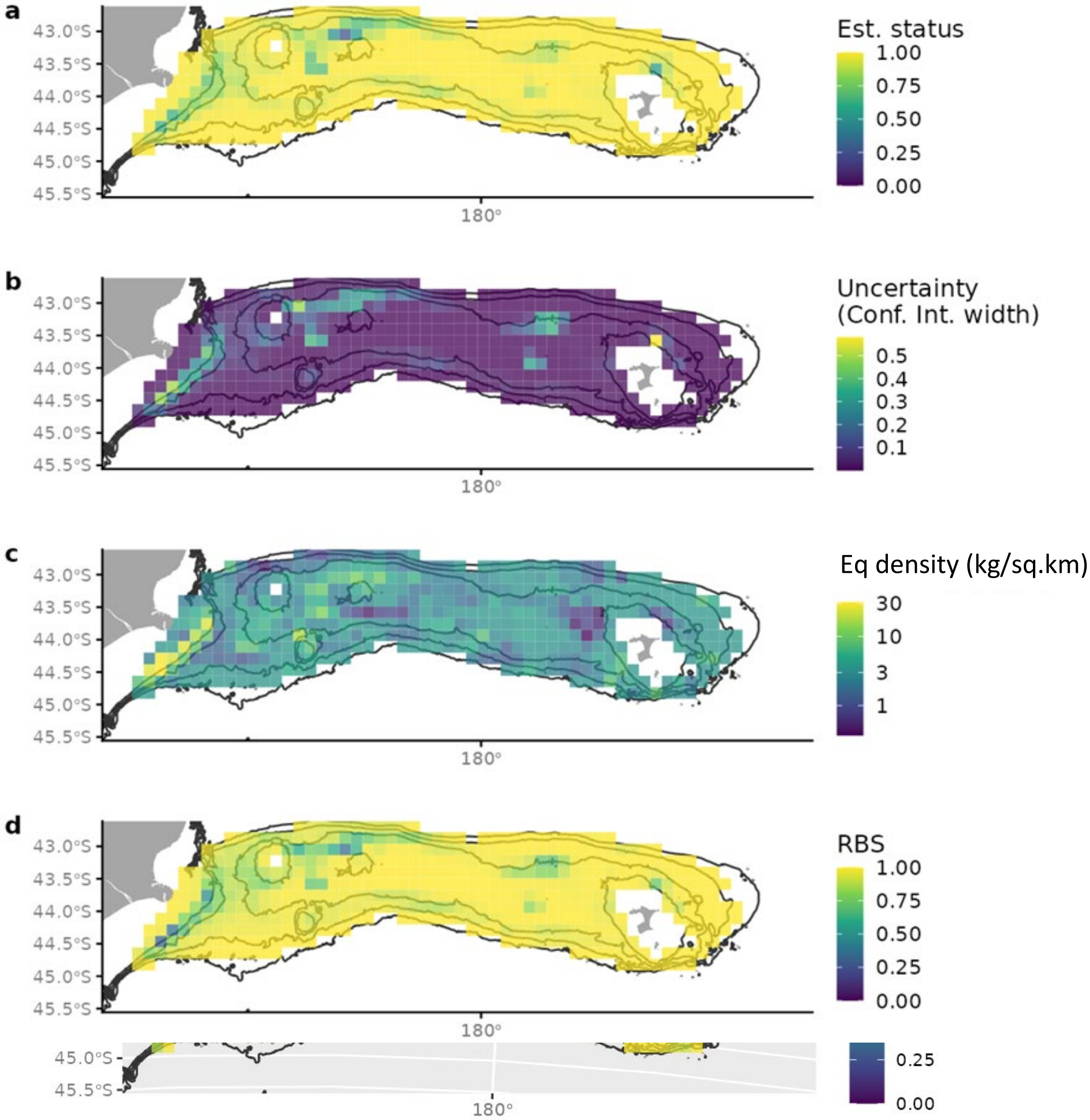


Figure 49: Estimates depletion status at 2019 (a), estimation uncertainty (on log-odds scale; b), estimated density at pre-fishing equilibrium (c), and estimated RBS (d) for demosponges within the extent of the hoki survey strata on the Chatham Rise. Values in panels a, c, and d are given at the mean of the posterior distribution.

Hexactinellida

Model convergence for hexactinellid sponges was acceptable, with chains well-mixed for most parameters except for parameters associated with the initial equilibrium biomass (Figure 50). Gear depletion rates *d* were slightly lower than the prior (between ~0.3 and ~0.5) for three of the gear categories that caught this group, the fourth (GC13) had a lower *d* around 0.12, despite high observed catch (Figure 51, Table 12). Fits to the catch were acceptable in most years, with a slight tendency to underestimate the lower catches and overestimate high catches in some years of the time series (e.g., 2008, 2012, 2017; Figure 52).

Estimated biomass depletion for hexactinellid sponges was higher on the western end of the Chatham Rise corresponding with high fishing effort for key gear categories (e.g., GC9) (Figure 53a). RBS predictions were more pessimistic than model-based estimates of depletion, especially in the area southwest of the Mernoo Bank (Figure 53d), driven by high effort footprint from fishing fleets operating gears belonging to GC8 and GC9.

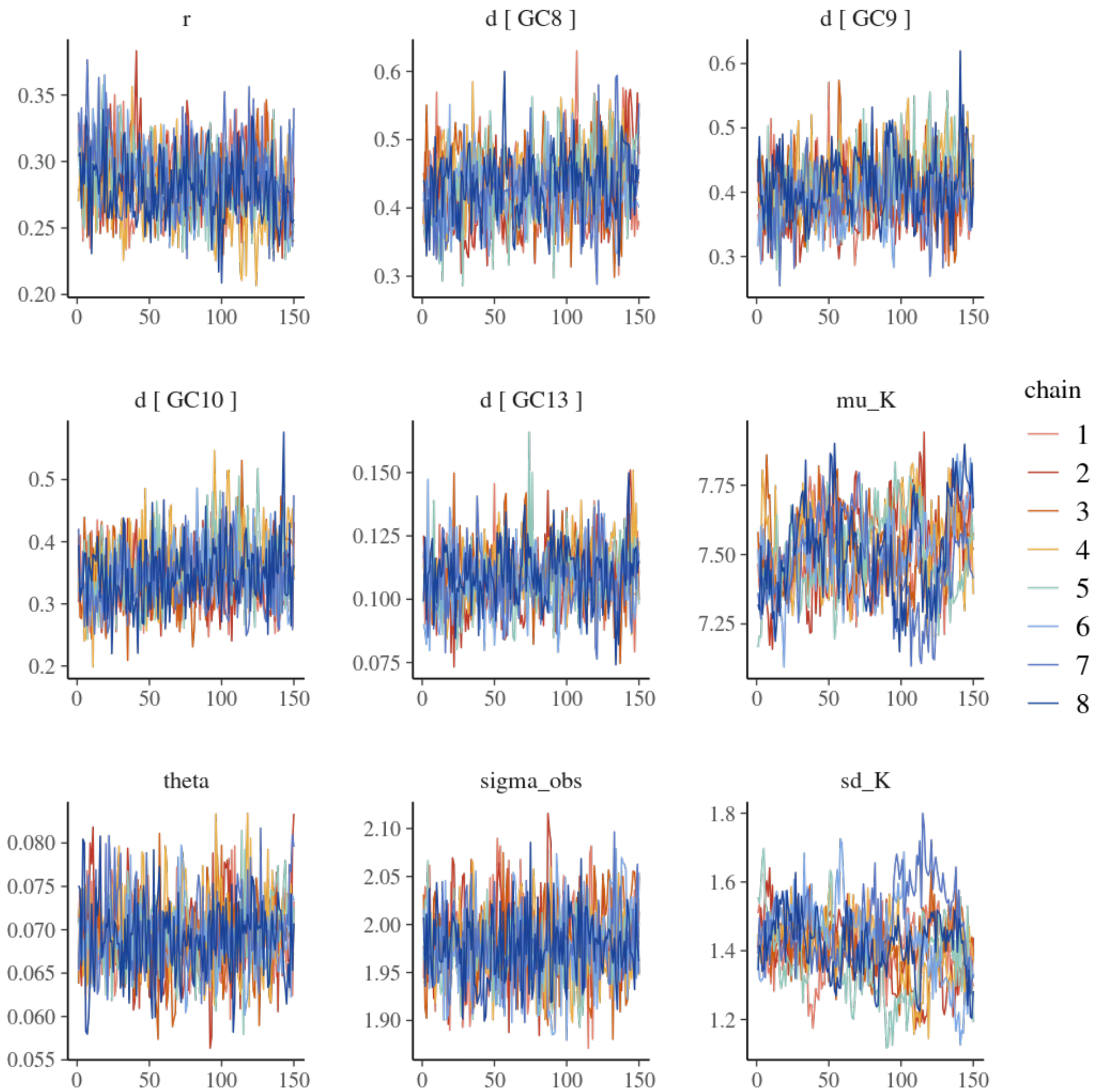


Figure 50: MCMC trace plots for key parameters from the spatial Schaefer surplus production model applied to observed catch of hexactinellid sponges on the Chatham Rise.

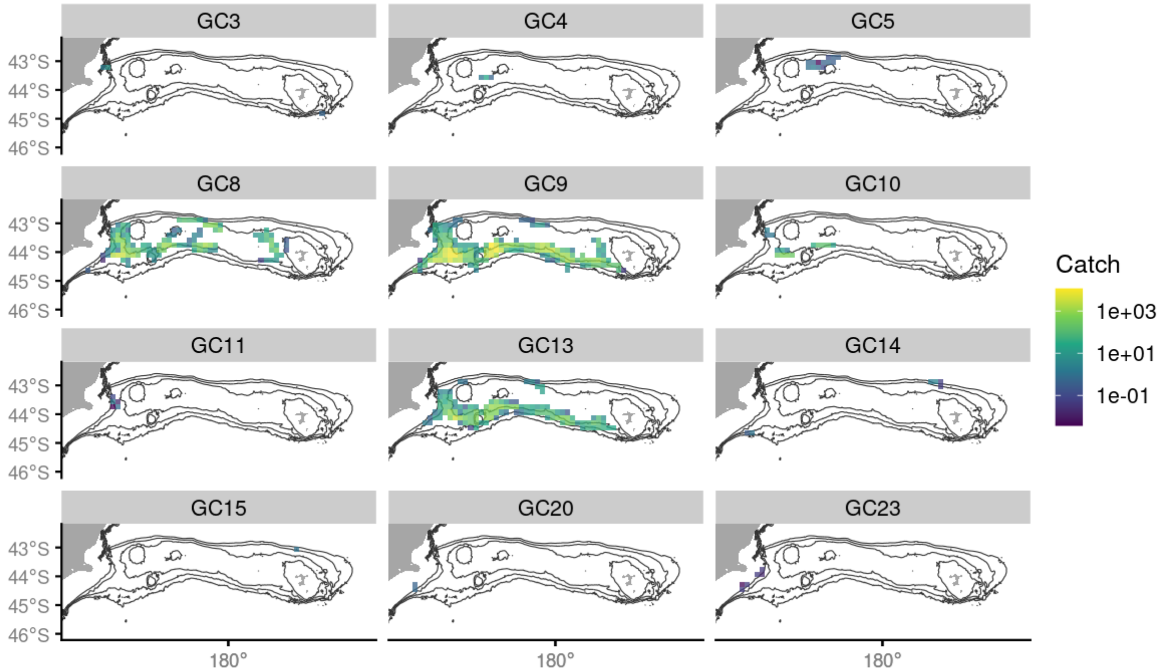


Figure 51: Spatial distribution of total catch (in kg) of hexactinellid sponges extracted from the *cod* database. Gears are included if they caught the taxa in at least one instance.

Table 12: Estimated quantities for the Chatham Rise model applied to hexactinellid sponges. Posterior mean, median, standard deviation (SD), 5th (5%), and 95th (95%) quantiles are given, alongside the Rhat statistic, which measures convergence across multiple chains (converged models will have Rhat close to 1 [<1.05]). Ess-bulk and ess tail estimate the effective (decorrelated) number of samples (effective sample size [ess]) from the posterior distribution in the bulk and tail of the posterior distribution, respectively (note, that due to the properties of this estimate, ess may be greater than the number of actual samples). Gear categories (GC) refer to those retained for analysis and described in Table 5.

Variable	Mean	Median	SD	5%	95%	Rhat	ess bulk	ess tail
r	0.28	0.28	0.03	0.24	0.33	1.04	138.24	657.18
d [GC8]	0.43	0.43	0.05	0.35	0.52	1.04	193.74	833.20
d [GC9]	0.40	0.40	0.05	0.33	0.49	1.07	100.45	315.37
d [GC10]	0.35	0.34	0.05	0.27	0.44	1.03	217.67	539.26
d [GC13]	0.11	0.11	0.01	0.09	0.13	1.03	516.36	941.37
μ_K	7.51	7.50	0.14	7.28	7.75	1.05	120.73	371.53
ξ	0.07	0.07	0.00	0.06	0.08	1.01	1095.92	685.87
σ_{Obs}	1.98	1.98	0.04	1.91	2.05	1.01	1114.72	601.22
σ_K	1.41	1.42	0.10	1.24	1.58	1.16	38.75	44.77

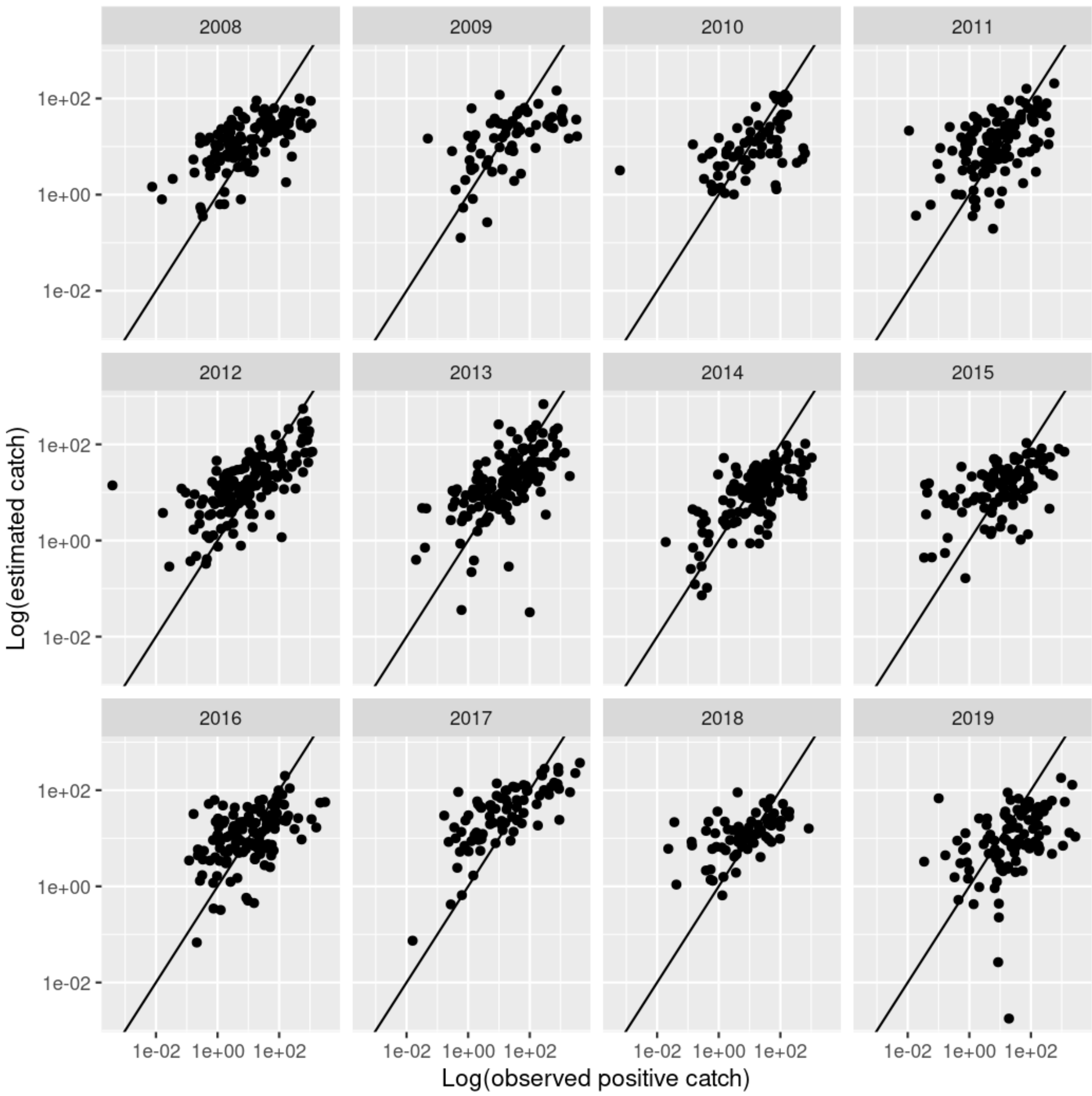


Figure 52: Plot of estimated (posterior mean) against observed hexactinellid sponge catches (in kg) by year, for events with positive catch.

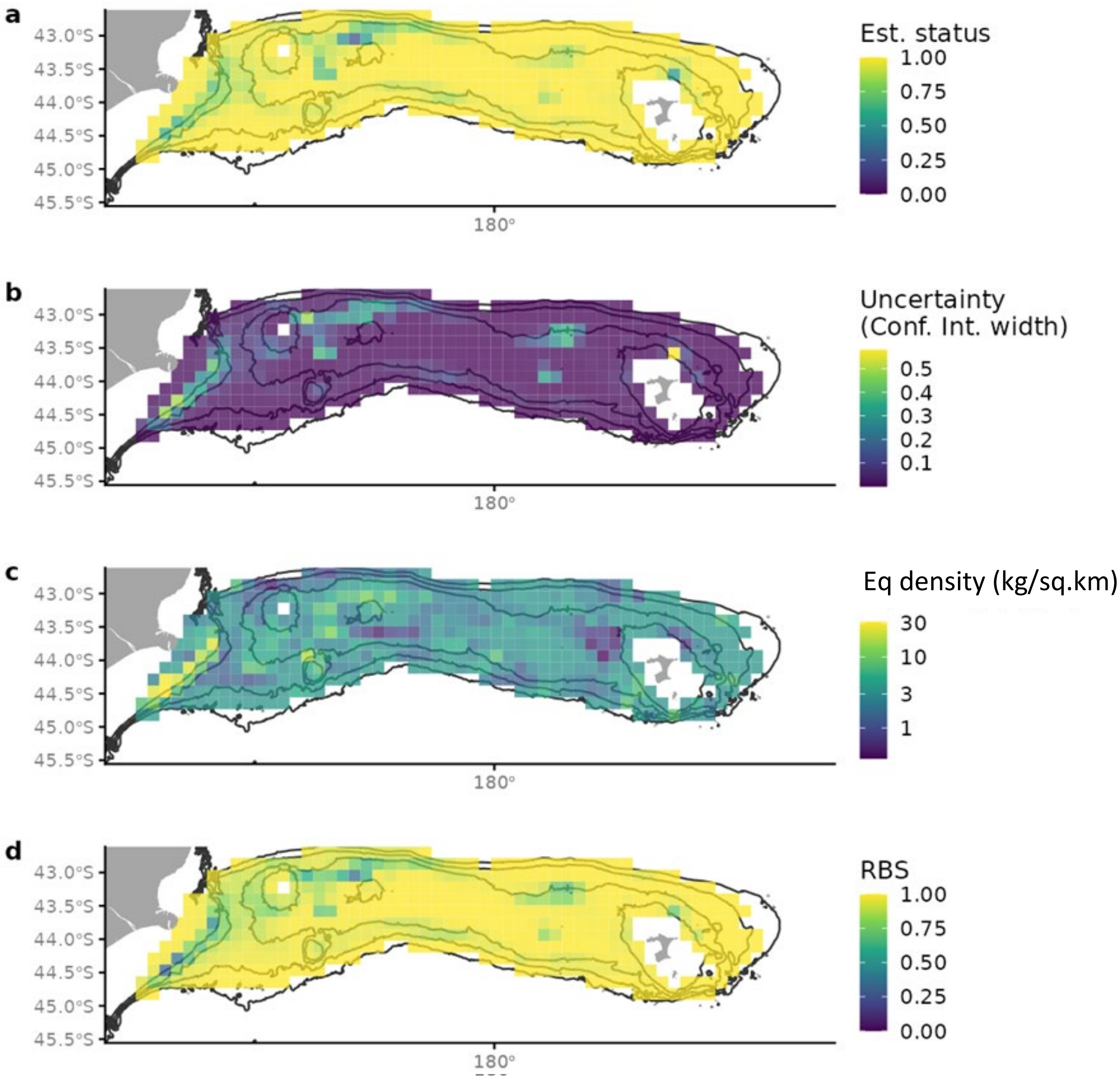


Figure 53: Estimates depletion status at 2019 (a), estimation uncertainty (on log-odds scale; b), estimated density at pre-fishing equilibrium (c), and estimated RBS (d) for hexactinellid sponges within the extent of the hoki survey strata on the Chatham Rise. Values in panels a, c, and d are given at the mean of the posterior distribution.

Pennatulacea

Catch for key gear categories (except for GC8, GC9, GC13) was relatively low for sea pens (Figure 54). Model convergence for this group was not acceptable, with all chains showing strong autocorrelation between iterations for all parameters (Figure 55). Given issues with model convergence we do not include predictions of biomass depletion for this taxonomic group.

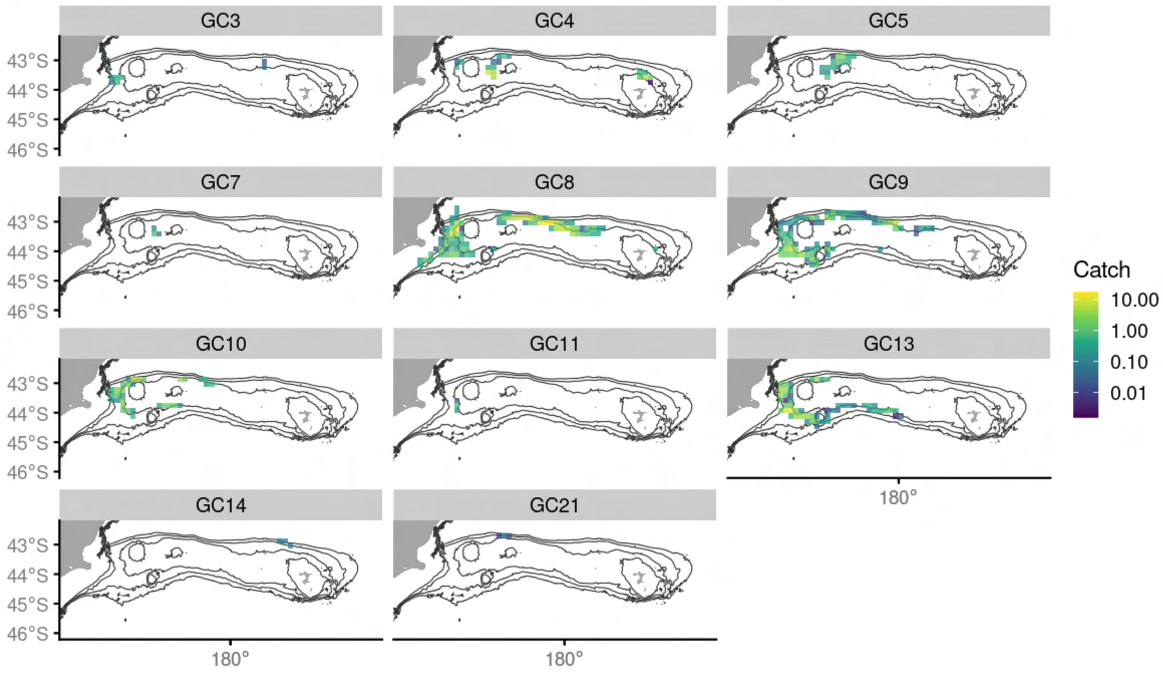


Figure 54: Spatial distribution of total catch (in log kg) of Pennatulacea as extracted from the *cod* database. Gears are included if they caught the taxa in at least one instance.

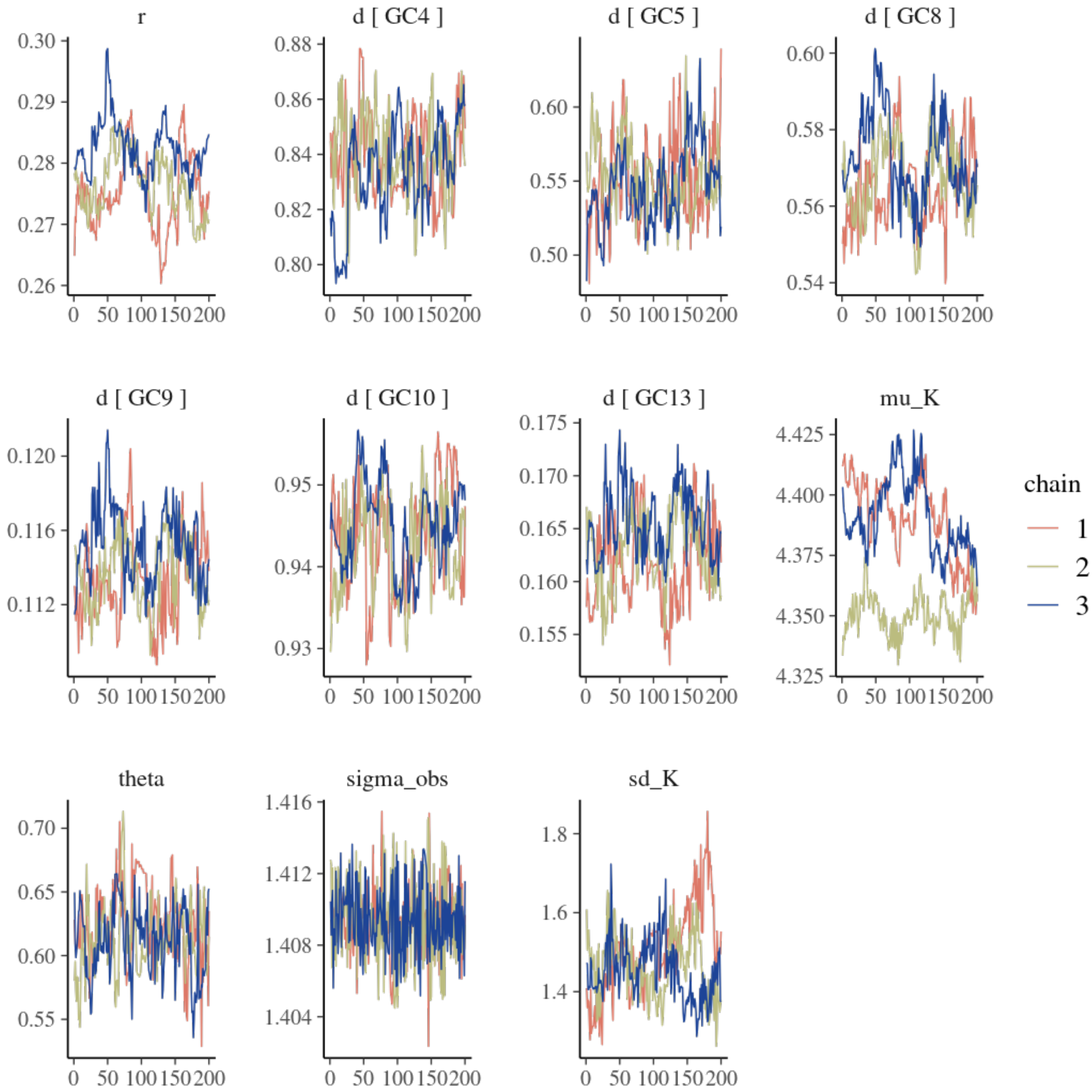


Figure 55: MCMC trace plots for key parameters from the spatial Schaefer surplus production model applied to observed catch of pennatulids on the Chatham Rise.

All taxa groups

Productivity *r* was estimated to be near 0.3 for all taxa groups, with the *r* estimate for Anthozoa, the taxa with the most records, being the most precise. The gear specific depletion rates *d* estimated for the three gear categories representing the highest effort footprint (GC8, GC9, and GC13) varied across groups, with GC8 and GC9 estimated to have higher depletion rates for sponges (demosponges and hexactinellid sponges) than Anthozoa, and GC13 having highest depletion rates for hexactinellid sponges.

3.3.4 Application of Mormede et al. (2021) methods for Chatham Rise

The VAST model for holothurians was constructed with spatial, temporal, and spatio-temporal effects only. However, a spatio-temporal effect was unable to be fitted in the encounter probability component of the model. This result was because the decorrelation distance for this component of the model was unable to be estimated. Model fit diagnosis is given in Appendix 7. The dharma residuals showed an even spread of values between zero and one across time and the spatial domain and followed the diagonal well of the Q-Q plot (Appendix 7). These results indicate that the model is well specified and gives confidence in the specification of the model.

Log biomass was estimated for holothurians on the Chatham Rise from 2007 to 2020 (Figure 56). Log biomass is approximately stable throughout the Chatham Rise, where very low biomass (dark blue) tends to be distributed throughout the central Chatham Rise and very high biomass tends to occur in the centre north and south Chatham Rise. There is little variability from one year to the next; however, there appears to be greater biomass throughout the central Chatham Rise in 2015. In the last year of the time series, there is very low biomass in two central Chatham Rise locations, and the central north and south Chatham Rise hotspots are reduced and at the very edges of the rise. Log standard error of the biomass estimates increases from 2007 to 2020 (Figure 57). However, log standard error remains relatively low over time, in particular from 2007 to 2015, when the log standard error is approximately stable. Higher uncertainty occurs in two hotspots corresponding to low biomass, and at the southern-central edge of the Chatham Rise where high biomass occurs. In comparison, there is low uncertainty at the north edge Chatham Rise, corresponding to high biomass, and the east and west of the rise.

The fishing impact overlap from the fish and squid fishery is very low for holothurians throughout the Chatham Rise in space and time (Figure 58). Areas in white are areas which are unfished and areas in yellow show only a small impact on holothurians. That is, within a grid cell, the percentage overlap between the fisheries trawl footprint and the estimated biomass of holothurians is small. However, there are small isolated areas where there is a more moderate overlap between the fishery footprint and the biomass. These grid cells are mostly located in the central-north Chatham Rise and in the central-south edge of the Chatham Rise. The percentage of biomass within a cell that is impacted by fish and squid trawling gets as high as 9.56% (Table 13). However, this is an outlier, and the mean/median are as low as 0.06% and 0.02%, respectively, and there is little variability (standard deviation of 0.28%). The most recent year where the fishing impact of the fish and squid fishery could be calculated was 2019 (Figure 59). The fishing impact overlap remains consistent with the patterns in previous years and is very low throughout the domain and with some isolated grids in the central-north Chatham Rise edge containing high fishing impact overlap values.

There is very little overlap between the scampi fishery and the distribution of holothurians in the Chatham Rise (white areas, Figure 60). Over time, the fishery persists throughout the north-west Chatham Rise and at small patches in the north-east and south-west Chatham Rise. However, the percentage of biomass within a grid cell impacted by the scampi fishery is consistently very low (light yellow areas) except within a few grid cells (red to purple areas) which still have a low impact (maximum impact of 2.74%, Table 13). Similarly to the fish and squid fishery, the impact is consistently low with a mean and median of 0.04% and 0.02% proportion of the biomass impacted (Table 13). The maximum value determined is an outlier as there is very little variability (standard deviation of 0.15) and the inter-quartile range only extends from 0.01% to 0.03%. The 2019 impact percentage remains consistent with the previous years (Figure 61).

The VAST model for Anthozoa was constructed with spatial, temporal, and spatio-temporal effects. When using a random walk for the model temporal effect, the model parameter which estimates temporal covariation ($L_{\beta}(f)$) was unable to be estimated. Therefore, temporal correlation was estimated as independent among years. All other effects were able to be estimated for the model. Model fit was assessed using residual and Q-Q diagnosis plots (Appendix 8). The residual diagnosis plot shows an even spread of dharma residuals values across time and space. Additionally, the observed vs. expected values fit well across the diagonal of the Q-Q plot (Appendix 8). These results give confidence in the specification of the Anthozoa model.

There is some variability in the log biomass of Anthozoa; however, the majority of the biomass remains stable (Figure 62). High biomass tends to occur in the central, centre-north, and centre-south Chatham Rise. However, very high biomass fluctuates considerably (red). In 2007, and particularly in 2015, very high biomass occurs throughout the Chatham Rise, whereas in other years high/very high biomass tends to occur at the identified hotspots. Low biomass occurs throughout the central-east Chatham Rise. In the last year of the time series, 2020, biomass is low to moderate throughout the Chatham Rise. The log standard error of Anthozoa biomass is approximately stable over time (Figure 63). However, the years 2007 and 2010 have low uncertainty hotspots in the east and central-north edge of the Chatham Rise, respectively. These correspond to high biomass estimates. Likewise, the years 2015, 2017, and 2019 have high uncertainty estimates corresponding to areas of mostly low biomass estimates. The years 2017 and 2019 were years where trawl surveys were not carried out. Therefore, uncertainty in these years is to be expected.

The impact of the fish and squid fishery on Anthozoa is small across the Chatham Rise (Figure 64). That is, the proportion of the biomass impacted remains very close to zero (mean of 0.08%, median of 0.06%, Table 13) throughout the Chatham Rise. There is fairly little variability (standard deviation of 0.09%, inter-quartile range between 0.03% and 0.09%) but, just to the south-west and north-east of the centre of the Chatham Rise, the percentage of biomass impacted increases to a maximum of 1.87%. However, this maximum is still very small. Across time there is little variability but there appears to be a small peak in 2014–2015 where the impact percentage reaches its maximum. The most recent year of data, 2019, is consistent with the pattern observed (Figure 65). It also appears that the hotspots identified in 2014–2015 have reduced and that there is very little impact on anthozoan biomass across the Chatham Rise.

The scampi fishery has a very small impact on the biomass of Anthozoa across the Chatham Rise (Figure 66). The scampi fishery only intersects with Anthozoa in the north-west Chatham Rise and at small patches in the north-east and south-west of the rise. The mean and median impact on the biomass is 0.06% and 0.05%, respectively. The maximum value for the proportion of anthozoan biomass impacted by scampi fishery is much greater than the upper quartile of the range (inter-quartile range between 0.03% and 0.08%, Table 13) which indicates that the maximum is an outlier (i.e., the maximum (0.5%) is greater than the upper quartile plus 1.5 times the inter-quartile range). The scampi fishery also has very little variability in its impact on Anthozoa (standard deviation of 0.05%). The most recent year studied, 2019, is consistent with the observed pattern and does not have any impact greater than 0.2% of the biomass (Figure 67).

Table 13: Summary statistics of the benthic impact due to the fish/squid and scampi fishery on holothurians and anthozoans on the Chatham Rise, expressed as the percentage of the taxon biomass that is impacted due to trawling for a fishery (Fish/Squid = gear categories 3, 8, 9, 13, 14, 15; Scampi = gear categories 4, 5, 6, 7; LQ= lower quartile; UQ = upper quartile of the data range).

	Holothuroidea					Anthozoa				
Fishery	Mean (sd)	Median	LQ	UQ	Min/Max	Mean (sd)	Median	LQ	UQ	Min/Max
Fish/Squid	0.06 (0.28)	0.02	0.01	0.03	0.00/9.56	0.08 (0.09)	0.06	0.03	0.09	0.00/1.87
Scampi	0.04 (0.15)	0.02	0.01	0.03	0.00/2.74	0.06 (0.05)	0.05	0.03	0.08	0.00/0.50

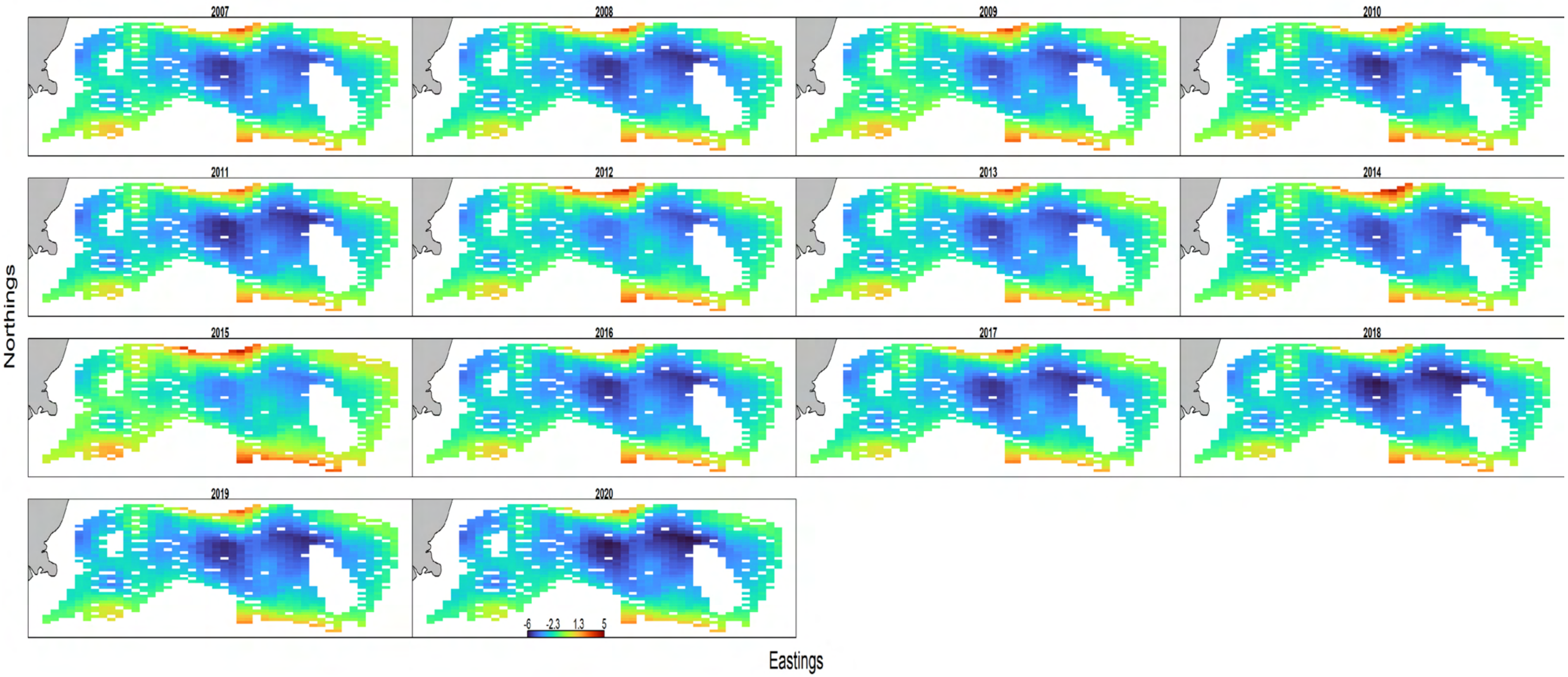


Figure 56: AST model log biomass estimates (log kg/km^2) for holothurians on the Chatham Rise.

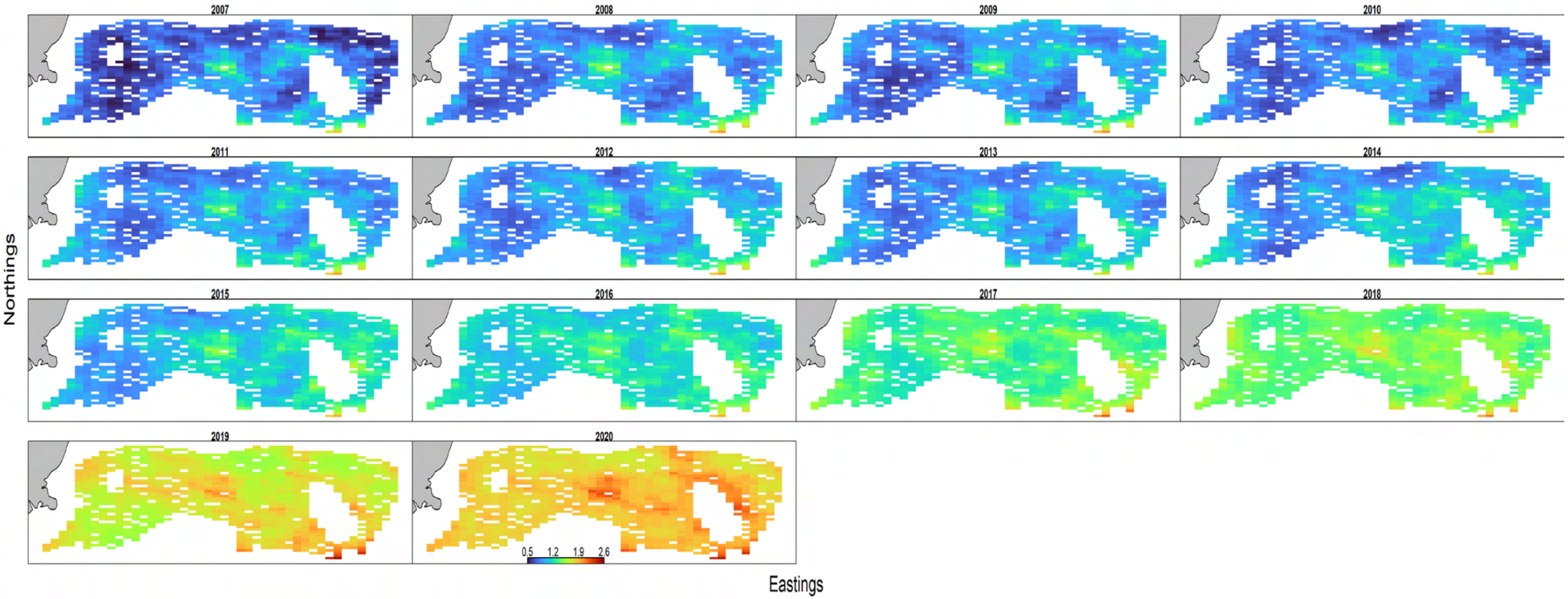


Figure 57: VAST model log standard error of estimated holothurian biomass on the Chatham Rise.

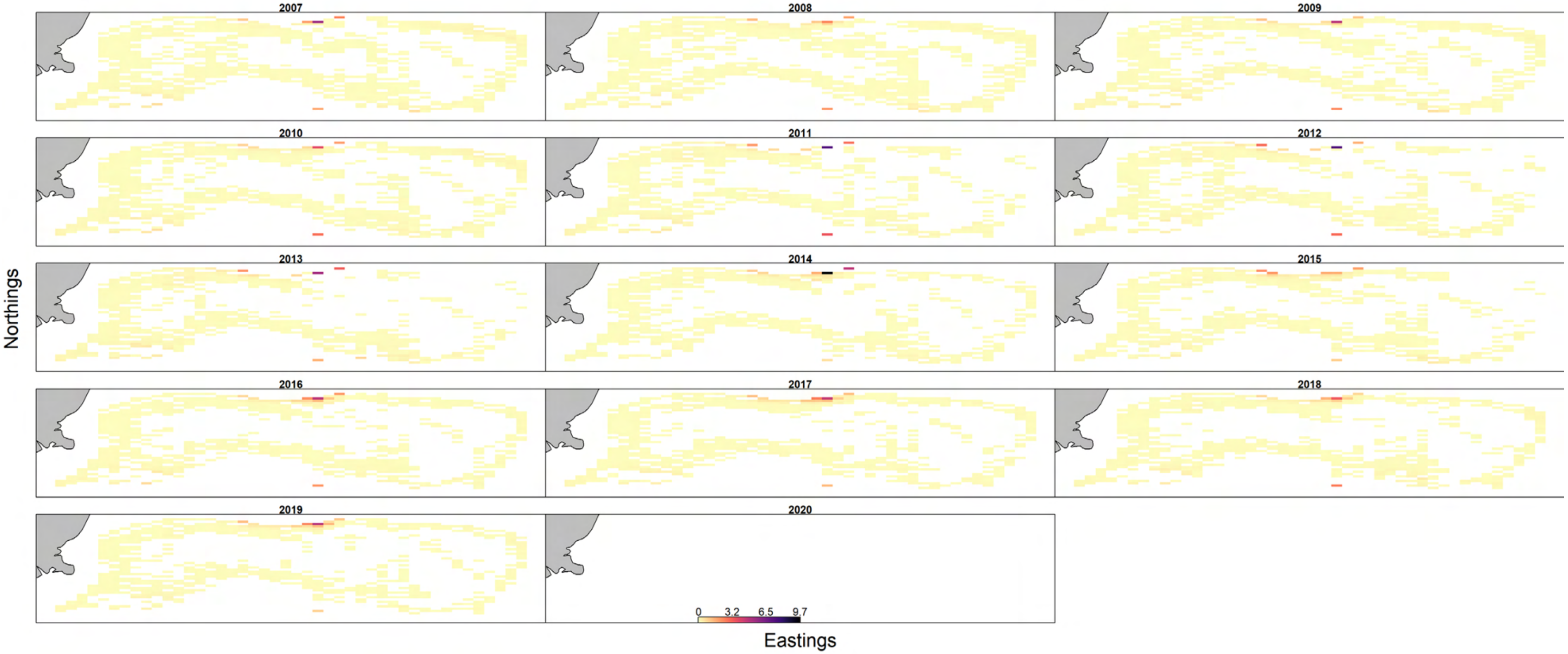


Figure 58: The impact of the fish and squid fishery on holothurians (% of biomass impacted) on the Chatham Rise from 2007 to 2019.

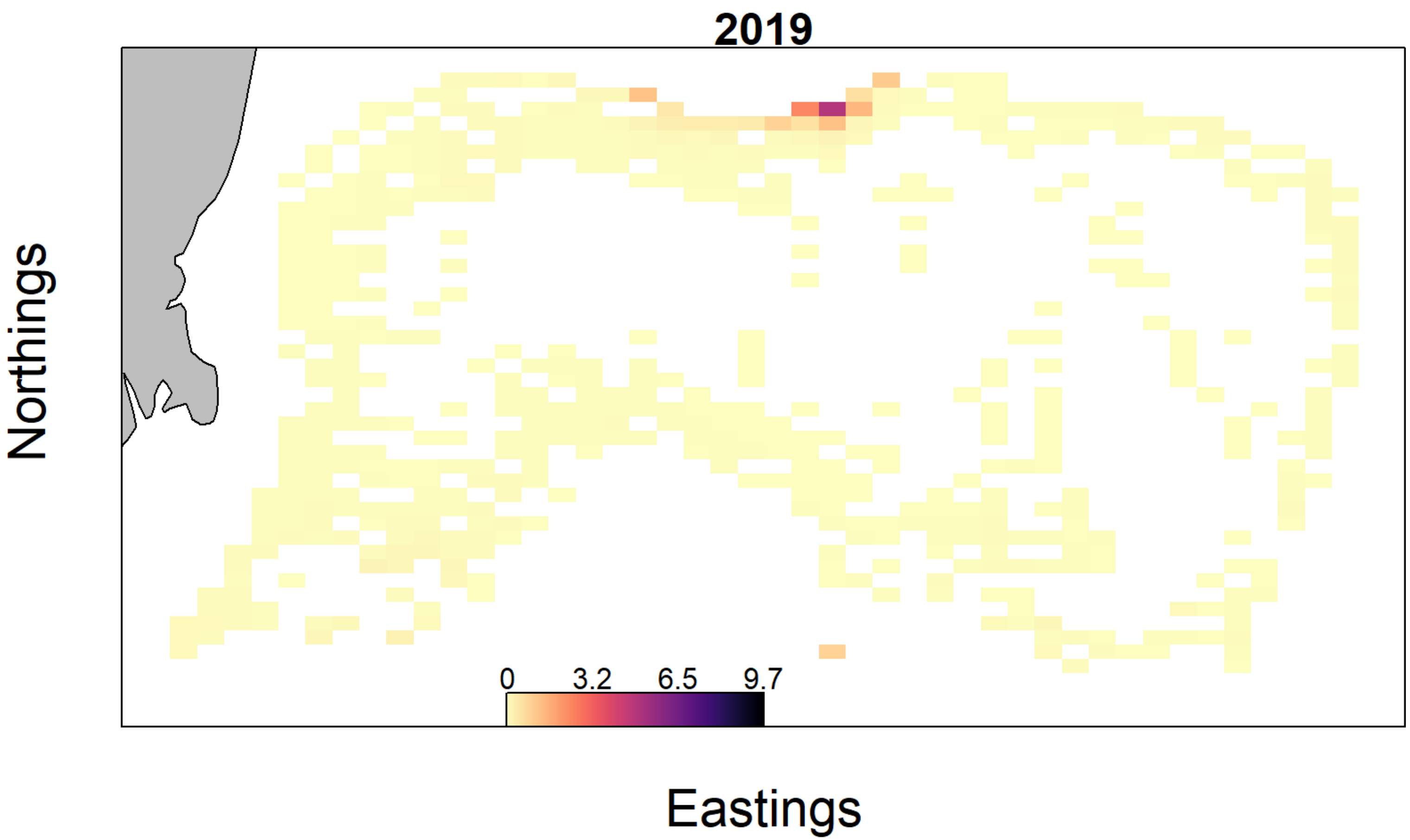


Figure 59: The impact of the fish and squid fishery on holothurians (% of biomass impacted) on the Chatham Rise in 2019.

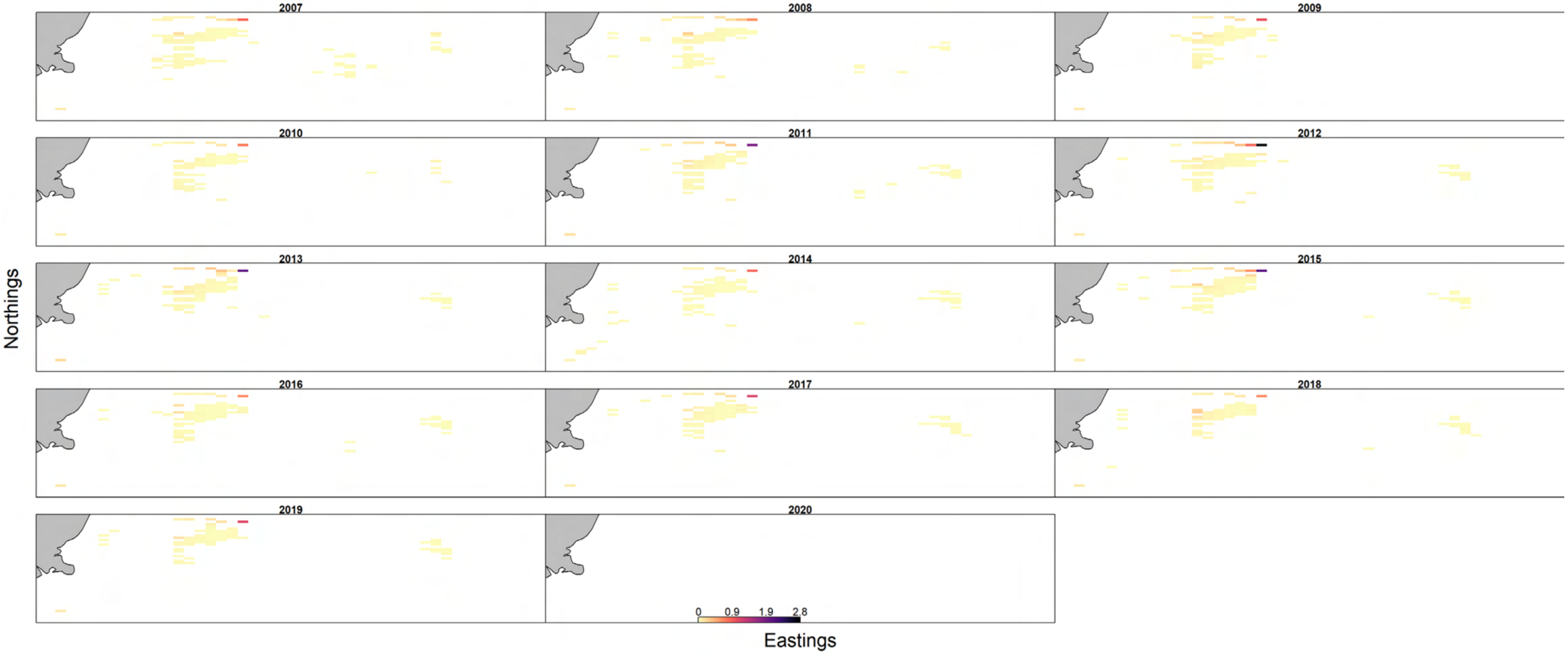


Figure 60: The impact of the scampi fishery on holothurians (% biomass impacted) on the Chatham Rise from 2007 to 2019.

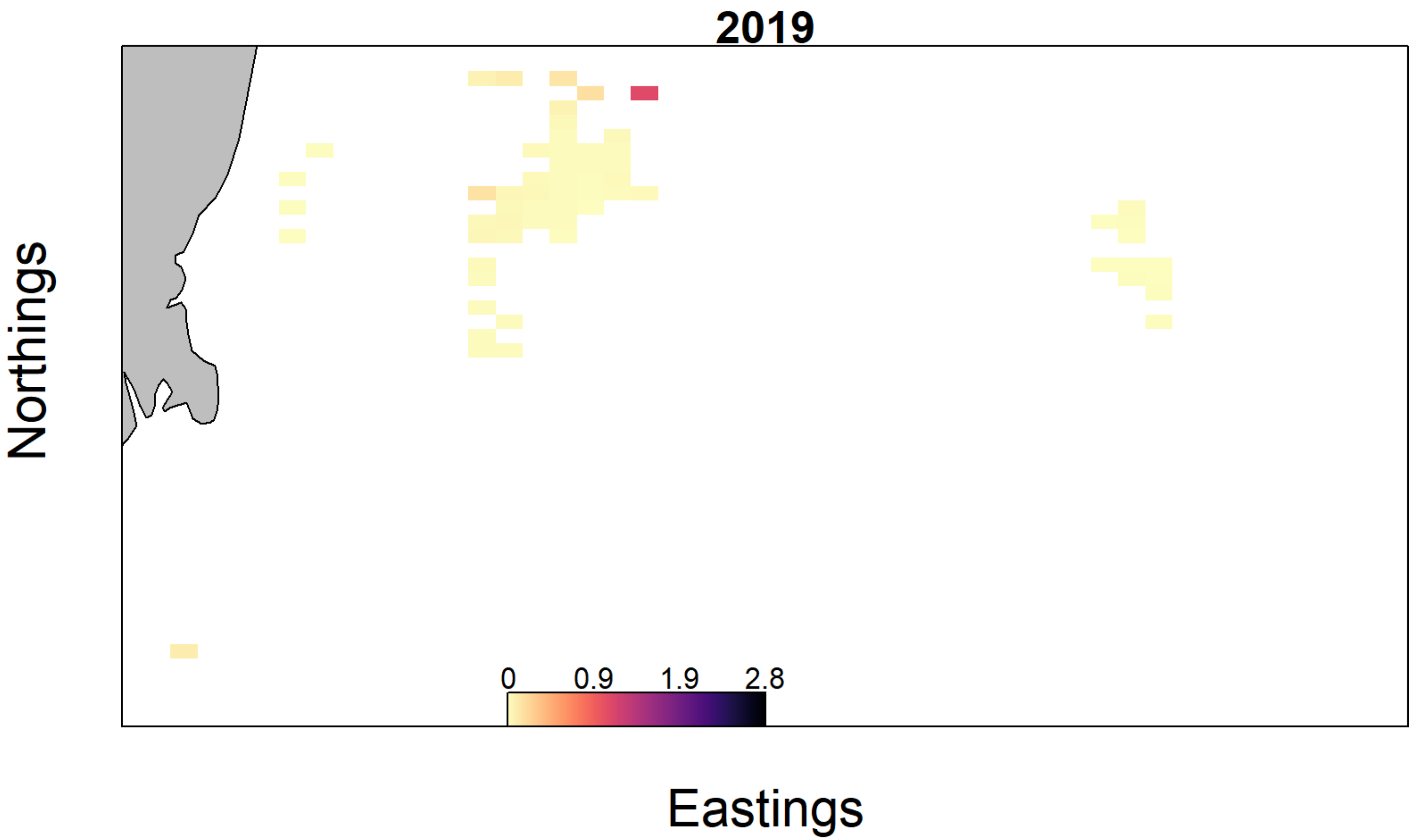


Figure 61: The impact of the scampi fishery on holothurians (% biomass impacted) on the Chatham Rise in 2019.

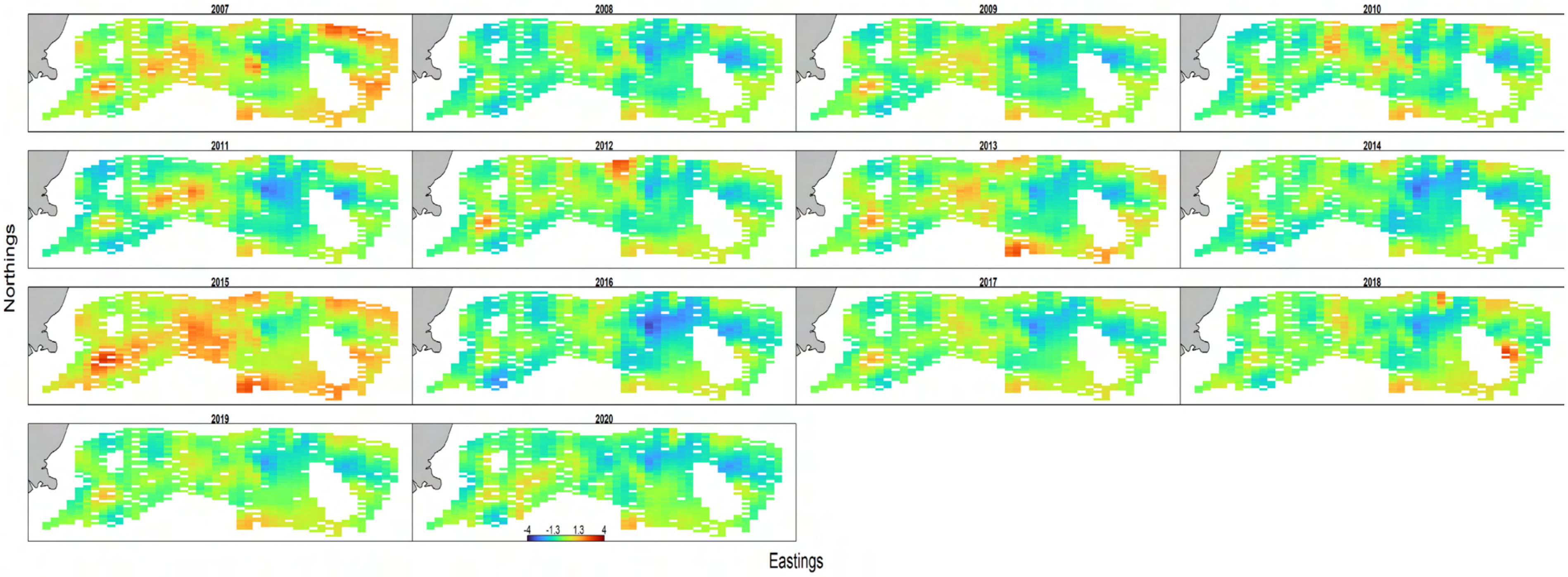


Figure 62: VAST model log biomass estimates (log kg/km^2) for Anthozoa on the Chatham Rise.

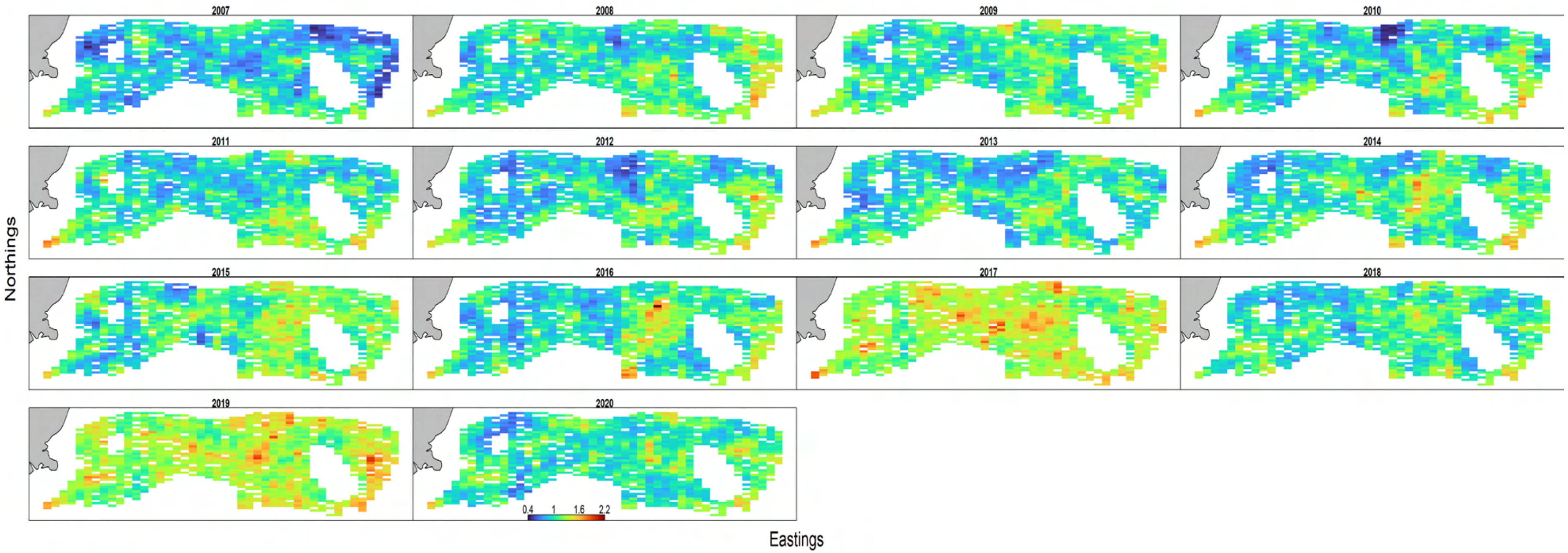


Figure 63: VAST model log standard error of estimated anthozoan biomass on the Chatham Rise.

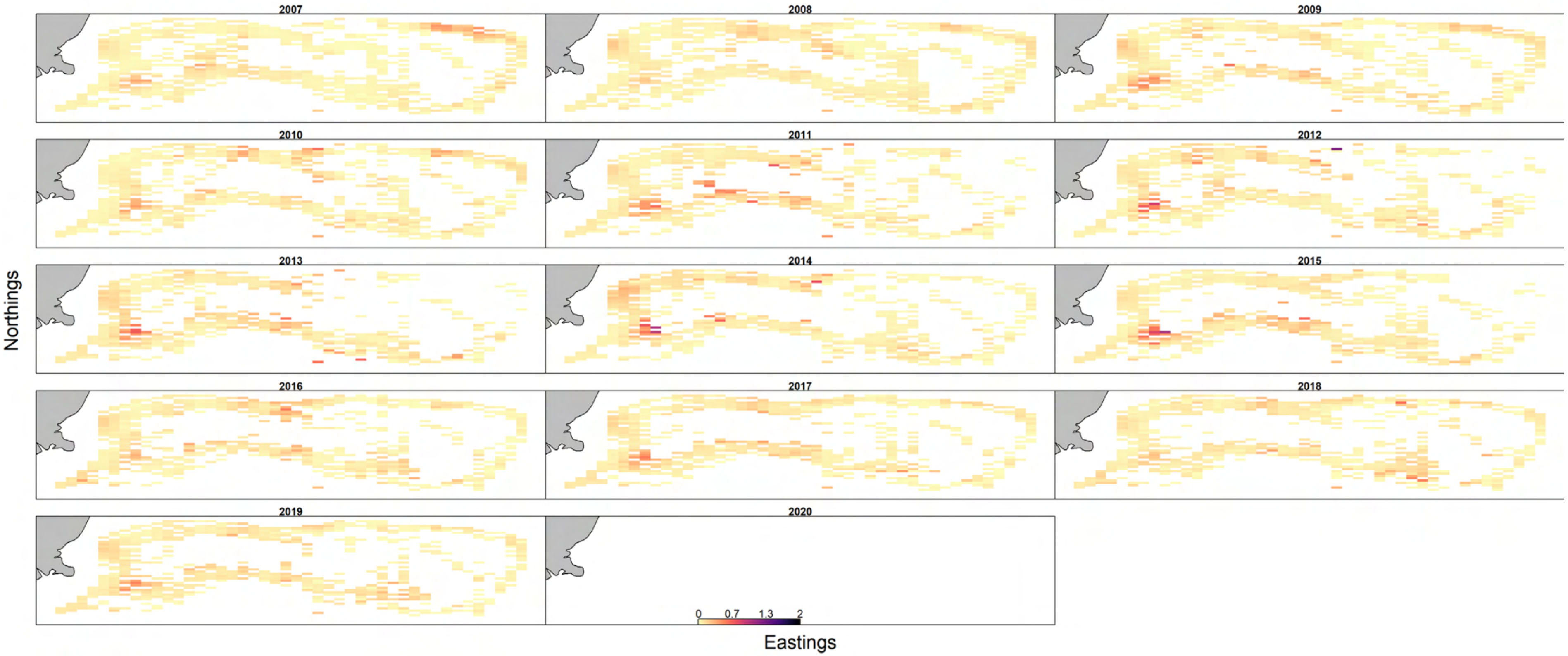


Figure 64: The impact of the fish and squid fishery on anthozoans (% biomass impacted) on the Chatham Rise from 2007 to 2019.

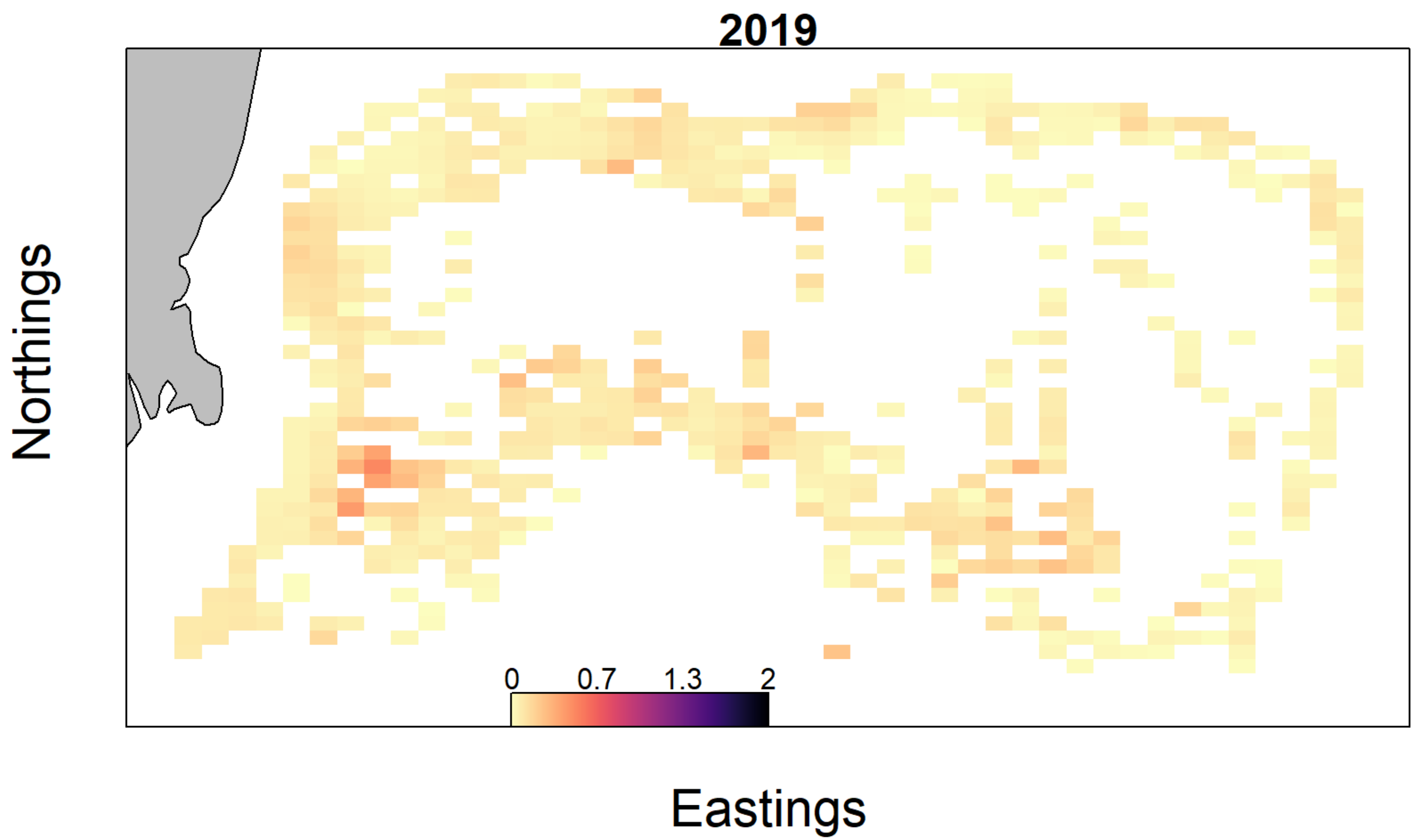


Figure 65: The impact of the fish and squid fishery on anthozoans (% biomass impacted) on the Chatham Rise in 2019.

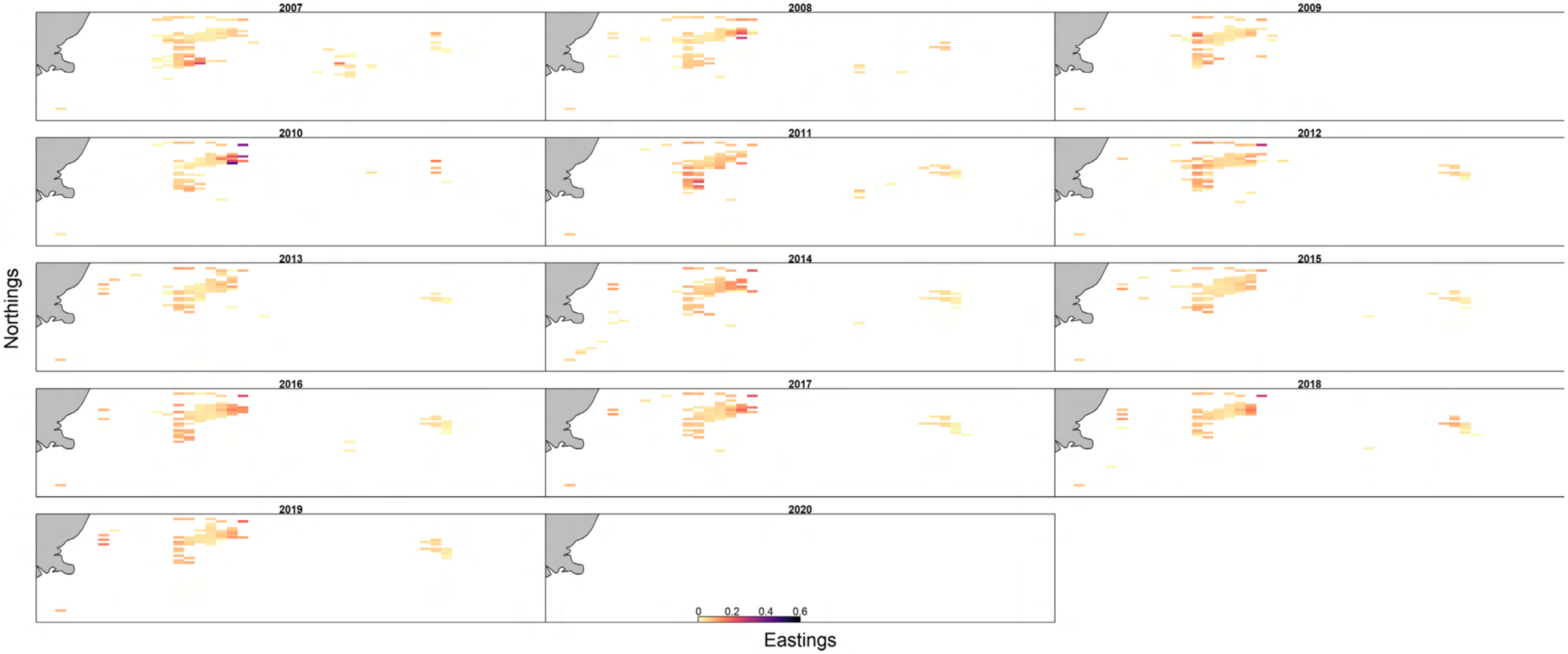


Figure 66: The impact of the scampi fishery on anthozoans (% biomass impacted) on the Chatham Rise from 2007 to 2019.

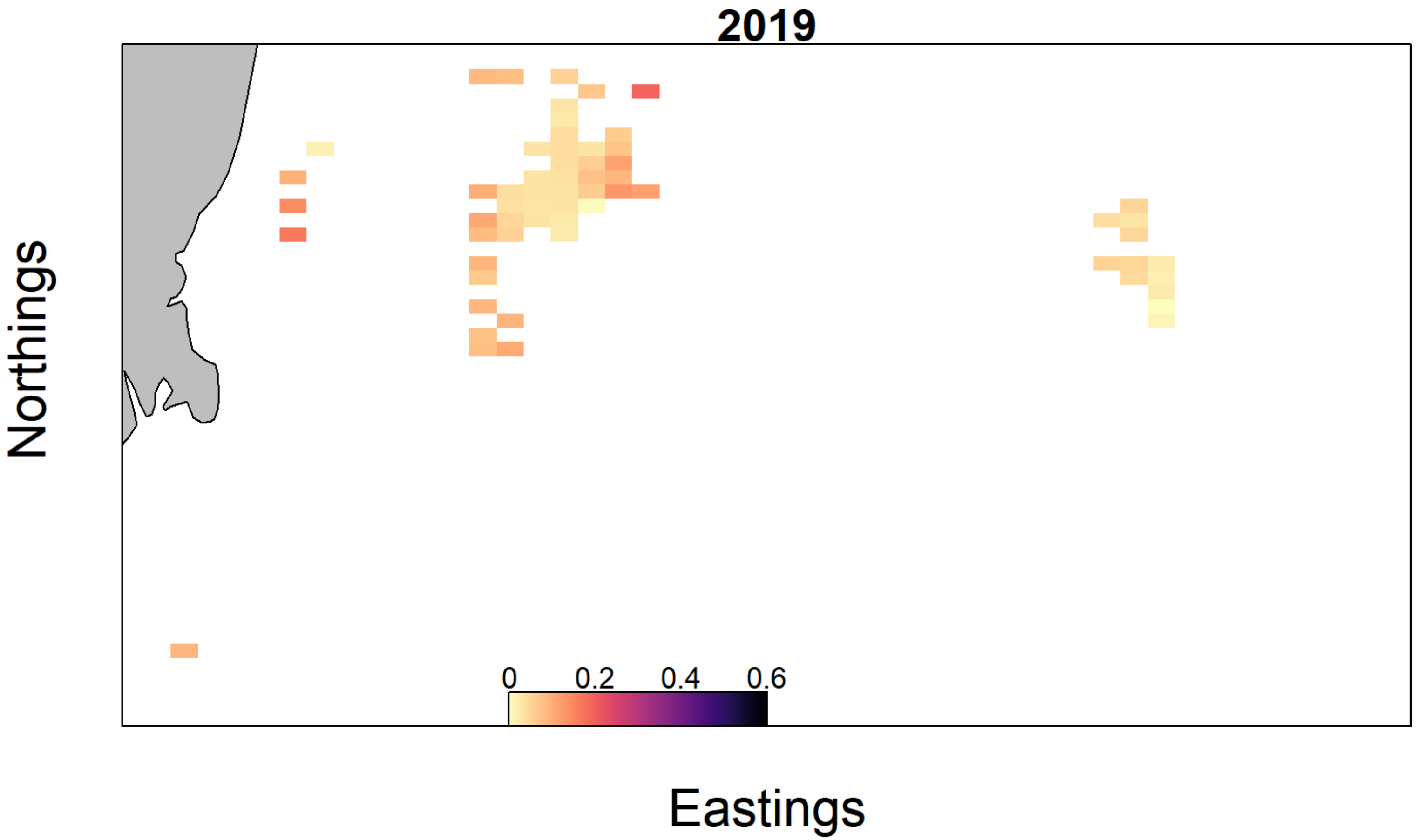


Figure 67: The impact of the scampi fishery on anthozoans (% biomass impacted) on the Chatham Rise in 2019.

4. DISCUSSION

The first objective of this project was to characterise all mobile bottom fishing gear configurations used since 2007–08 for inshore fisheries and since 1989–90 for deepwater fisheries. While it was possible to use a variety of information sources to provide descriptions of all trawl and dredge gear types and configurations, data on dredge configurations on a tow-by-tow basis were not available to categorise these deployments in an objective manner. However, data recorded in the Fisheries New Zealand's database *cod* was sufficient to be used to categorise trawl deployments throughout the TS and EEZ. Trawl deployments were categorised into 24 gear categories. These gear categories were assigned to all trawls represented in the most recent trawl footprint analysis (BEN2019-01; Baird & Mules 2021) and, together with expert-derived information on the effective seafloor contact of different trawl gear components, were used to achieve the second project objective to map the spatial and temporal extent of bottom contact by different fishing gear configurations. The final objective of the project was to characterise the impacts of different gear configurations on key benthic taxa and/or communities. Four methods were employed to achieve this objective using outputs from the first two project objectives. The first two methods used published approaches to provide spatially explicit estimates of the benthic impact of fishing for three functional groups of benthic fauna (Mormede et al. 2017) and the relative benthic status (RBS) of a selection of vulnerable marine ecosystem (VME) indicator taxa (Pitcher et al. 2017) within New Zealand's TS and EEZ. The application of the second two methods was more exploratory and for a limited spatial area. The first of these methods explored a modification to the RBS approach that could incorporate specific information on benthic bycatch taxa (also VME indicator taxa) from the Chatham Rise. The final method used additional data to extend a previous estimate of the impact of trawling on a soft-sediment indicator taxon (Mormede et al. 2021), as well as using this approach to assess the impact on a taxon indicative of hard substratum habitat on the Chatham Rise. The utility of all four methods is evaluated below. Such an evaluation is conducted here because while the use of more sophisticated models, which include the population dynamics of the benthos and explicitly depend on data from New Zealand (e.g., Mormede et al. 2021), might be preferred (Ford et al. 2016), they have a greater data requirement and may not be applicable to large spatial scales. Issues common to all or most of the four methods are highlighted first.

4.1 Trawl data inputs

The quality and precision of reported catch and effort data has improved considerably since 1989, particularly in the inshore fleet since 2007–08, with increasing incorporation of GPS to record positions and updated forms to better accommodate this requirement. The recent introduction of the Electronic Reporting System (ERS) for capturing these data now allows high precision location data over much of the extent of New Zealand fisheries, and there is continued collection of key information on gear type and width of the gear used. Future assessments of bottom contact by fishing gear should consider whether commercial effort data could be used more directly to estimate individual trawl footprints where possible, with less reliance on assignment of tows into the broad categories applied here. To assess whether this is worthwhile, it may also be useful to examine the difference in estimated bottom contact between the current method and one based on high-precision ERS location data and recorded gear parameters, within a small test case area.

The 1×1 km resolution used for the impact assessments is a conservative reflection of the overall precision of the reported effort locations over time, with much greater precision available in recent data. However, the assumptions used in the data grooming process, including jittering of rounded values and coarse estimation of tow end positions, may result in fishing effort assigned to locations during years in which these locations were closed to fishing. Such areas include Benthic Protection Areas, seamount closures, marine reserves, and cable exclusion zones, with variable dates of first closure. As such, when using the spatially gridded benthic impact data layers produced here for marine spatial planning (e.g., to identify priority areas for conservation/fisheries management using decision-support tools such as Zonation; Geange et al. 2017), these fishing exclusion zones should be included both temporally as well as spatially to set impacts to zero in grid cells where necessary. For a full account of the underlying assumptions and uncertainty in the trawl data used in the analyses see Baird & Mules (2021).

4.2 Trawl footprint estimates

In this project, the proportion of bottom contact by particular components of the bottom trawls (ground gear doors, bridles, sweeps) was based partly on a previous expert workshop (Mormede et al. 2017) and also on a Delphi survey of NIWA experts carried out for this study in order to extend this information to the complete range of gear categories used in the analysis. While expert-derived information has been used previously to determine bottom contact of trawl gears used in Europe (Eigaard et al. 2016), such information has inherent issues (e.g., subjectivity, bias, non-repeatability), and these cannot be removed even via a Delphi process. There are very few published studies available that have determined quantitative data on the extent and duration of bottom contact by trawl gear components during commercial, research, or experimental trawling (e.g., Ivanovic et al. 2011, Depestele et al. 2016), and no similarly quantitative studies have been carried out for trawl gear used in New Zealand fisheries. Such studies should be carried out in the future in order to improve estimates of bottom trawl seafloor contact for the determining assessments of benthic impact by this gear. Bottom contact from midwater trawl gear is likely to be particularly poorly estimated, due to the definition used for identifying such tows that were likely to have contacted the seafloor, and the lack of data supporting the estimates of percentage bottom contact time made by experts in our survey. The dataset of trawl positions used in our study (from Baird & Mules 2021) considered any trawl using a midwater net within 1 m of the seafloor as a bottom trawl. These trawls are possible to identify because fishers record both bottom depth and gear depth on catch and effort forms, but it may be useful to investigate the appropriateness of this distance as midwater trawls fished further off the seafloor may also sometimes have bottom contact. This potential contact could be assessed by examining the reported or observed catch of midwater trawls fished at different heights off the seafloor, in particular looking for species, e.g., flatfish, invertebrates, that are unlikely to be caught in midwater.

As in the study of Mormede et al. (2017), the Mercator projection was applied for the calculation of the cumulative proportional footprint and swept area ratio for the MSRP and RBS methods for assessing benthic impact, and therefore the cell size changes with latitude. Following that methodology, we also applied an average cell area for the study region to minimise the bias introduced from this inconsistency. Future implementations of this methodology should attempt to incorporate a variable cell size in the calculations, or an alternative projection such as the Albers equal area projection (EPSG:9191, Wood et al. 2022), to avoid this bias.

4.3 Estimates of depletion/recovery and catchability for benthic taxa

Estimates of mortality/depletion and recovery used in the benthic impact assessment methods presented here were: either expert-derived (MSRP, population-based model for holothurians); from meta-analysis of studies conducted in a range of environments outside the New Zealand region, for a range of gear types, and for a limited number of benthic taxa (RBS, modified RBS); or simulation-derived (population-based model for anthozoans). Only some of these values of depletion and recovery had estimates of uncertainty around them. Considering the fundamental importance of these parameters for the benthic impact assessment methods used here, it is obviously far from ideal that there is no depletion/recovery data for benthic taxa derived from studies of New Zealand-specific trawling and habitats. Furthermore, estimates of trawl catchability of benthic taxa are generally lacking, and this trawl efficiency parameter is also important for improving population-based methods for assessing benthic impact.

4.4 Benthic bycatch data

The application of the modified RBS and population-based methods is dependent on benthic invertebrate bycatch data. Such data have been collected from research surveys and by observers on commercial vessels since the early 1990s but, importantly, inshore commercial vessels have been poorly sampled due to the lack of observer coverage in these fisheries. It is notable that the taxonomic resolution and reliability of these data improved from 2008 with the introduction of the Benthic Bycatch Form and improved identification guides (Bowden et al. 2015), although these guides primarily cover deepwater species. Thus data available for developing the modified RBS approach here were restricted to the last 10 years only (2008–2019); a longer term dataset would have been useful for more reliably deriving model parameters and estimating the impact of trawling on benthic biomass. Clearly the utility of biomass-based impact assessment methods in the future relies on the continued and regular collection of benthic invertebrate bycatch data: the continued existence of supporting processes that influence its quality (e.g., observer training, identification guides, quality assurance checks on taxonomic identifications), extending observer coverage and supporting identification guides into the inshore fleet, and the accessibility of the data (maintenance of the *cod* database). Note that the regularity of the research trawl surveys funded by Fisheries New Zealand and the fishing industry, the primary source of benthic invertebrate bycatch data, is no longer ensured as it was previously.

4.5 MSRP and RBS methods applied at TS and EEZ scale

The utility of MSRP method for assessing the potential impact of trawling on benthic fauna, or the 'naturalness' of the seafloor where these fauna reside, has already been demonstrated in the management of fishing in the CCAMLR and SPRFMO convention areas. For example, a spatial data layer for naturalness derived using the MSRP method was used with spatial predictions of habitat suitability for vulnerable marine ecosystem indicator taxa (i.e., taxa that were 'large, erect, hard, sessile') to proportionally discount the conservation priority value of those grid cells where suitable habitat for those taxa was predicted to occur but which was previously subjected trawling. This analysis was undertaken within the decision-support tool Zonation, and along with other data inputs, was used to identify suitable areas for VME protection/closure to fishing in part of the SPRFMO Convention Area (SPRFMO 2020). This project has now produced data layers that can be used for assessing the impact of bottom trawling in New Zealand's TS and EEZ, as well as practically expanding the utility of this method by producing naturalness data layers for the two functional groups of benthic taxa ('small, fragile, encrusting' and 'deep, burrowing infauna') so far not produced for previous management processes elsewhere. Furthermore, the application of this method has been expanded to include an assessment of impacts from more than one gear/fishery type. Similar approaches to determining the impact of fishing on benthic fauna in Europe have also used categorisations of fishing gear; however, these have typically used a smaller number of categories (e.g., 14 categories or "métiers", Eigaard et al. 2017). The level of gear categorisation achieved here presents the opportunity to achieve a nuanced understanding of the potential impact of fishing in the New Zealand TS and EEZ. It is now possible to understand where particular gear categories are likely to impact particular functional groups of benthic fauna to a greater or less extent. Areas of relatively high naturalness, even within the overall fishing footprint, can also be identified across all faunal groups.

The utility of the RBS method has already been demonstrated for parts of the New Zealand region in a recent global study to assess relative status of benthic communities in 24 regions where sufficient data were available (Pitcher et al. 2022). While here we used the general RBS approach of Pitcher et al. (2017), significant differences in the input data were made from this first application of the method and the global analysis. That is, for the present study we followed the implementation of the RBS method used in the recent SPRFMO analysis (SPRFMO 2020), which rather than using depletion/recovery rates for individual gears/habitats instead used such rates for a range of VME indicator taxa. As for the MSRP approach, here the number of gear categories was greatly expanded from previous applications of the RBS approach. However, it was not possible to account for the varying impacts on the benthos from the range of ground gear types in use (i.e., rubber disks, rockhopper, bobbins, chain, wire/chain) in the calculation of the SAR used in the RBS method. Although there are many differences between the previous application of RBS to areas of the New Zealand EEZ (Pitcher et al. 2022) and the present study, as noted above, a generally high level of status for the New Zealand region was common to both studies, with fished areas within the EEZ dominated by a status greater than 80% of the pre-fishing level. Likewise, few comparisons of our results can be made with the original RBS study (Pitcher et al. 2017) which was specific to a region in Western Australia; however, this agreed with our study in illustrating very low status in highly fished areas, but high status (approaching 80–100%) in areas in which fishing levels were below that at which recovery of benthic taxa may be severely compromised.

There are fundamental differences between the MSRP and RBS approaches used here for assessing the impact of bottom trawling on benthic fauna. As implemented, the MSRP method does not allow for recovery over time, therefore it will lead to an underestimate current naturalness to some degree, whereas the RBS method, although accounting for recovery, estimates an equilibrium state based on assumptions about the level of future trawling that may lead to an underestimate of naturalness if future trawling intensity decreases, or vice versa. It should be emphasised that without combination with a predictive spatial data layer for the habitat suitability/abundance of a benthic taxon or community, the outputs created here for both methods represent only the potential for impact or the naturalness of the particular functional group/VME indicator taxon *if* it occurs there.

The suitability of the use of MSRP and RBS data layers may depend on their intended application. However, for the general purposes of spatial management of impacts from fishing, it is useful to have both data layers available for use. For example, both data layers were used in a complementary manner in the SPRFMO management process (SPRFMO 2020).

4.6 Enhanced modification of the RBS approach at the local scale

The RBS approach has much to recommend it as a data-limited method for assessing the impact of fishing on benthic communities at broad spatial scales (Pitcher et al. 2022). However it can be further improved, and made relevant for local benthic fauna, by directly estimating vulnerability parameters (and, potentially, recovery rates, depending on the time and spatial scales of observations) from observer data (Zhou et al. 2014, Neubauer et al. 2019). This project attempted to enhance the RBS approach in this way and to examine some of the assumptions underlying the approach using data from the Chatham Rise for test simulations.

Estimating some of the parameters used for RBS from spatial trends in catch and effort from the Chatham Rise appeared to hold some promise, in that spatial patterns in depletion were relatively well estimated in simulation experiments. However, especially in situations with relatively short time series that allow for little contrast, and for fishing patterns where areas of highest equilibrium density are preferentially targeted, estimates of the scale of the focal taxa populations were driven by estimates from areas with more sporadic effort/ areas with lower density, leading estimated population scale to be biased low. Consequently, to fit observed catches, productivity was biased high in the models. This discrepancy explains why the model performed best for fishing patterns where effort distribution was inversely proportional to equilibrium biomass. Here the scale of the unfished population is maintained in areas of highest density and low effort, leading to smaller bias in the estimated unfished population size.

Overall, such biased outcomes are well known to exist in stock assessments with relatively short and/or stable series of catch and effort, which provide too little information to estimate the scale of the population reliably. In addition, in Bayesian assessments with poorly informative data, the determination of suitable non-informative priors remains an active topic of research (Thorson & Cope 2018, Kim & Neubauer in prep.). Given that the spatial effort pattern strongly determines the degree to which the estimates of stock scale, and consequently stock productivity, are biased, priors may provide a way to better delimit the problem; e.g., early surveys or video surveys of closed areas may provide sufficient data for average unfished population scales, provided they cover comparable habitats.

In addition, the simulations used a single set of life history and parameter (e.g., observation error) values. The relative importance of factors affecting estimation accuracy would need to be explored in a larger simulation experiment with a factorial design that could include additional factors of possible importance, such as grid size in estimation models, as well as estimation model formulations (e.g., separate catchabilities vs. model-based standardisation of catchability). Such an experiment would be computationally intensive given the need to simulate, and subsequently apply the estimation model, to a potentially large number of scenarios. Although such a study was deemed to be beyond the scope of the present project, such a study could deliver valuable insights into the performance of model-based estimates of benthic status.

The simulation work may have underestimated the performance of the model in settings where multiple gears operate simultaneously in a given area. The contrasting catch rates from these gears provide an avenue to estimate trawl depletion rates (Zhou et al. 2013, Zhou et al. 2014, Neubauer et al. 2019, Edwards 2021). This mechanism was not explored in the simulations undertaken here, but such simulations would be useful to further investigate the utility of the present method for impact assessments of benthic bycatch. Indeed, the application of the spatial surplus production model to Chatham Rise anthozoan bycatch suggested that the model can identify differential depletion rates when co-occurring gears fish similar areas. This application also did not show patterns seen in biased simulation outcomes, namely that areas of high bias (i.e., overestimated depletion status) were associated with estimates of low equilibrium biomass due to missing information on previous scales of these populations. On the Chatham Rise, estimates of highest equilibrium biomass were co-located with areas of high effort (i.e., simulation scenario 1), providing additional evidence that the present simulations may have been overly pessimistic. Further work should be undertaken to understand in what circumstances the estimation model performs well.

The present application of the modified RBS approach to broad taxonomic groupings includes a large number of species and serves here only as a test ground for the estimation model. Estimated benthic status for Anthozoa, for example, will likely underestimate impacts on more vulnerable species, such as branching stony coral. Therefore, careful delineation of species groups would provide a clearer picture of the status of those with varying life histories. Such applications could also include additional information about species distributions, either from model-based estimates or by including survey data and environmental covariates directly in the estimation model. In addition, models could be built to operate on a number of species that are thought to share comparable vulnerability and/or are similarly affected by fishing gear. While fishing effort was not standardised beyond the gear categorisation (i.e., estimating different catchability for different gear categories), future applications of these models could also examine the potential to improve model performance and reduce noise by standardising fishing effort further, or by using model-based standardisation rather than applying independent catchability estimates in the model.

Overall, the modified RBS method displayed promising utility, but further development is required (as outlined above) before this enhanced approach can be implemented more widely and with confidence. For this method to be applied to the entire TS/EEZ, similar temporal bycatch data that were available for the highly sampled/fished Chatham Rise would be required to develop d and R values. Nonetheless, where it was possible to visually compare spatial patterns for benthic impact determined by RBS (at the TS/EEZ scale) and the modified RBS on the Chatham Rise, these were similar even though the former did not incorporate any bycatch catch or distributional data while the latter used actual benthic bycatch biomass data (compare Sponge in Figure 27 with demosponges/hexactinellid in Figures 49d and 53d).

4.7 VAST-based modelling approach

Vector autoregressive spatio-temporal (VAST) modelling of benthic bycatch data combined with trawl footprint data was used to update the previous assessment of trawling impact on holothurians (as an indicator of soft-sediment habitat) on the Chatham Rise. The additional 3 years of data for this analysis indicated that the benthic impact, for the two types of trawling assessed, remains consistent with the patterns in previous years and is very low throughout much of the Chatham Rise, with higher values only on the central-north edge of the rise (maximum of 9.56% of holouthurian biomass impacted). The same approach was extended here to quantify the impact on anthozoans as indicator taxon for hard substratum habitat on the rise. This analysis found that benthic impact on the Chatham Rise was relatively low overall, which is not surprising given the limited available hard substratum habitat for anthozoans on the rise. However, the analysis identified that the spatial distribution of relatively higher areas of impact (though still small; maximum value of 1.87% of anthozoan biomass impacted) did vary temporally and between the two types of trawling assessed. Where it was possible to visually compare patterns for benthic impact determined by this population model-based approach and the modified RBS approach, spatial patterns of relative benthic impact were similar (compare Figures 65 and 67 with

Figure 44 for anthozoans). This is perhaps not surprising because both methods are based on benthic bycatch biomass data.

Beyond providing a complementary assessment for hard substratum habitat, the partial application of the Mormede et al. (2021) approach for quantifying spatially and temporally explicit estimates of impact on benthic fauna provided no development of the approach per se. Therefore, the issues and limitations of the previous test case of Mormede et al. (2021) remain. Some of these are highlighted above in the issues common to most of the methods employed in this study; for example, the availability, quality, and quantity of benthic bycatch data, and the determination of site- and taxa-specific estimates of depletion which are essential to construct reliable population model-based approaches to assessing benthic impact from fishing. See Mormede et al. (2021) for further details on how this approach can be further developed in the future to improve its utility.

5. POTENTIAL RESEARCH

The present study has produced outputs from the application of two published methods, MSRP and RBS, that can be used to assess the impact to benthic fauna from bottom trawling. These data layers can now be used to inform risk assessments and spatial management planning processes across New Zealand's TS and EEZ, including for fisheries management. The study has also demonstrated that modifications to the RBS approach can improve its local utility, although further development of this enhanced approach is still required. Similarly, further development of the population model-based approach considered by this study is also warranted.

There were a number of data-related issues identified and re-identified by this study that serve as the basis for a number of recommendations that, if addressed, could lead to significant future improvements in benthic impact assessments. These are:

- Collect/use spatial data for dredge deployments at same resolution as for trawls.
- Where possible, use high-precision trawl data collected by the Electronic Reporting System to inform future benthic impact assessments.
- Undertake studies to determine bottom contact by different trawl gear components.
- Undertake studies to determine depletion/recovery and catchability values for benthic fauna in a New Zealand context.
- Ensure the continuation of the collection of benthic bycatch data, at the lowest taxonomic level possible, from regular and frequent research trawl surveys.
- Improve observer coverage of inshore fisheries, or otherwise improve collection of benthic bycatch data from these fisheries.
- Develop identification guides for inshore benthic invertebrates.

6. ACKNOWLEDGEMENTS

This work was completed Fisheries New Zealand projects BEN2019-04 and BEN2019-04B. Thanks are given to the following fishers, vessel managers, and fisheries experts for information on fishing gear types and configurations: Andrew Heatherington, Justin Rilstone, Curly Brown, Rex Chapman, (vessel skippers), Mike Gibson (Sealord Group), Aaron Frazer (Sanford Ltd), Richard Wells, (Deepwater Group Ltd.), Carrol Collier (Hampidjan), Carol Scott (FINZ), Owen Hoggard, Glen Curtis, Andrew Hope (Motueka Nets). Thanks are similarly given to the NIWA staff that provided their input to the Delphi survey concerning the assessment of the bottom contact of different trawl gear components, and the impact of this gear on benthic fauna: Owen Anderson, David Bowden, Mike Beentjes, Malcolm Clark, Matt Dunn, Niki Davey, Mark Fenwick, Jason Hamill, Daniel Leduc, Dan MacGibbon, Sadie Mills, Richard O'Driscoll, Ashley Rowden, Dianne Tracey, and Darren Stevens.

Helpful feedback on this work and various drafts of the report was provided by members of the AEWG and Fisheries New Zealand staff. Finally, special thanks are given to Karen Tunley of Fisheries New Zealand for her patient and considered management of this project.

7. REFERENCES

Amoroso, R.O.; Pitcher, C.R.; Rijnsdorp, A.A. et al. (2018). Bottom trawl fishing footprints on the world's continental shelves. P*roceedings of the National Academy of Sciences 115 (43)*: E10275-E10282. doi.org/10.1073/pnas.1802379115

Anderson, O.F.; Finucci, B. (2022). Non-target fish and invertebrate catch and discards in New Zealand orange roughy and oreo trawl fisheries from 2002–03 to 2019–20. *New Zealand Aquatic Environment and Biodiversity Report No. 282*. 117 p.

Anderson, O.F.; Finucci, B.; Edwards, C.T.T. (2023). Non-target fish and invertebrate catch and discards in New Zealand arrow squid and scampi trawl fisheries from 2002–03 to 2020–21. *New Zealand Aquatic Environment and Biodiversity Report No. 315*. 167 p.

Baird, S.J.; Bagley, N.W.; Wood, B.A.; Dunn, A.; Beentjes, M. (2002). The spatial extent and nature of mobile bottom fishing methods within the New Zealand EEZ, 1989–90 to 1998–99. Final Research Report for Objective 1 of Ministry of Fisheries project ENV2000/05. (Unpublished report held by Ministry for Primary Industries, Wellington.)

Baird, S.J.; Hewitt, J.E.; Wood, B.A. (2015). Benthic habitat classes and trawl fishing disturbance in New Zealand waters shallower than 250 m. *New Zealand Aquatic Environment and Biodiversity Report No. 144*. 184 p.

Baird, S.J.; Mules, R. (2021). Extent of bottom contact by commercial trawling and dredging in New Zealand waters, 1989–90 to 2018–19. *New Zealand Aquatic Environment and Biodiversity Report No. 260*. 157 p.

Baird, S.J.; Wood, B.A. (2018). Extent of bottom contact by New Zealand commercial trawl fishing for deepwater Tier 1 and Tier 2 target fish stocks, 1989–90 to 2015–16. *New Zealand Aquatic Environment and Biodiversity Report No. 193*. 102 p.

Baird, S.J.; Wood, B.A.; Bagley, N.W. (2011). Nature and extent of commercial fishing effort on or near the seafloor within the New Zealand 200 n. mile Exclusive Economic Zone, 1989–90 to 2004–05. *New Zealand Aquatic Environment and Biodiversity Report No. 259*. 143 p.

Beentjes, M.P.; Baird, S.J. (2004). Review of dredge fishing technologies and practice for application in New Zealand. *New Zealand Fisheries Assessment Report 2004/37*. 40 p

Bowden D.A.; Clark, M.R.; Hewitt, J.E.; Rowden, A.A.; Leduc, D.; Baird, S.J. (2015). Designing a programme to monitor trends in deep-water benthic communities. *New Zealand Aquatic Environment and Biodiversity Report No. 143*. 61 p.

Carpenter, B.; Hoffman, M.D.; Brubaker, M.; Lee, D.; Li, P.; Betancourt, M. (2015). The Stan Math Library: Reverse-Mode Automatic Differentiation in C++. *arXiv*: 1509.07164 [cs.MS]. URL https://arxiv.org/abs/1509.07164

Clark, M.; Tracey, D.; Anderson, O.; Parker, S. (2014). Pilot ecological risk assessment for protected corals. (NIWA Client Report WLG2014-70 prepared for Department of Conservation.)

Clark, M.R.; Althaus, F.; Schlacher, T. A.; Williams, A.; Bowden, D. A.; Rowden, A. A. (2016a). The impacts of deep-sea fisheries on benthic communities: a review. *ICES Journal of Marine Science 73*: i51–i69. doi: 10.1371/journal.pone.0022588

Clark, M.R.; Bagley, N.W.; Harley, B. (2016b). Trawls. Chapter 7. *In*: Clark, M.R.; Consalvey, M.; Rowden, A.A. (eds), pp. 126–158. *Biological sampling in the deep se*a. Wiley Blackwell, Oxford, UK.

Clark, M.R.; Rowden, A.A. (2009). Effect of deep water trawling on the macro-invertebrate assemblages of seamounts on the Chatham Rise, New Zealand. *Deep Sea Research I 56*: 1540–1554.

Clement and Associates (2008). New Zealand inshore trawl gear and operations survey. A report commissioned by Seafood Innovations and the Seafood Industry Council.

Cryer, M; Hartill, B.; O'Shea, S. (2002). Modification of marine benthos by trawling: toward a generalization for the deep ocean? *Ecological Applications 12 (6)*: 1824–1839.

Depestele, J.; Ivanovic´, A.; Degrendele, K.; Esmaeili, M.; Polet, H.; Roche, M.; Summerbell, K.; Teal, L. R.; Vanelslander, B.; O'Neill, F. G. (2016). Measuring and assessing the physical impact of beam trawling. *ICES Journal of Marine Science 73*: i15–i26.

Eayrs, S.; Craig, T.; Short, K. (2020). Mitigation techniques to reduce benthic impacts of trawling. a review for the Department of Conservation by Terra Moana Limited.

Edwards, C.T.T. (2021). Integrated estimation of density and catchability parameters from fisheries catch-effort data. *New Zealand Fisheries Assessment Report 2021/32*. 32 p.

Eigaard, O.R.; Bastardie, F.; Breen, M.; Dinesen, G.E.; Hintzen, N.T.; Laffargue, P.; Mortensen, L.O.; Nielsen, J.R.; Nilsson, H.C.; O'Neill, F.G.; Polet, H. (2016). Estimating seabed pressure from demersal trawls, seines, and dredges based on gear design and dimensions. *ICES Journal of Marine Science 73*(suppl_1): i27-i43.

Eigaard, O.R.; Bastardie, F.; Hintzen, N.T.; Buhl-Mortensen, L.; Buhl-Mortensen, P.; Catarino, R.; Dinesen, G.E.; et al. (2017). The footprint of bottom trawling in European waters: distribution, intensity, and seabed integrity. *ICES Journal of Marine Science 74*: 847–865.

Ellis, N.; Pantus, F.J.; Pitcher, C.R. (2014) Scaling up experimental trawl impact results to fishery management scales – a modeling approach for a 'hot time'. *Canadian Journal of Fisheries & Aquatic Science 71*: 1– 14.

Finucci, B.; Anderson, O.F.; Edwards, C.T.T. (2022). Non-target fish and invertebrate catch and discards in New Zealand jack mackerel trawl fisheries from 2002–03 to 2018–19. *New Zealand Aquatic Environment and Biodiversity Report No. 279*. 81 p.

Ford, R.B.; Arlidge, W.N.S.; Bowden, D.A.; Clark, M.R.; Cryer, M.; Dunn, A.; Hewitt, J.E.; Leathwick, J.R.; Livingston, M.E.; Pitcher, C.R.; Rowden, A.A.; Thrush, S.F.; Tingley, G.A.; Tuck, I.D. (2016). Assessing the effects of mobile bottom fishing methods on benthic fauna and habitats. *New Zealand Fisheries Science Review 2016/2*. 47 p.

Ford, R.B.; Galland, A.; Clark, M.R.; Crozier, P.; Duffy, C.A.J.; Dunn, M.R.; Francis, M.P.; Wells, R. (2015). Qualitative (Level 1) Risk Assessment of the impact of commercial fishing on New Zealand Chondrichthyans. *New Zealand Aquatic Environment and Biodiversity Report No. 157*. 103 p.

Geange, S.W.; Leathwick, J.; Linwood, M.; Curtis, H.; Duffy, C.; Funnell, G.; Cooper, S. (2017). Integrating conservation and economic objectives in MPA network planning: A case study from New Zealand. *Biological Conservation 210*: 136–144.

Grieve, C.; Brady, D.C.; Polet H. (2015). Best practices for managing, measuring and mitigating the benthic impacts of fishing – Part 1. *Marine Stewardship Council Science Series 2*: 18–88.

Harley, B.; Ellis, J. (2007). The Modified GOV and Ground gear 'D'. ICES Theme Session Q: Science underpinning stock abundance survey practice. ICES CM/Q:02. https://doi.org/10.17895/ices.pub.25258363.v1

Hartig, F. (2020). DHARMa: residual diagnostics for hierarchical (multi-level/mixed) regression models. R package v. 0.2. 0

Hiddink, J.G.; Jennings, S.; Sciberras, M.; Szostek, C.L.; Hughes, K.M.; Ellis, N.; et al. (2017). Global analysis of depletion and recovery of seabed biota following bottom trawling disturbance. *Proceedings of the National Academy of Sciences of the United States of America 114*: 8301–8306.

Hobday, A.J.; Smith, A.D.M.; Stobutzki, I.C.; Bulman, C.; Daley, R.; Dambacher, J.M.; Deng, R.A.; Dowdney, J.; Fuller, M.; Furlani, D.; Griffiths, S.P.; Johnson, D.; Kenyon, R.; Knuckey, I.A.; Ling, SD.; Pitcher, R.; Sainsbury, K.J.; Sporocic, M.; Smith, T.; Walker, T.I.; Wayte, S.E.; Webb, H.; Williams, A.; Wise, B.S.; Zhou, S. (2011). Ecological risk assessment for the effects of fishing. Fisheries Research, 108: 372–384.

Hurst, R.J.; Bagley, N.; Chatterton, T.; Hanchet, S.; Schofield, K.; Vignaux, M. (1992). Standardisation of hoki/middle depth time series trawl surveys. MAF Fisheries Greta Point Internal Report No. 194. 89 p. (Unpublished report held by NIWA library, Wellington.)

Ivanovic´, A.; Neilson, R.D.; O'Neill, F.G. (2011). Modelling the physical impact of trawl components on the seabed and comparison with sea trials. *Ocean Engineering 38*: 925–933.

Jennings S; Kaiser M. J. (1998). The effects of fishing on marine ecosystems. *In*: Blaxter, J.H.S.; Southward, A.J.; Tyler P.A. (eds.), pp. 201–212+. *Advances in Marine Biology Vol 34*. Academic Press Ltd., Elsevier Science Ltd, London.

Kim, K.; Neubauer, P. (in prep). The importance of prior-pushforward checks for Bayesian stock assessments. (Unpublished work from Dragonfly Data Science.)

Michael, K.P. (2009). Bluff Oyster Management Company oyster fishery logbook programme: A summary of data for the 2006 to 2009 oyster seasons. (NIWA Client Report: WLG2009-74.) 59 p.

Mormede, S.; Baird, S.J.; Roux, M.-J. (2021). Developing quantitative methods for the assessment of risk to benthic habitats from bottom fishing activities using the test case of holothurians on the Chatham Rise. *New Zealand Aquatic Environment and Biodiversity Report No. 274.* 38 p.

Mormede, S.; Dunn, A. (2013). An initial development of spatially explicit population models of benthic impacts to inform Ecological Risk Assessments in New Zealand deepwater fisheries. *New Zealand Aquatic Environment and Biodiversity Report No. 106*. 16 p.

Mormede, S.; Sharp, B.; Roux, M.J.; Parker, S. (2017). Methods development for spatially-explicit bottom fishing impact evaluation within SPRFMO: 1. Fishery footprint estimation. SC5-DW06. 5th Meeting of the Scientific Committee Shanghai, China, 23 - 28 September 2017.

Neubauer, P.; Richard, Y.; Tremblay-Boyer, L. (2019). Alternative assessment methods for oceanic whitetip shark. WCPFC-SC15-2019/SA-WP-13. Report to the Western and Central Pacific Fisheries Commission Scientific Committee. Fifteenth Regular Session, 12–20 August 2019, Pohnpei, Federated States of Micronesia.

Parker, S.J. (2008). Development of a New Zealand High Seas Bottom Trawling Bottom Fishery Impact Assessment Standard for Evaluation of Fishing Impacts to Vulnerable Marine Ecosystems in the South Pacific Ocean. (Fisheries Research Report, IFA2007-02, Ministry of Fisheries.)

Parker, S.J.; Penney, A.J.; Clark, M.R. (2009). Detection criteria for managing trawl impacts on vulnerable marine ecosystems in high seas fisheries of the South Pacific Ocean. *Marine Ecology Progress Series 397*: 309e317.

Pitcher, C.R.; Ellis, N.; Jennings, S.; Hiddinck, J.G.; Mazor, T.; Kaiser, M.J.; et al. (2017). Estimating the sustainability of towed fishing-gear impacts on seabed habitats: A simple quantitative risk assessment method applicable to data limited fisheries. *Methods in Ecology and Evolution 8*: 472–480.

Pitcher, C.R.; Hiddink, J.G.; Jennings, S.; Collie, J.; Parma, A.M.; et al. (2022). Trawl impacts on the relative status of biotic communities of seabed sedimentary habitats in 24 regions worldwide. *PNAS Vol. 119 No. 2*: e2109449119. https://doi.org/10.1073/pnas.2109449119

Richard, Y.; Abraham, E.R. (2015). Assessment of the risk of commercial fisheries to New Zealand seabirds, 2006–07 to 2012–13. *New Zealand Aquatic Environment and Biodiversity Report No. 162*. 85 p.

Rijnsdorp, A.D.; Bastardie, F.; Bolam, S.G.; Buhl-Mortensen, L.; Eigaard, O.R.; Hamon, K.G.; Hiddink, J. G.; Hintzen, N. T.; Ivanovic´, A.; Kenny, A.; Laffargue, P.; Nielsen, J.R.; O'Neill, F.G.; Piet, G.J.; Polet, H.; Sala, A.; Smith, C.; van Denderen, P.D.; van Kooten, T.; Zengin, M. (2016). Towards a framework for the quantitative assessment of trawling impact on the seabed and benthic ecosystem. *ICES Journal of Marine Science 73*: i127–i138.

Roux, M.-J.; Doonan, I.; Edwards, C.; Clark, M.R. (2017). Low information stock assessment of orange roughy *Hoplostethus atlanticus* in the South Pacific Regional Fisheries Management Organisation convention area. *New Zealand Fisheries Assessment Report 2017/01*. 62 p.

Rowden, A.A.; Oliver, M.; Clark, M.R.; Mackay, K. (2008). New Zealand's "SEAMOUNT" database: recent updates and its potential use for ecological risk assessment. *New Zealand Aquatic Environment and Biodiversity Report No. 27*. 49 p.

Schaefer, M.B. (1954). Some aspects of the dynamics of populations important to the management of commercial marine fisheries. *Bulletin of the Inter-American Tropical Tuna Commission 1*: 27–56.

Sharp, B.R.; Parker, S.J.; Smith, N. (2009). An impact assessment framework for bottom fishing methods in the CCAMLR area. *CCAMLR Science 16*: 195–210.

SPRFMO (2020). Cumulative Bottom Fishery Impact Assessment for Australian and New Zealand bottom fisheries in the SPRFMO Convention Area, 2020. SC8-DW07. 8th Meeting of the Scientific Committee, New Zealand, 3–8 October 2020. https://www.sprfmo.int/assets/2020-SC8/SC8-DW07-rev-1-Cumulative-Bottom-Fishery-Impact-Assessment-for-Australia-and-New-Zealand.pdf

Stephenson, F.; Rowden, A.A.; Anderson, O.F.; Pitcher, C.R.; Pinkerton, M.H.; Petersen, G.; Bowden, D.A. (2021). Presence-only habitat suitability models for vulnerable marine ecosystem indicator taxa in the South Pacific have reached their predictive limit. *ICES Journal of Marine Science 78(8)*: 2830–2843. https://doi.org/10.1093/icesjms/fsab162

Stevens, D.W.; O'Driscoll, R.L.; Ballara, S.L.; Schimel, A.C.G. (2021). Trawl survey of hoki and middle depth species on the Chatham Rise, January 2020 (TAN2001). *New Zealand Fisheries Assessment Report 2021/33*. 122 p.

Thorson, J.T. (2018). Three problems with the conventional delta-model for biomass sampling data, and a computationally efficient alternative. *Canadian Journal of Fisheries and Aquatic Sciences 75(9)*: 1369–1382.

Thorson, J.T. (2019). Guidance for decisions using the Vector Autoregressive Spatio-Temporal (VAST) package in stock, ecosystem, habitat and climate assessments. *Fisheries Research 210:* 143–161. doi:https://doi.org/10.1016/j.fishres.2018.10.013

Thorson, J.T.; Barnett, L.A.K. (2017). Comparing estimates of abundance trends and distribution shifts using single- and multispecies models of fishes and biogenic habitat. *ICES Journal of Marine Science 74(5)*: 1311–1321. doi:https://doi.org/10.1093/icesjms/fsw193

Thorson, J.T.; Cope, J.M. (2017). Uniform, uninformed or misinformed?: The lingering challenge of minimally informative priors in data-limited Bayesian stock assessments. *Fisheries Research 194*: 164–172.

Thorson, J.T.; Ianelli, J.N.; Larsen, E.A.; Ries, L.; Scheuerell, M.D.; Szuwalski, C.; Zipkin. E.F. (2016). Joint dynamic species distribution models: a tool for community ordination and spatio-temporal monitoring. *Global Ecology and Biogeography 25(9)*: 1144–1158. https://doi.org/10.1111/geb.12464

Thrush, S.F.; Hewitt, J.E.; Cummings, V.J.; Dayton, P.K. (1995). The impact of habitat disturbance by scallop dredging on marine benthic communities: What can be predicted from the results of experiments? *Marine Ecology Progress Series 129*: 141–150.

Thrush, S.F.; Hewitt, J.E; Cummings, V.J.; Dayton, P.K.; Cryer, M.; Turner, S.J.; Funnell, G.A.; Budd, R.G.; Milburn, C.J.; Wilkinson, M.R. (1998). Disturbance of the marine benthic habitat by commercial fishing: Impacts at the scale of the fishery. *Ecological Applications 8*: 866–879.

Tingley, G. (2014). An assessment of the potential for near-seabed midwater trawling to contact the seabed and to impact benthic habitat and Vulnerable Marine Ecosystems (VMEs). SPRFMO SC 02-10. *MPI Technical Paper No: 2014/30*. 11 p.

Welsford, D.C.; Ewing, G.P.; Constable, A.J.; Hibberd, T.; Kilpatrick, R. (2014). *Demersal fishing interactions with marine benthos in the Australian EEZ of the Southern Ocean: An assessment of the vulnerability of benthic habitats to impact by demersal gears*. Final report FRDC Project 2006/042. Australian Antarctic Division. 258 p.

Williams, J.R.; Hartill, B.; Bian, R.; Williams, C.L. (2014). Review of the Southern scallop fishery (SCA 7). *New Zealand Fisheries Assessment Report 2014/07*. 71 p.

Wood, B.; Finucci, B.; Black, J. (2022). A standardised approach for creating spatial grids for New Zealand marine environment and species data. *New Zealand Aquatic Environment and Biodiversity Report No. 288.* 6 p.

Zhou, S.; Daley, R.; Fuller, M.; Bulman, C.; Hobday, A.; Courtney, T.; Ryan, P.; Ferrel, D. (2013). *Era extension to assess cumulative effects of fishing on species*. Final Report on FRDC Project No. 2011/029. 134 p.

Zhou, S.; Klaer, N.L.; Daley, R.M.; Zhu, Z.; Fuller, M.; Smith, A. D. (2014). Modelling multiple fishing gear efficiencies and abundance for aggregated populations using fishery or survey data. *ICES Journal of Marine Science 71(9)*: 2436–2447.

APPENDIX 1: DETAILS FROM BAIRD & MULES (2021)

Vessel categories and door spread assignment used by Baird & Mules (2021) to determine the extent of bottom contact by trawl fishing in the New Zealand TS/EEZ.

Vessel categories:

A. Domestic vessel ≤ 28 m
B. Domestic vessel > 28 m and ≤ 46 m
C. Domestic vessel > 46 m and ≤ 82 m
D. Any vessel > 82 m

Assigned door spread values

- 70 m for category A vessels under 20 m in length, with a single net
- 100 m for category A vessels over 20 m (max. 28 m) in length, with single net
- 150 m for category B vessels, with a single net
- 50 m for scampi tows with two nets and 70 m for scampi tows using three nets for category A vessels
- 70 m for scampi tows with two nets and 90 m for scampi tows using three nets for category B vessels
- 150 m for all targets, except HAK/HOK/LIN/SWA, for category C vessels that used one net
- 200 m for category C vessels targeting HAK/HOK/LIN/SWA with a single net
- 400 m for category C vessels targeting HAK/HOK/LIN/SWA with two nets [bottom trawl, BT] and for a single category D vessel that used two nets [BT]
- 150 m for all category D BATM vessels [BT and midwater within 1 m seafloor]
- 200 m for remaining category D vessels with single net.

APPENDIX 2: DETAILS FROM DELPHI SURVEY

Respondents were asked to assess the impact (i.e. "proportion [%] damaged or destroyed") to different functional groups of benthic fauna (i.e., those that are large, erect, hard, and sessile (LEHS), small, flexible/encrusting (SFE), and deep-burrowing infauna (DBI)) of different ground gear types (e.g., bobbin rigged ground gear). Respondents were also asked to assess the percent bottom contact time for different parts of midwater trawls. A spreadsheet was provided to record answers, with an illustration of each of the ground gear types that are used for bottom trawling in New Zealand.

Bottom contact for Midwater trawl gear components

	% time in bottom contact	**% time in bottom contact**	**% time in bottom contact**	**Expert**
Trawl type	Door furrow/wing-end weights	Sweep/bridle	Ground gear	
Midwater trawl	8	3	23	DB
Midwater trawl	10	10	10	DT
Midwater trawl	0	0	5	MD
Midwater trawl	25	13	13	MF
Midwater trawl	0	35	100	DM
Midwater trawl	10	10	10	OA
Midwater trawl	23	5	40	RO
Midwater trawl	5	5	10	MC

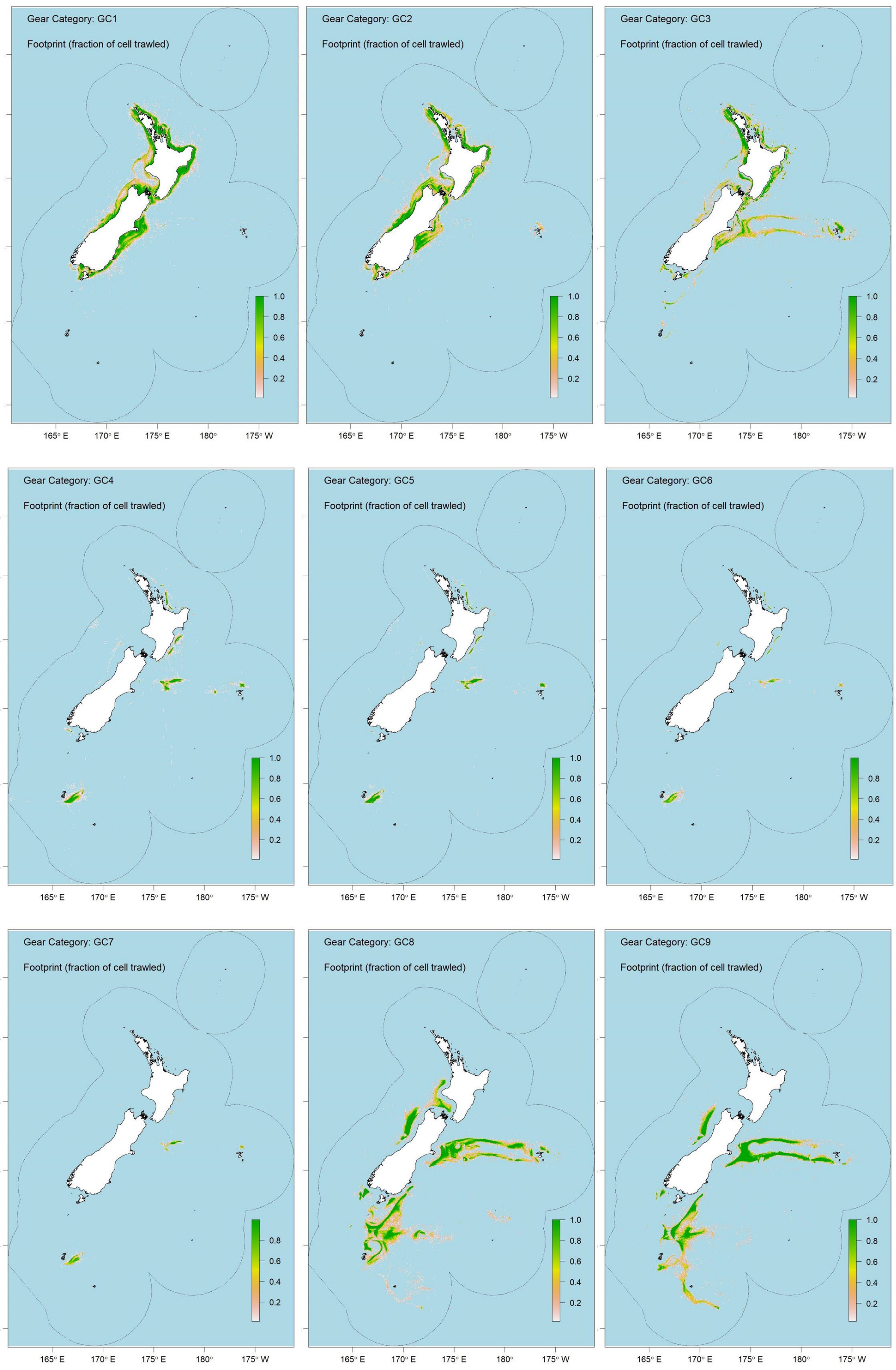
Ground gear impact on benthic fauna

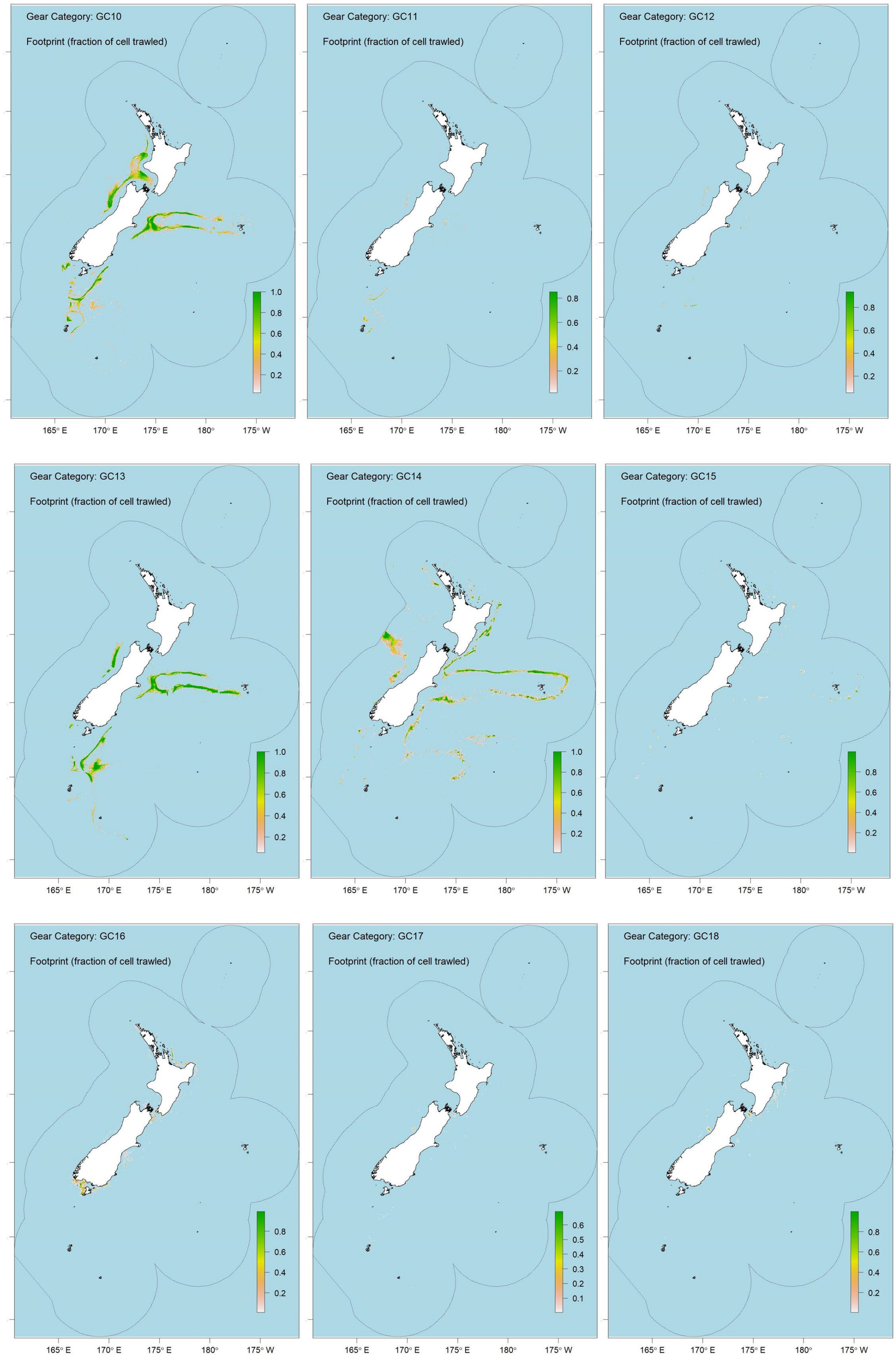
	% impact (i.e., "proportion [%] damaged or destroyed") of Ground gear	**% impact of Ground gear**	**% impact of Ground gear**	**Expert**
Ground gear type	**Large, erect, hard, sessile**	**Small, flexible/encrusting**	**Deep-burrowing infauna**	**Expert**
Bobbin	95	40	5	DB
Rockhopper	95	40	5	DB
Rubber discs (Cookies)	95	25	2.5	DB
Chain	95	65	2.5	DB
Wire	95	65	2.5	DB
Bobbin	95	40	5	DL
Rockhopper	95	40	5	DL
Rubber discs (Cookies)	95	40	5	DL
Chain	75	25	2.5	DL
Wire	75	25	2.5	DL
Bobbin	95	40	5	DT
Rockhopper	95	40	5	DT
Rubber discs (Cookies)	80	20	5	DT
Chain	40	10	0	DT
Wire	40	10	0	DT
Bobbin	70	45	15	JH
Rockhopper	95	55	25	JH
Rubber discs (Cookies)	70	25	5	JH
Chain	75	35	15	JH
Wire	65	35	15	JH
Bobbin	80	30	5	MD

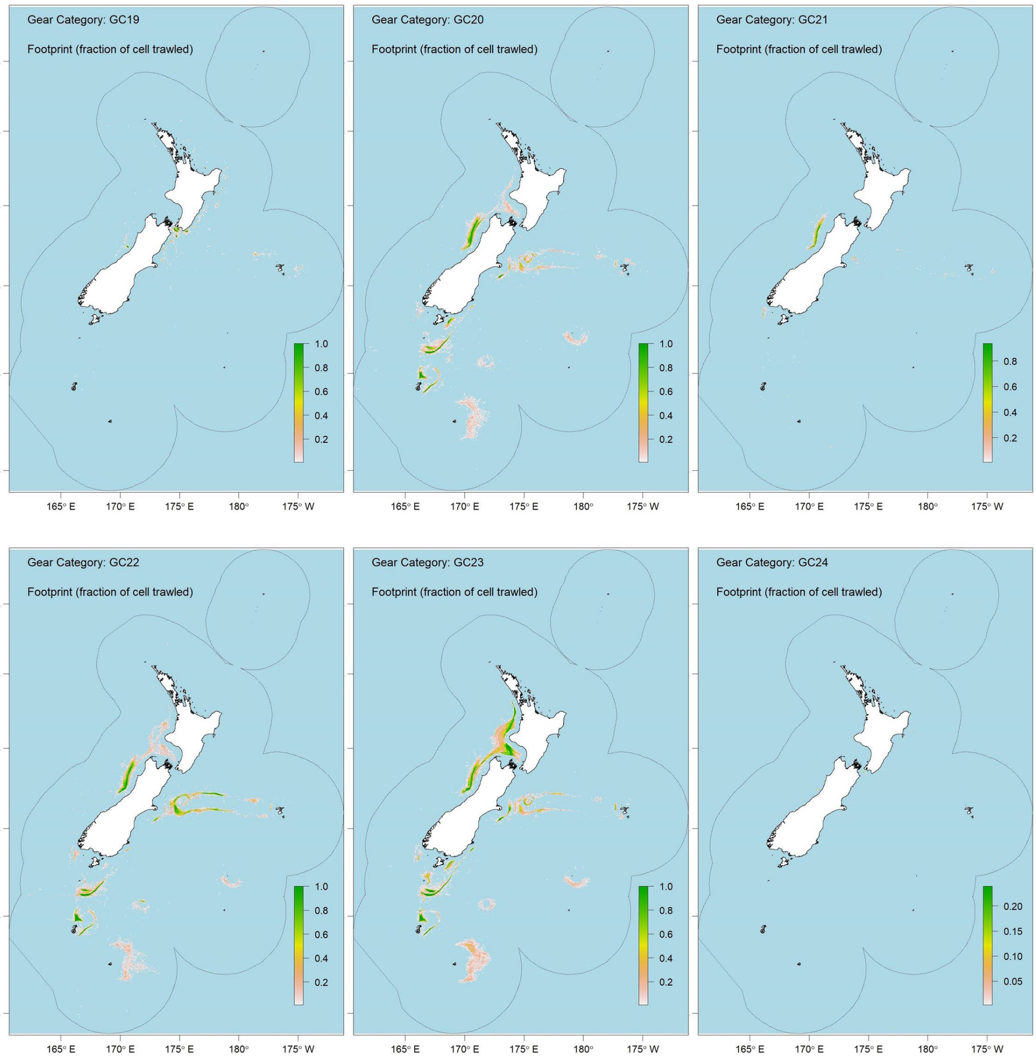
Rockhopper	95	40	5	MD
Rubber discs (Cookies)	100	95	5	MD
Chain	100	100	40	MD
Wire	100	100	40	MD
Bobbin	97.5	65	27.5	MF
Rockhopper	95	40	15	MF
Rubber discs (Cookies)	75	45	45	MF
Chain	75	75	22.5	MF
Wire	55	75	22.5	MF
Bobbin	95	15	2.5	SM
Rockhopper	95	40	2.5	SM
Rubber discs (Cookies)	95	40	2.5	SM
Chain	95	65	5	SM
Wire	95	65	5	SM
Bobbin	85	35	5	DM
Rockhopper	95	40	5	DM
Rubber discs (Cookies)	85	40	5	DM
Chain	75	40	2.5	DM
Wire	75	40	2.5	DM
Bobbin	95	40	25	DS
Rockhopper	95	40	15	DS
Rubber discs (Cookies)	100	95	10	DS
Chain	100	95	10	DS
Wire	100	95	10	DS
Bobbin	85	30	2.5	MB
Rockhopper	95	40	5	MB
Rubber discs (Cookies)	95	30	2.5	MB
Chain	95	30	2.5	MB
Wire	95	30	2.5	MB
Bobbin	90	50	5	ND
Rockhopper	55	40	5	ND
Rubber discs (Cookies)	50	30	0	ND
Chain	50	20	0	ND
Wire	40	20	0	ND
Bobbin	95	50	5	OA
Rockhopper	95	40	5	OA
Rubber discs (Cookies)	97.5	60	2.5	OA
Chain	97.5	60	2.5	OA
Wire	97.5	60	2.5	OA
Bobbin	95	40	5	RO
Rockhopper	80	20	5	RO
Rubber discs (Cookies)	85	20	5	RO
Chain	100			RO
Wire	100			RO
Bobbin	90	40	7.5	AR
Rockhopper	80	40	7.5	AR
Rubber discs (Cookies)	95	65	10	AR
Chain	95	75	15	AR
Wire	95	75	15	AR
Bobbin	85	40	10	MC
Rockhopper	75	30	7	MC
Rubber discs (Cookies)	90	35	0	MC
Chain	100	60	8	MC
Wire	100	50	5	MC

APPENDIX 3: CUMULATIVE PROPORTIONAL TRAWL FOOTPRINTS BY GEAR CATEGORY

Spatial extent of bottom contact (cumulative proportional footprint) by different trawl gear categories for the period 2007–08 to 2018–19 (inshore fisheries) and 1989–90 to 2018–19 (deepwater fisheries), within the New Zealand TS/EEZ. Map resolution (cell size) is 1 × 1 km. See Table 5 for gear category descriptions.

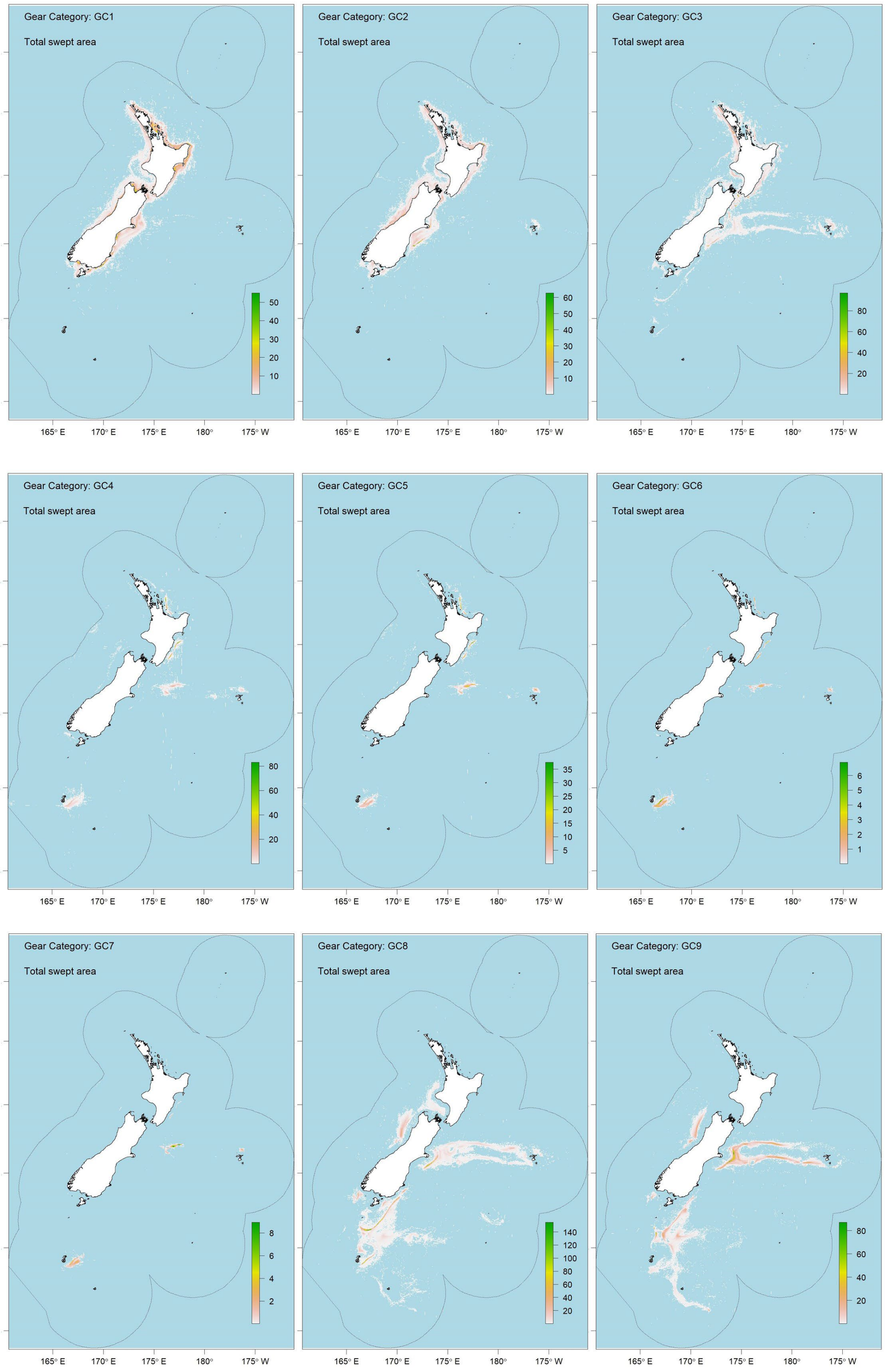

Gear Category: GC1
Footprint (fraction of cell trawled)
1.0
0.8
0.6
0.4
0.2
Gear Category: GC2
Footprint (fraction of cell trawled)
Gear Category: GC3
Footprint (fraction of cell trawled)
Gear Category: GC4
Footprint (fraction of cell trawled)
Gear Category: GC5
Footprint (fraction of cell trawled)
Gear Category: GC6
Footprint (fraction of cell trawled)
Gear Category: GC7
Footprint (fraction of cell trawled)
Gear Category: GC8
Footprint (fraction of cell trawled)
Gear Category: GC9
Footprint (fraction of cell trawled)
165° E
170° E
175° E
180°
175° W

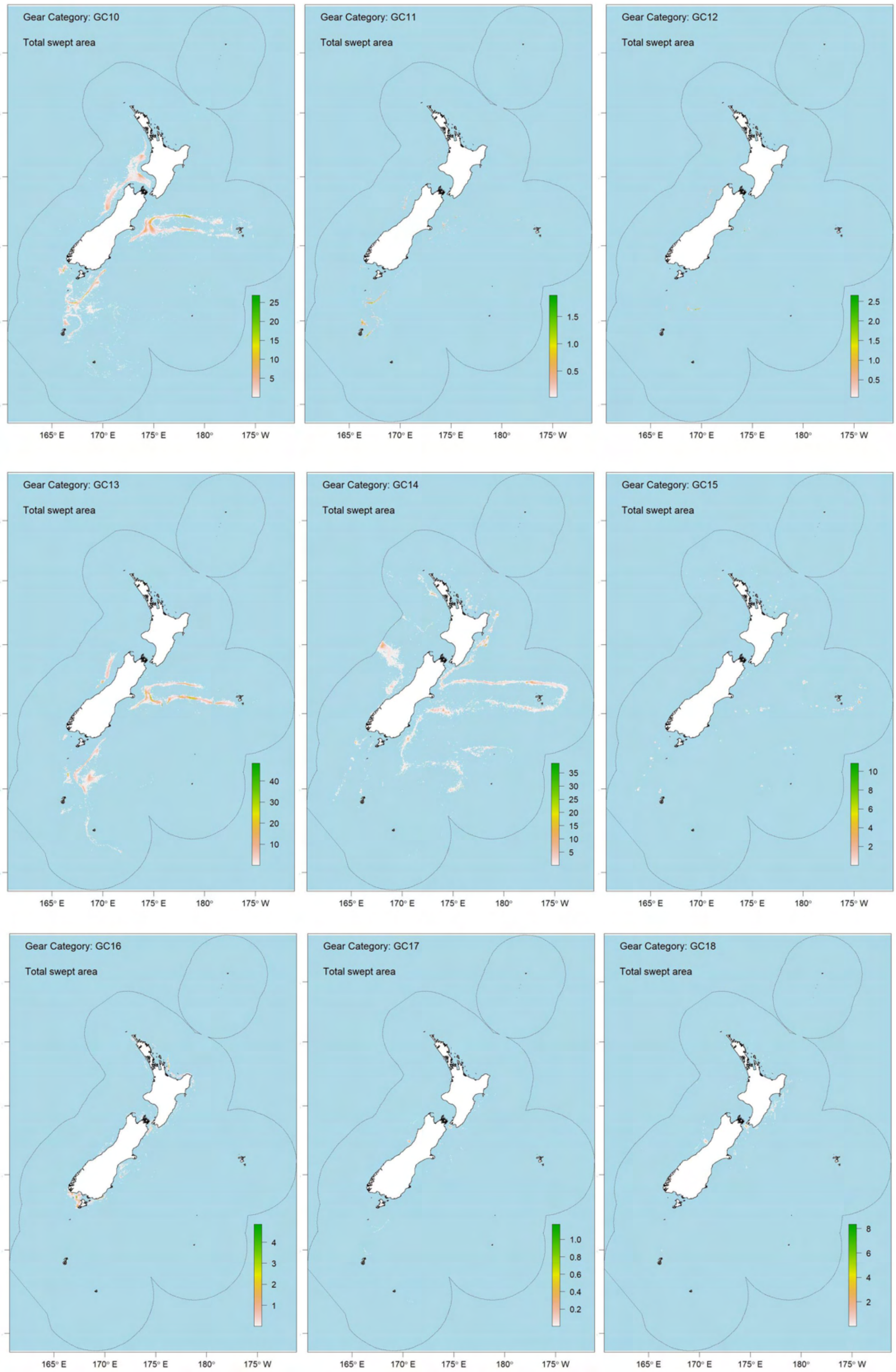

Gear Category: GC10
Footprint (fraction of cell trawled)
1.0
0.8
0.6
0.4
0.2
165° E
170° E
175° E
180°
175° W
Gear Category: GC11
Footprint (fraction of cell trawled)
0.8
0.6
0.4
0.2
165° E
170° E
175° E
180°
175° W
Gear Category: GC12
Footprint (fraction of cell trawled)
0.8
0.6
0.4
0.2
165° E
170° E
175° E
180°
175° W
Gear Category: GC13
Footprint (fraction of cell trawled)
1.0
0.8
0.6
0.4
0.2
165° E
170° E
175° E
180°
175° W
Gear Category: GC14
Footprint (fraction of cell trawled)
1.0
0.8
0.6
0.4
0.2
165° E
170° E
175° E
180°
175° W
Gear Category: GC15
Footprint (fraction of cell trawled)
0.8
0.6
0.4
0.2
165° E
170° E
175° E
180°
175° W
Gear Category: GC16
Footprint (fraction of cell trawled)
0.8
0.6
0.4
0.2
165° E
170° E
175° E
180°
175° W
Gear Category: GC17
Footprint (fraction of cell trawled)
0.6
0.5
0.4
0.3
0.2
0.1
165° E
170° E
175° E
180°
175° W
Gear Category: GC18
Footprint (fraction of cell trawled)
0.8
0.6
0.4
0.2
165° E
170° E
175° E
180°
175° W

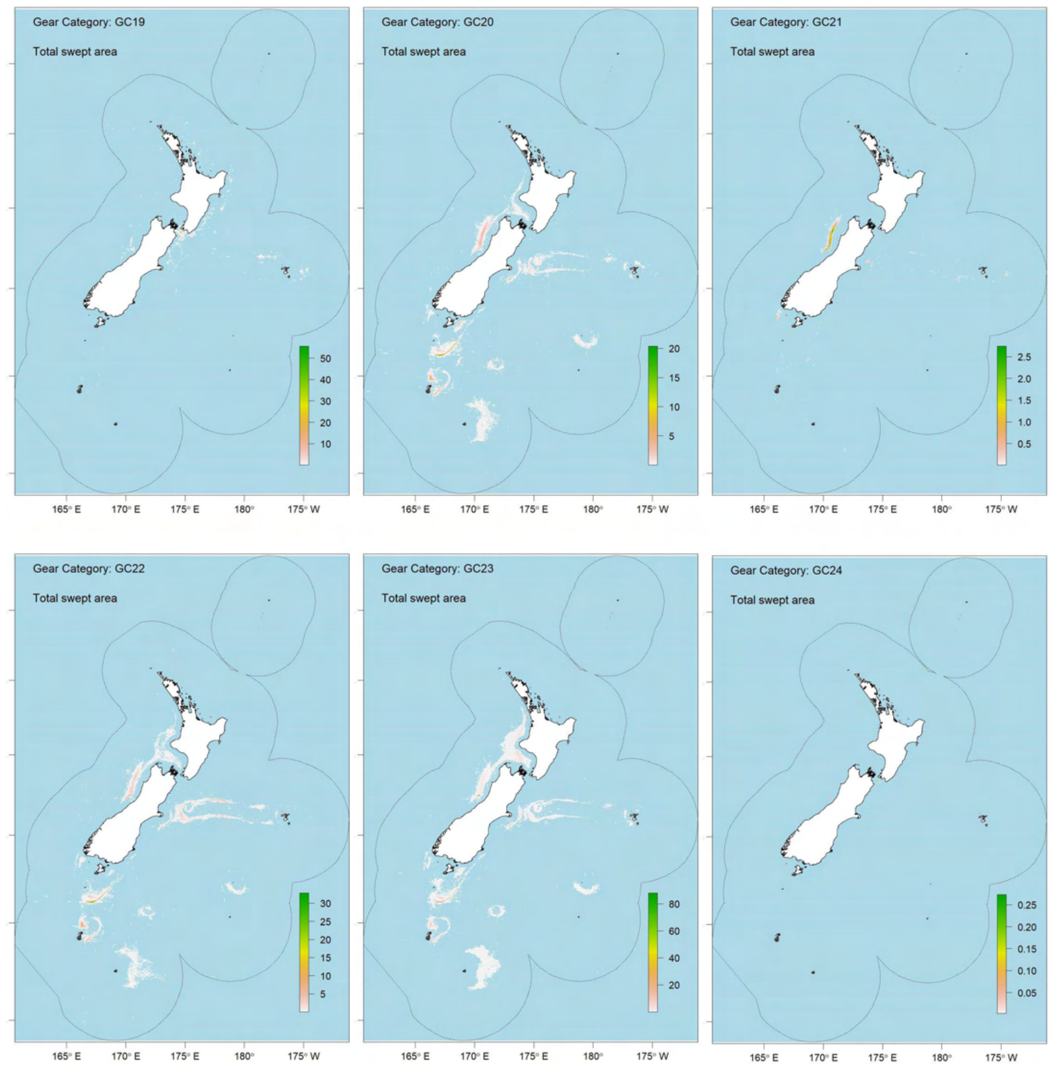

Gear Category: GC19
Footprint (fraction of cell trawled)
1.0
0.8
0.6
0.4
0.2
Gear Category: GC20
Footprint (fraction of cell trawled)
1.0
0.8
0.6
0.4
0.2
Gear Category: GC21
Footprint (fraction of cell trawled)
0.8
0.6
0.4
0.2
165° E
170° E
175° E
180°
175° W
Gear Category: GC22
Footprint (fraction of cell trawled)
1.0
0.8
0.6
0.4
0.2
Gear Category: GC23
Footprint (fraction of cell trawled)
1.0
0.8
0.6
0.4
0.2
Gear Category: GC24
Footprint (fraction of cell trawled)
0.20
0.15
0.10
0.05
165° E
170° E
175° E
180°
175° W

APPENDIX 4: TOTAL TRAWL SWEPT AREA BY GEAR CATEGORY

Spatial extent of bottom contact (total swept area) by different trawl gear categories for the period 2007–08 to 2018–19 (inshore fisheries) and 1989–90 to 2018–19 (deepwater fisheries), within the New Zealand TS/EEZ. Map resolution (cell size) is 1 × 1 km. See Table 5 for gear category descriptions.



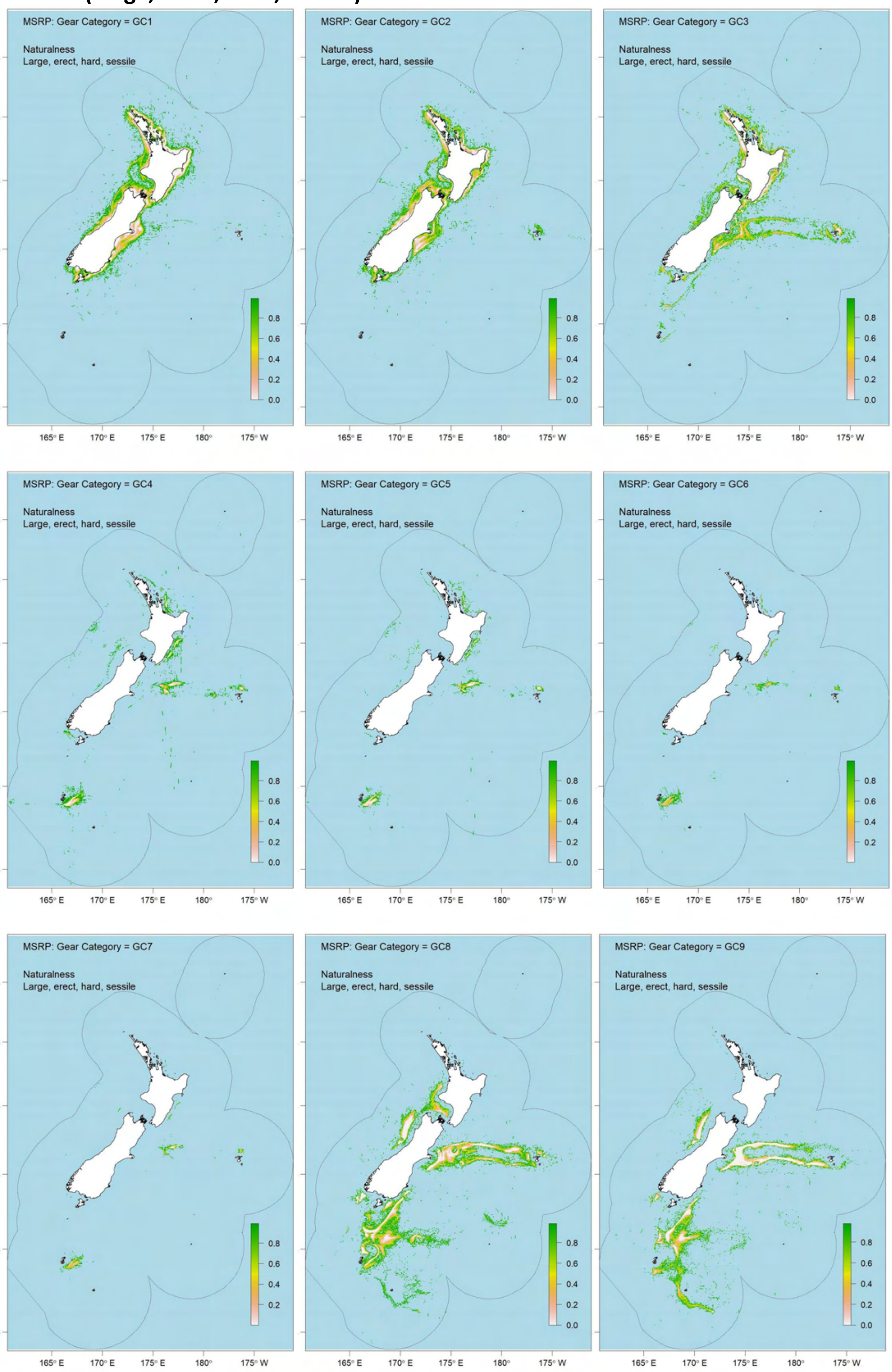

Gear Category: GC10
Total swept area
25
20
15
10
5
Gear Category: GC11
Total swept area
1.5
1.0
0.5
Gear Category: GC12
Total swept area
2.5
2.0
1.5
1.0
0.5
Gear Category: GC13
Total swept area
40
30
20
10
Gear Category: GC14
Total swept area
35
30
25
20
15
10
5
Gear Category: GC15
Total swept area
10
8
6
4
2
Gear Category: GC16
Total swept area
4
3
2
1
Gear Category: GC17
Total swept area
1.0
0.8
0.6
0.4
0.2
Gear Category: GC18
Total swept area
8
6
4
2
165° E
170° E
175° E
180°
175° W


Gear Category: GC19
Total swept area
50
40
30
20
10
Gear Category: GC20
Total swept area
20
15
10
5
Gear Category: GC21
Total swept area
2.5
2.0
1.5
1.0
0.5
Gear Category: GC22
Total swept area
30
25
20
15
10
5
Gear Category: GC23
Total swept area
80
60
40
20
Gear Category: GC24
Total swept area
0.25
0.20
0.15
0.10
0.05
165° E
170° E
175° E
180°
175° W

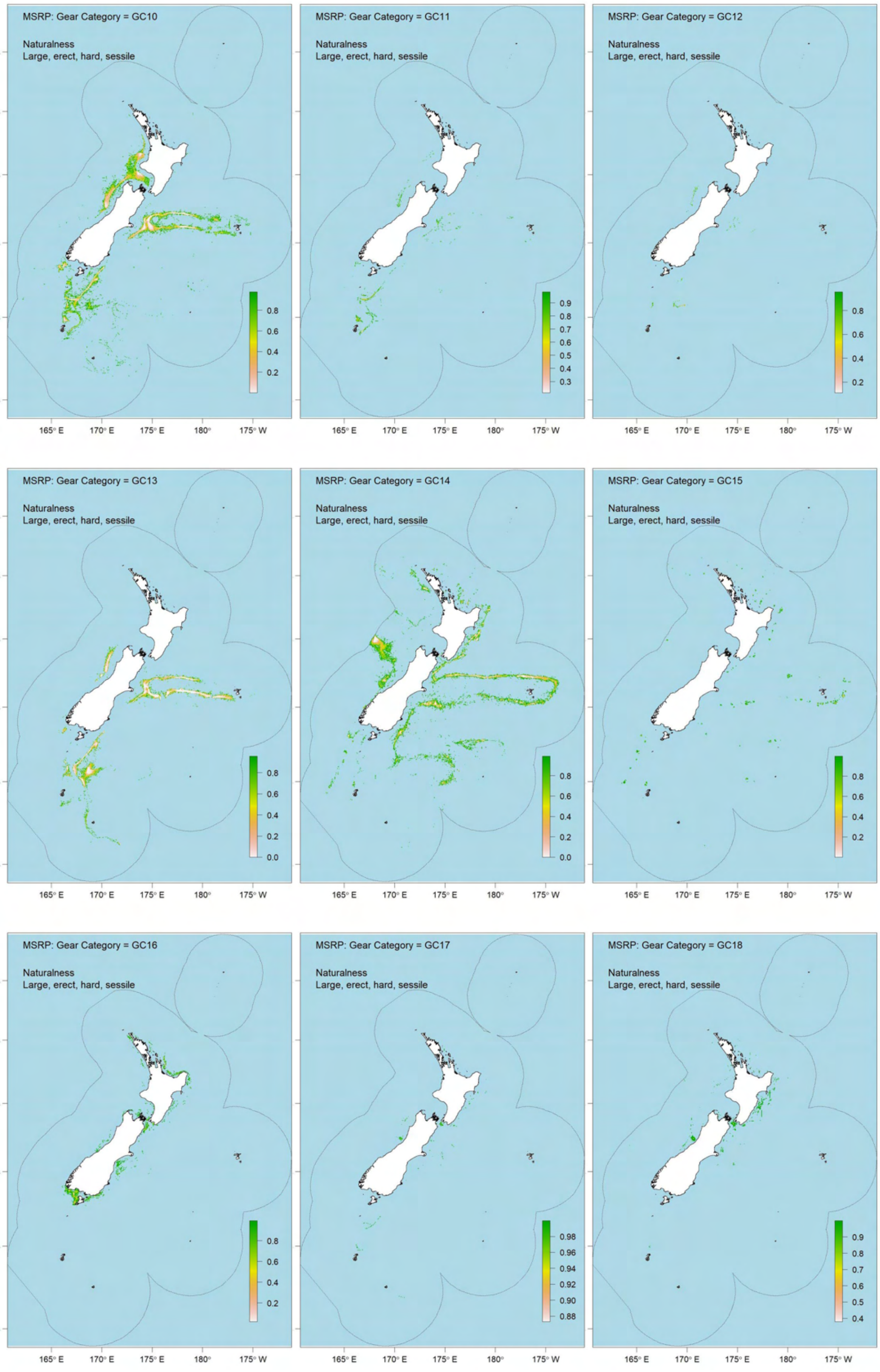
APPENDIX 5: MSRP BENTHIC IMPACT BY FAUNAL AND GEAR CATEGORIES

MSRP benthic impact estimates, as relevant to separate benthic faunal and gear categories. Note that these maps do not include any information on the actual distribution of the benthic faunal categories. The range in values is 0–1, where 0=completely impacted and 1=unimpacted, for three categories on benthic fauna within the New Zealand TS/EEZ, based on the MSRP method (Mormede et al. 2017). See Table 8 for the mortality values behind the differences in these plots. See Table 5 for gear category descriptions.

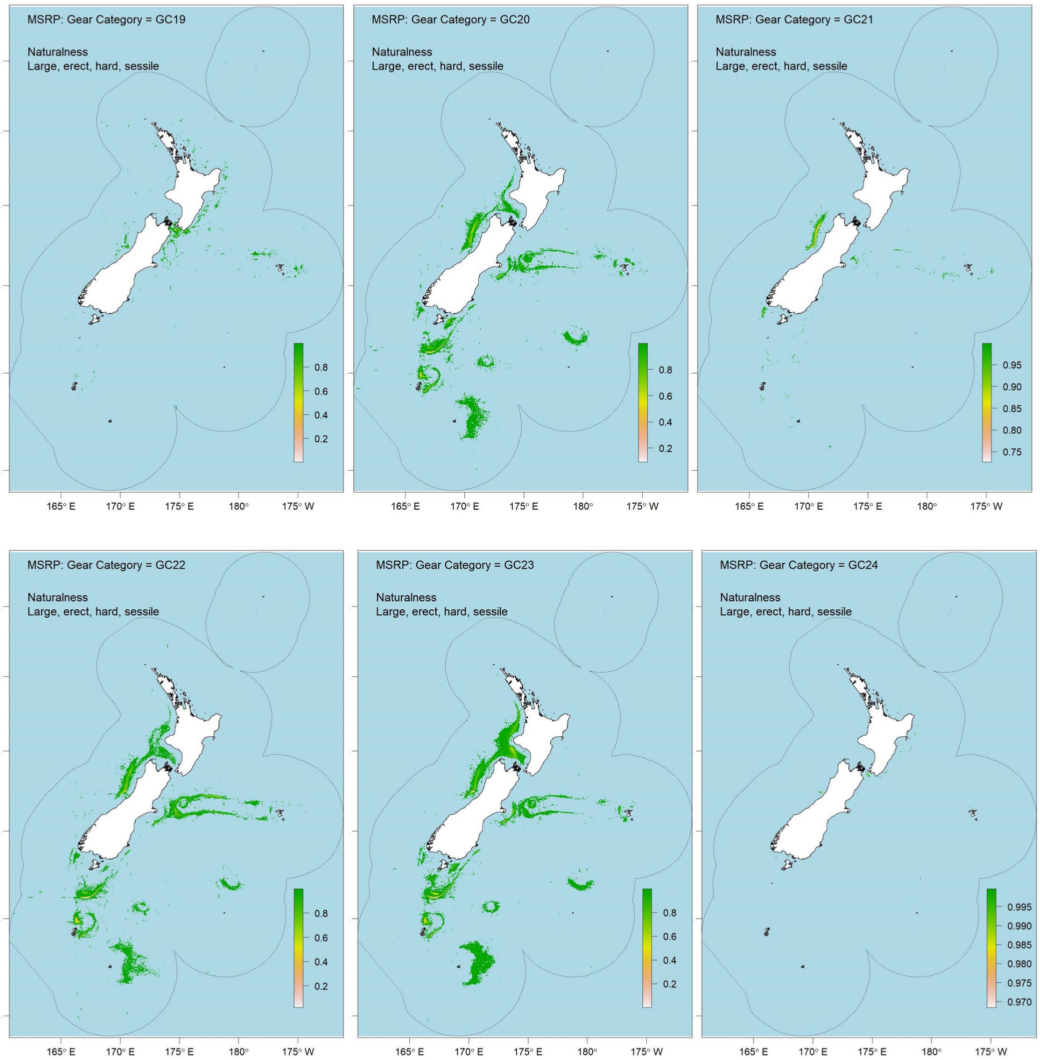
A: LEHS (Large, erect, hard, sessile)



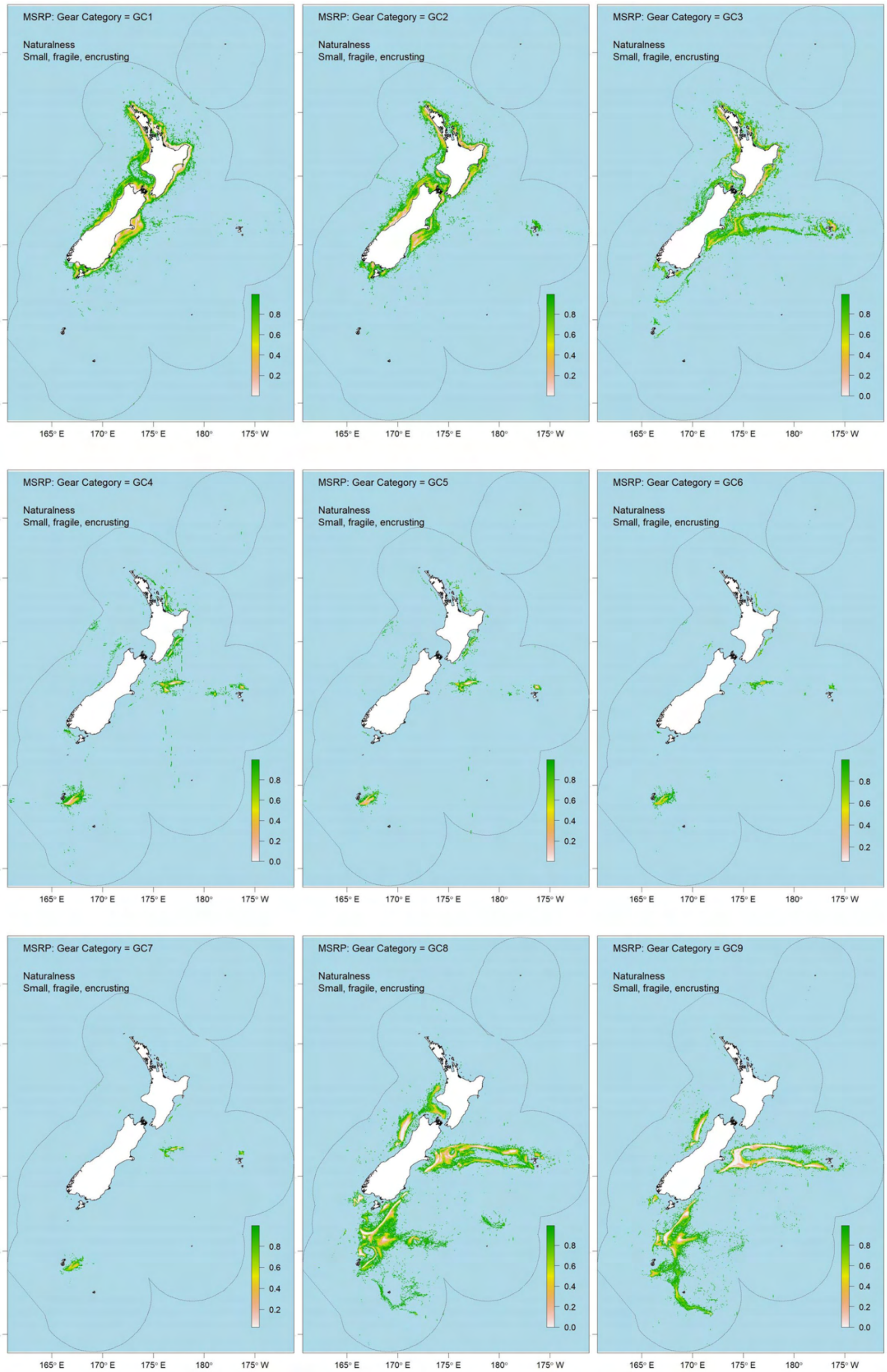
A: LEHS (Large, erect, hard, sessile) — *continued*



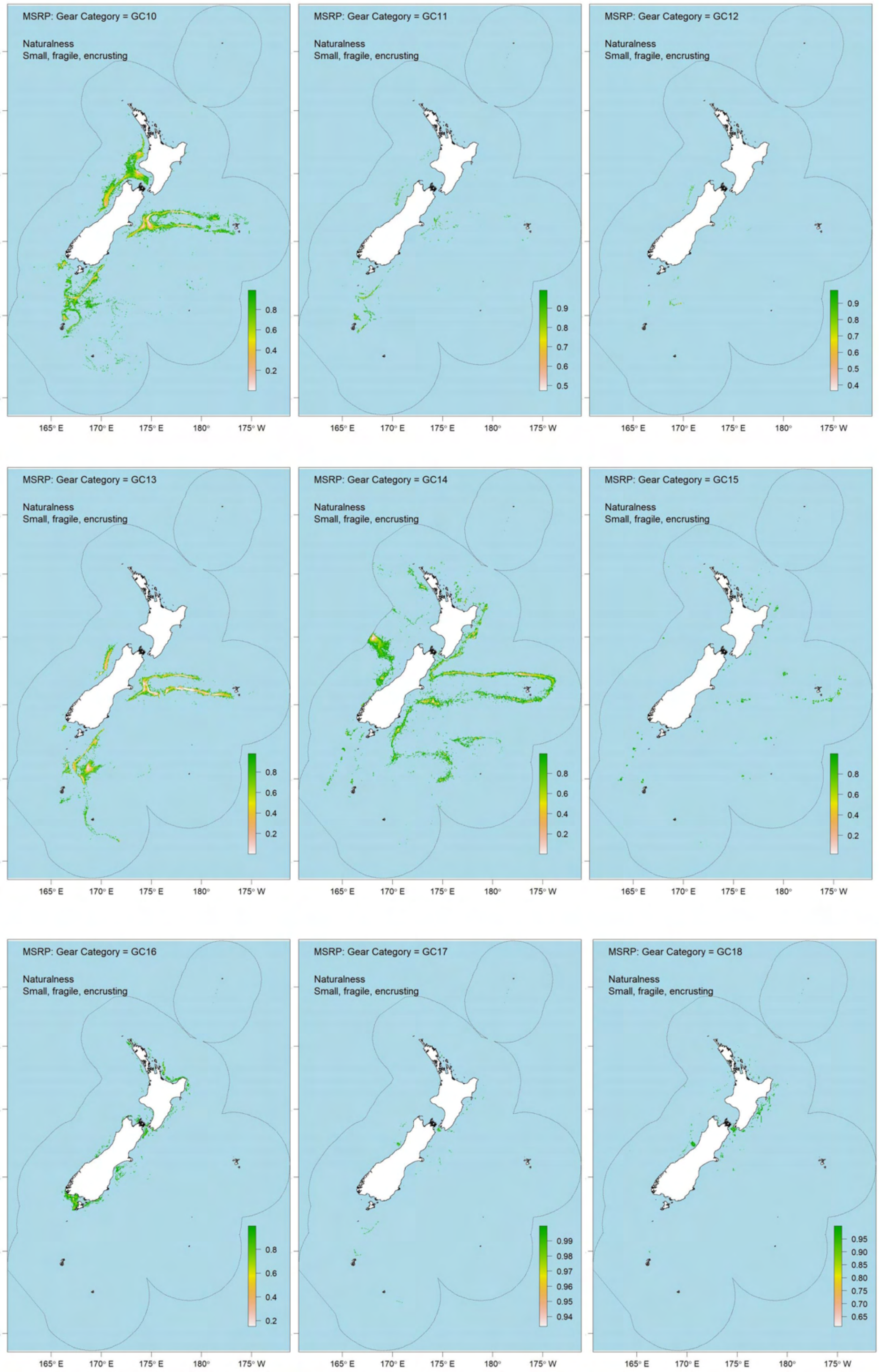
A: LEHS (Large, erect, hard, sessile) — *continued*



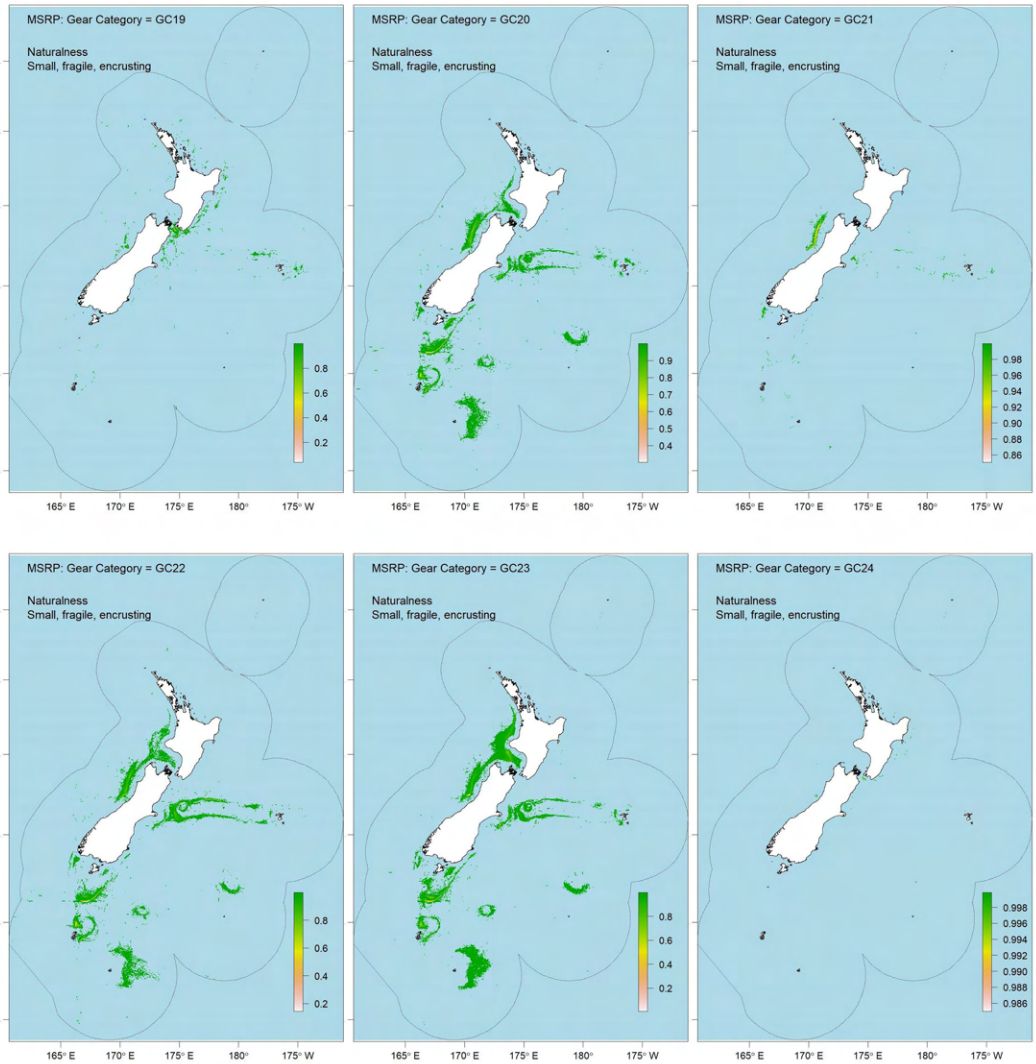
B: SFE (Small, fragile, encrusting)



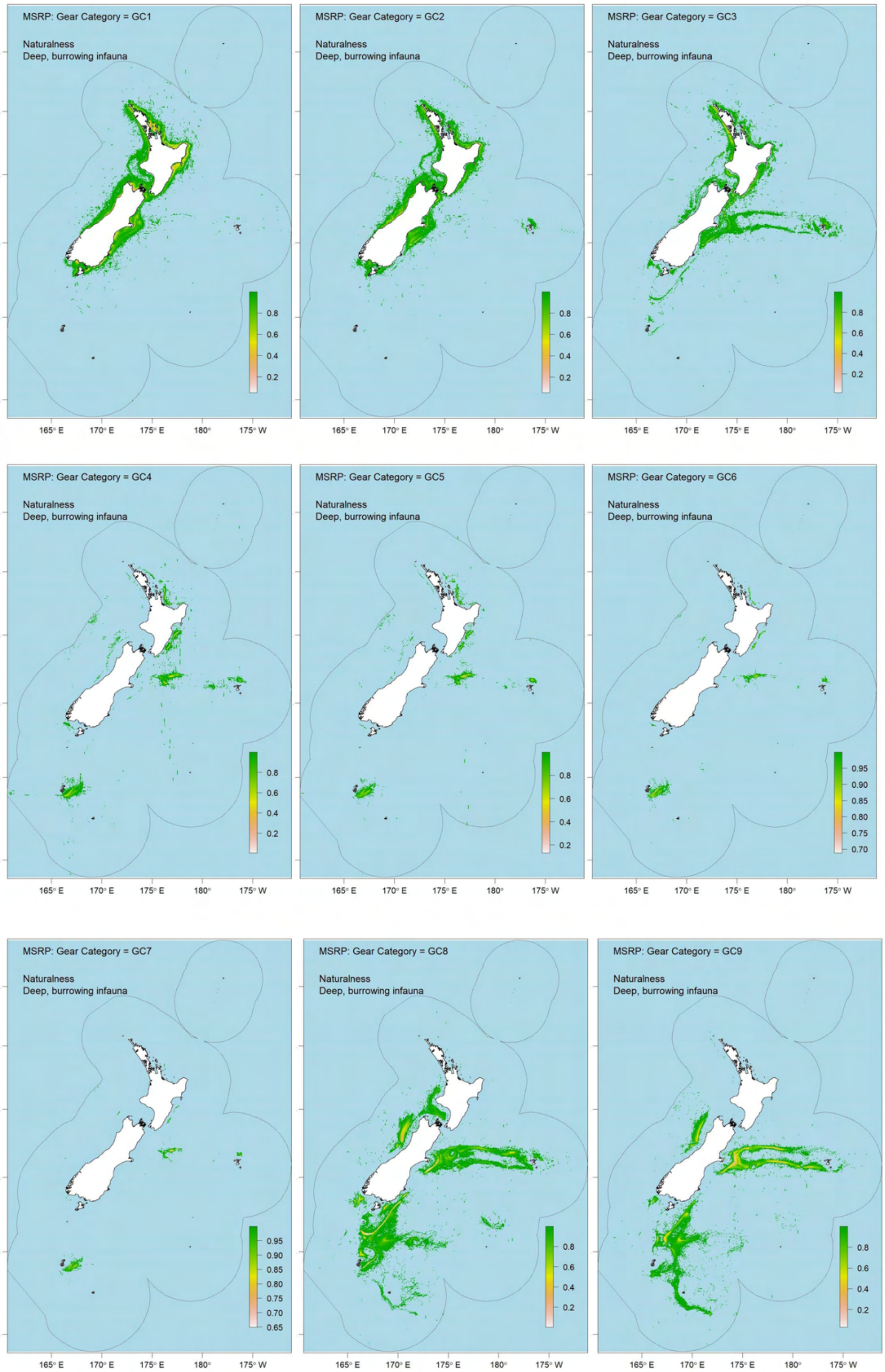
B: SFE (Small, fragile, encrusting)—*continued*



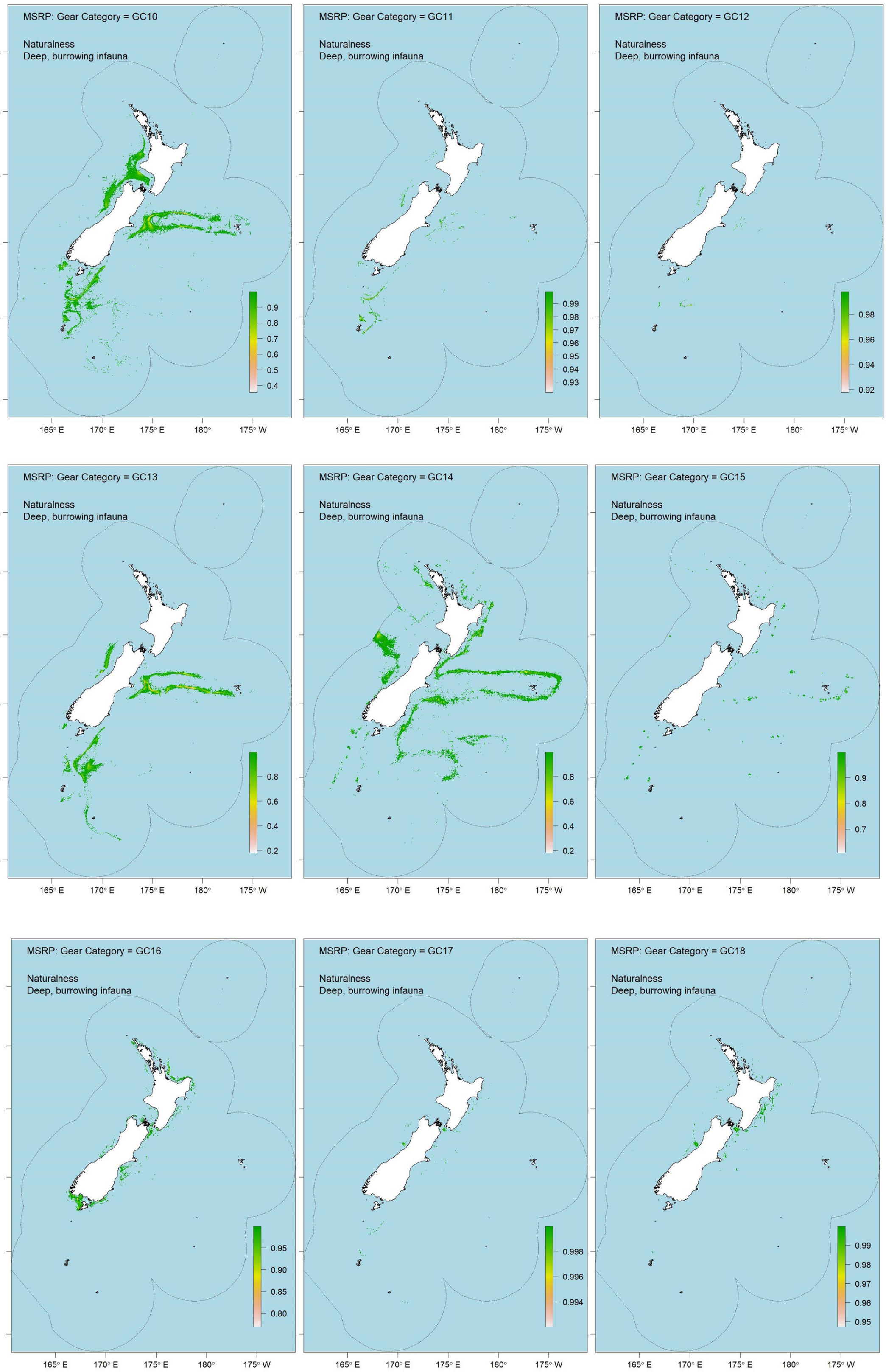
B: SFE (Small, fragile, encrusting)—*continued*



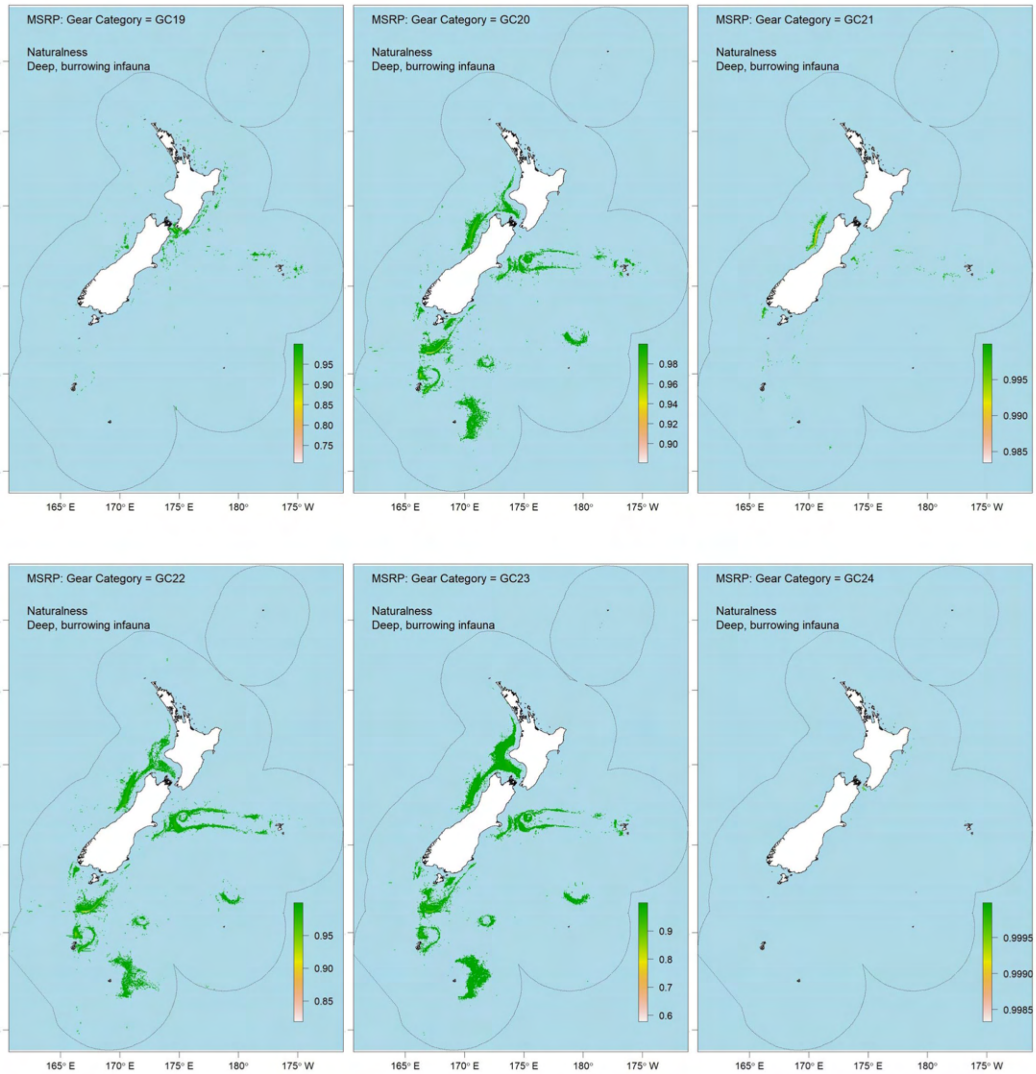
C: DBI (Deep, burrowing infauna)



C: DBI (Deep, burrowing infauna)—*continued*

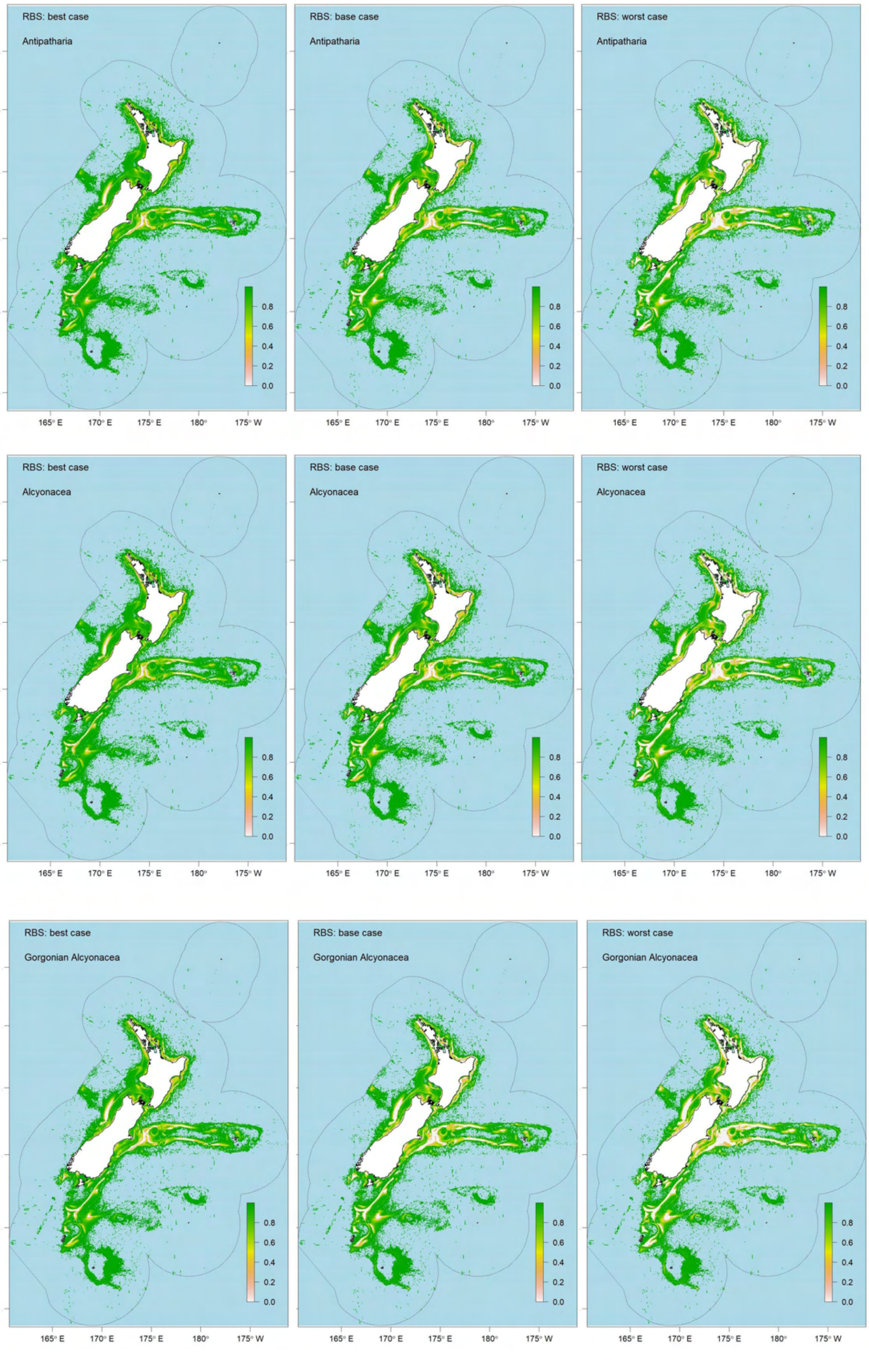


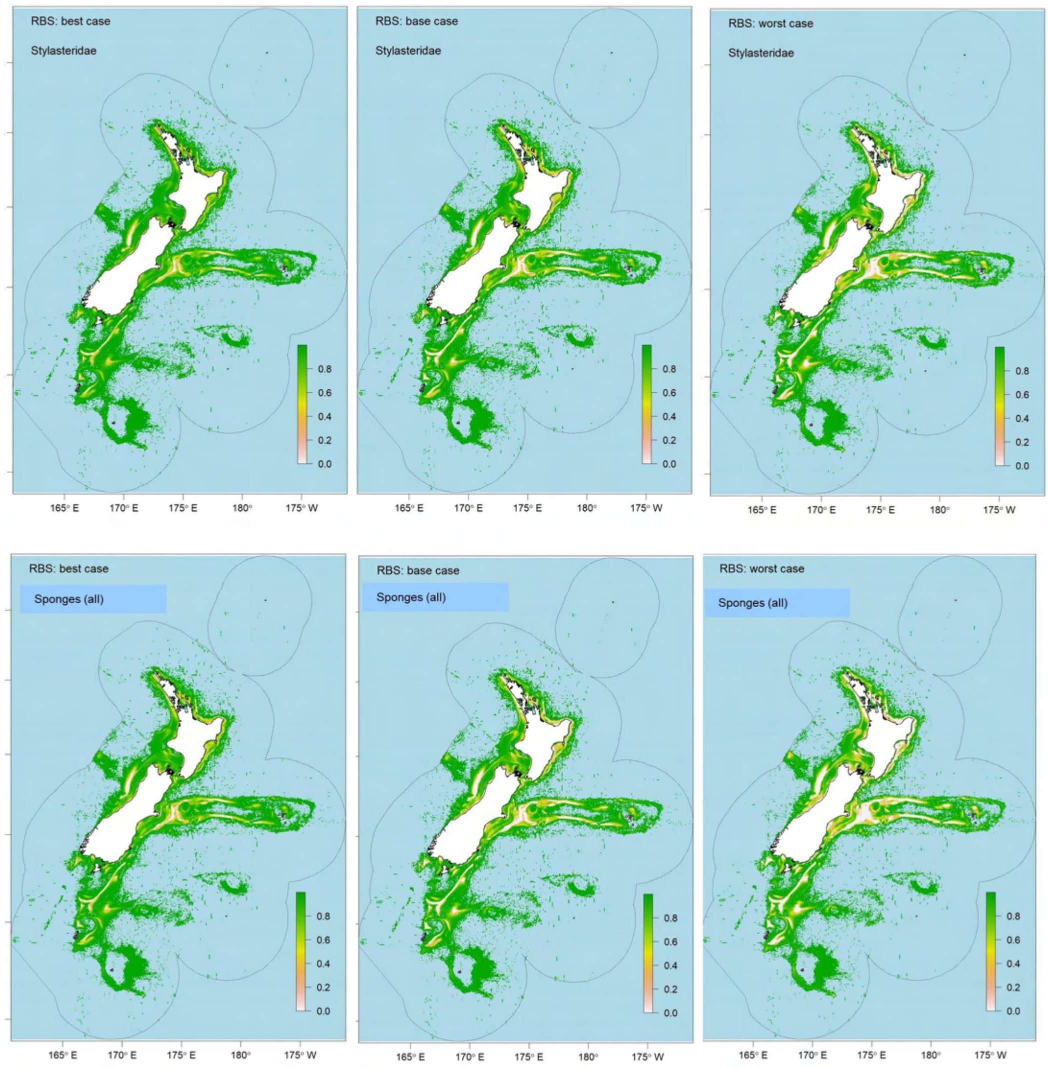
C: DBI (Deep, burrowing infauna)—*continued*



APPENDIX 6: RBS BENTHIC STATUS BY TAXON (GEAR CATEGORIES COMBINED)

Relative Benthic Status. Sensitivities (best, base, and worst cases as determined from values in Table 2) for vulnerable marine ecosystem indicator taxa not shown in Figure 28, within the New Zealand TS/EEZ, based on the RBS method (Pitcher et al. 2017). Note that these maps do not include any information on the actual distribution of the VME indicator taxa.

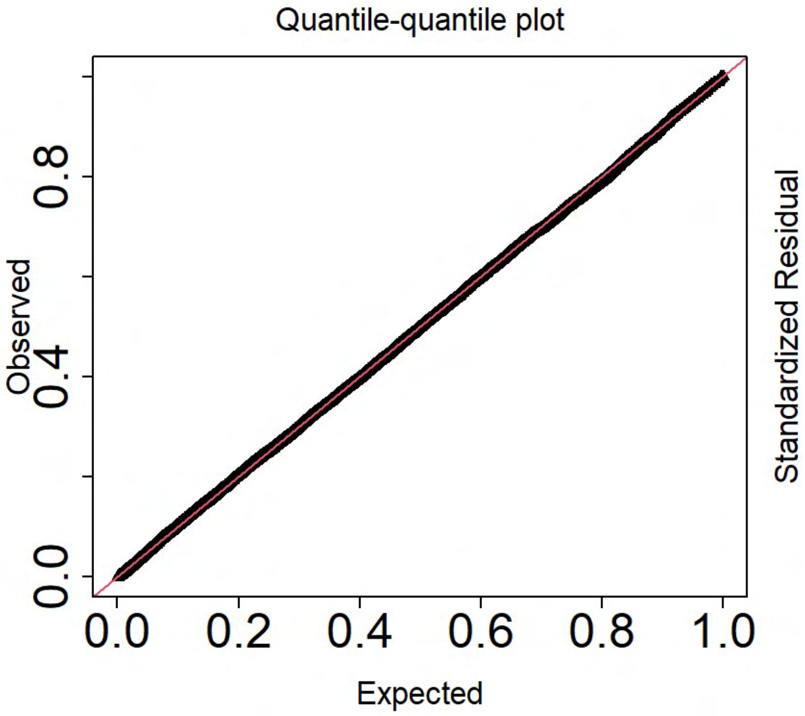



RBS: best case
Stylasteridae
RBS: base case
Stylasteridae
RBS: worst case
Stylasteridae
RBS: best case
Sponges (all)
RBS: base case
Sponges (all)
RBS: worst case
Sponges (all)
0.8
0.6
0.4
0.2
0.0
165° E
170° E
175° E
180°
175° W

APPENDIX 7: HOLOTHURIAN VAST MODEL FITS

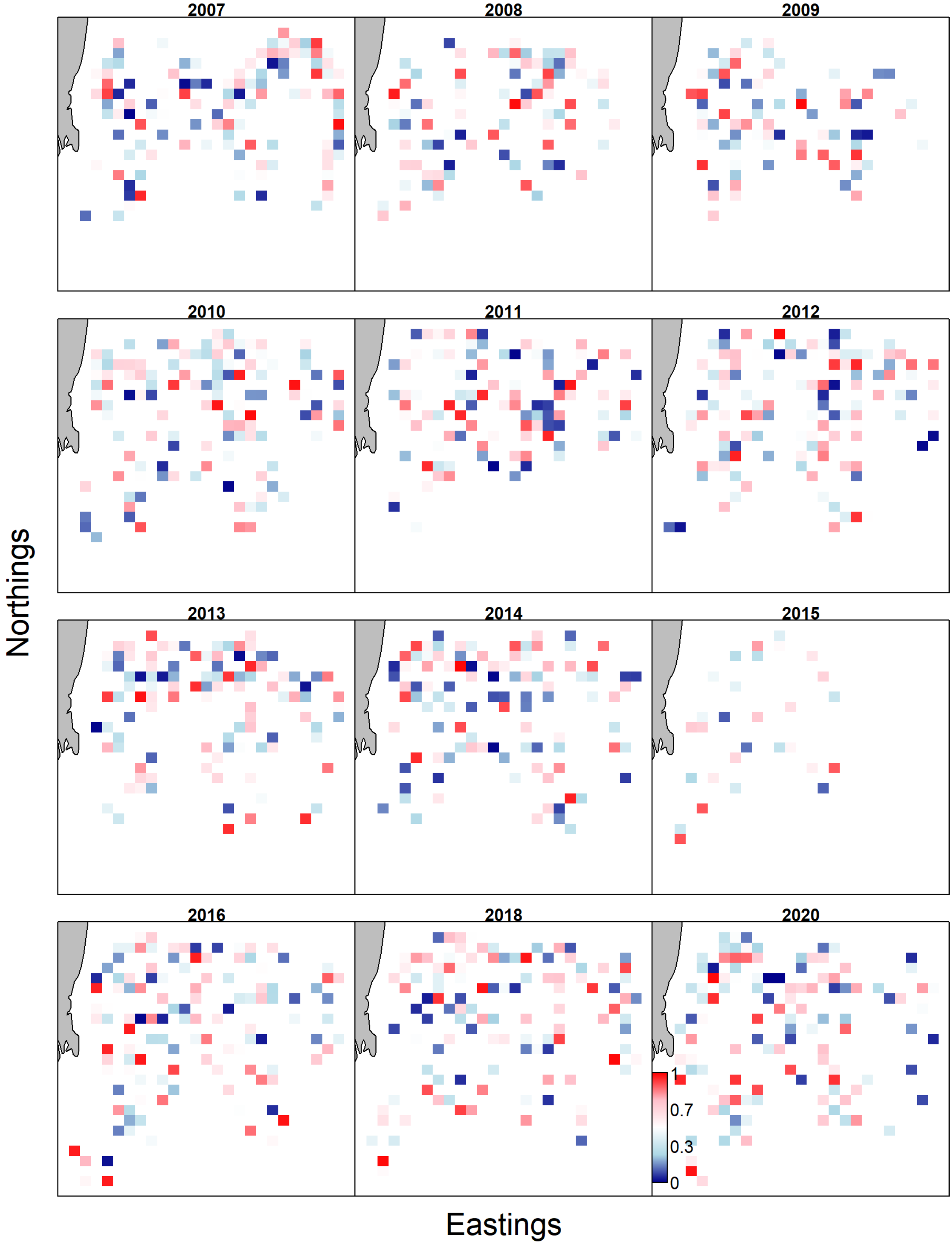
A: Dharma residuals

B: Q-Q plot of dharma residuals



APPENDIX 8: ANTHOZOAN VAST MODEL FITS

A: Dharma residuals



B: Q-Q plot of the dharma residuals