



Assessing the effects of fishing on soft sediment habitat, fauna and process

New Zealand Aquatic Environment and Biodiversity Report No. 178

I.D. Tuck,
J.E. Hewitt,
S.J. Handley
C.J. Lundquist

ISSN 1179-6480 (online)
ISBN 978-1-77665-465-9 (online)

January 2017



Requests for further copies should be directed to:

Publications Logistics Officer
Ministry for Primary Industries
PO Box 2526
WELLINGTON 6140

Email: brand@mpi.govt.nz
Telephone: 0800 00 83 33
Facsimile: 04-894 0300

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

© Crown Copyright - Ministry for Primary Industries

TABLE OF CONTENTS

EXECUTIVE SUMMARY	1
1. INTRODUCTION	2
2. SAMPLING STRATEGIES	4
2.1 Experimental approach	4
2.2 Sampling methodologies	5
2.3 Statistical approaches	9
2.4 Design decisions	11
2.5 Study locations	16
3. TASMAN AND GOLDEN BAYS	19
3.1 Dedicated benthic community sampling	19
3.1.1 Methods	19
3.1.2 Results	32
3.1.3 Conclusions	45
3.2 Opportunistic analysis of survey bycatch data	46
3.2.1 Conclusions	51
3.3 Tasman and Golden Bays overall conclusions	52
4. SOUTH CANTERBURY BIGHT	52
4.1 Dedicated benthic community sampling	52
4.1.1 Methods	52
4.1.1 Results	60
4.1.1 Conclusions	72
4.2 Opportunistic analysis of survey bycatch data	73
4.2.1 Conclusions	75
4.3 South Canterbury Bight overall conclusions	76
5. CHATHAM RISE AND CHALLENGER PLATEAU	76
5.1 Opportunistic benthic community sampling	76
5.1.1 DTIS sampling	77
5.1.2 Seamount sledge and beam trawl	82
5.2 Opportunistic analysis of survey bycatch data	84
5.3 Chatham Rise and Challenger Plateau overall conclusions	89
6. SCAMPI FISHERIES	90
6.1 Bay of Plenty (SCI 1) and Hawke Bay / Wairarapa (SCI 2)	90
6.1.1 Trawl survey bycatch	90
6.1.2 Photographic survey data	93
6.2 Auckland Islands (SCI 6A)	95
6.2.1 Trawl survey bycatch	95

6.2.2	Photographic survey data	95
6.3	Scampi fishery overall conclusions	99
7.	OTHER PREVIOUS NEW ZEALAND STUDIES	99
7.1	Spirits Bay	99
7.2	Hauraki Gulf region	100
8.	SUMMARY OF NEW ZEALAND STUDIES, AND THE IMPLICATIONS OF THE EFFECTS OF FISHING	101
9.	ACKNOWLEDGMENTS	109
10.	REFERENCES	110
11.	APPENDIX 1: Marginal tests for DistLM models for Tasman and Golden bays case study	123
12.	APPENDIX 2: Environmental data used to stratify South Canterbury Bight study area	129
13.	APPENDIX 3: Marginal tests for DistLM models for the South Canterbury Bight case study	138

EXECUTIVE SUMMARY

Tuck, I.D; Hewitt, J.E.; Handley, S.J.; Lundquist, C.J. (2017). Assessing the effects of fishing on soft sediment habitat, fauna and process.

New Zealand Aquatic Environment and Biodiversity Report No. 178. 143 p.

This report provides an overview of the effects of fishing on soft bottom communities and habitats, and describes the sampling designs and results of new studies undertaken, which are combined with existing New Zealand research.

There is a long history of global concern over the effects of fishing on benthic communities, and a large number of studies have been conducted, largely within Europe and North America. Experimental approaches for the early studies were largely based on before/after, control/impact designs, but with improvements in availability of fine scale fishing effort data, and research focussing more on “fishery scale” effects, rather than “experimental plot scale” effects, the application of gradient based approaches has become more widespread. The studies conducted within this research have applied this gradient based approach at the scale of fisheries, and sampled benthic communities using a range of intensive (e.g., dedicated grab sampling of infauna) to opportunistic (e.g., trawl survey benthic invertebrate bycatch) methods, across a range of habitats. Specific case studies were focussed on Tasman and Golden Bays, and the South Canterbury Bight. Previously collected Chatham Rise and Challenger Plateau Oceans Survey 20/20 data were also analysed, along with a range of other opportunistic data sets. In conjunction with previous New Zealand studies, these provide case studies from the northern tip of the North Island (Spirits Bay) to the sub-Antarctic (Auckland Islands), and from coastal waters down to depths of 1000 m.

The magnitude of the effects of fishing (% variability explained) varied between studies, and as would be expected, greater effects were detected over stronger effort gradients. The levels of effect detected were reasonably consistent between dedicated sampling approaches (within study), while opportunistic data sets were less effective at detecting effects. When effects were detected, fishing was associated with reductions in the number of taxa, diversity and evenness of both epifaunal and infaunal communities, but more consistently for epifauna. Fishing appears to have reduced epifaunal biomass and productivity (whole community and fish prey) by up to 50% in some of the study sites, but effects on infauna were less consistent (increasing by up to 20% in the one area an effect was detected). The species that were most consistently identified as being negatively correlated with fishing pressure were those that either stand erect out of the seabed (e.g., horse mussels, sponges, bryozoans, hydroids, sea pens, tube building polychaetes), or live on the sediment surface, and thus are particularly sensitive to physical disturbance through either direct physical impact (e.g., *Echinocardium*), smothering (e.g., small bivalves) or increased vulnerability to predation following disturbance (e.g., brittle stars). Where examined, even relatively modest levels of fishing effort (i.e., fishing an area between once and twice per year, estimated at the 5 km × 5 km scale) reduced the density of the combined group of long lived sedentary habitat forming species and individual species group densities of holothurians, crinoids, cnidarians and bryozoans by at least 50%. This level of impact is likely to occur for less than 10 percent of the area shallower than 100 m and in less than 5 percent of the rest of the fishable area down to a depth of 1600 m.

1. INTRODUCTION

The first documented concerns about the use of towed fishing gear on benthic habitats were from UK fishermen in the fourteenth century (Lokkeborg 2005; Tracey et al. 2012), related to the capture of juvenile fish and the detrimental effects on food sources for harvestable fish. Despite this long history of concern, it is really only since the 1990s that research efforts have focused strongly on the effects of fishing on benthic communities, biodiversity, and production. The rapid expansion of studies in this area, and the controversy associated with the effects of fishing has led to numerous reviews, summarizing the research and identifying overall patterns (Gislason 1994; Dayton et al. 1995; Jennings & Kaiser 1998; Lindeboom & de Groot 1998; Hall 1999; Collie et al. 2000; Gislason et al. 2000; Kaiser & de Groot 2000; Dayton et al. 2002; Thrush & Dayton 2002; Lokkeborg 2005; DFO 2006; Kaiser et al. 2006; Rice 2006). These assorted reviews are in general agreement, concluding that benthic disturbance from mobile fishing varies in relation to the habitat, fishing gear, and environment, and is likely to have predictable and potentially substantial effects on benthic community structure and function. These effects can lead to regional-scale reductions in some components of biodiversity, reduce benthic community productivity (Jennings et al. 2001; Hiddink et al. 2006), alter natural sediment fluxes and reduce organic carbon turnover (Pusceddu et al. 2014), and modify the shape of the upper continental slope (Puig et al. 2012), reducing morphological complexity and benthic habitat heterogeneity.

The various reviews and meta-analyses of published studies (Collie et al. 2000; Kaiser et al. 2006) are in general agreement that dredges tend to be more damaging than trawls, with other gears having less, but still detectable impact. Physical impacts of bottom gear are greatest on hard, complex seabed, and least on unconsolidated sandy habitats. The greatest effects have been recorded in low energy environments, and high energy environments are considered more robust, with effects sometimes thought negligible. However, this particular conclusion may partly reflect the lack of a historical perspective on the benthic communities, and the long fishing history of the study locations (mostly Northeastern Europe and North America), as the reviews also concluded that long-lived species, surface-living species, structurally fragile species, and biogenic habitat forming species were all particularly vulnerable, and such species are frequently found in high energy environments.

Soft-sediment habitats dominate the seafloor throughout the world's oceans and support most bottom trawl and dredge fisheries. These fisheries are likely to have predictable and in some cases, substantial effects on benthic community structure. These changes in benthic communities have further implications for ecosystem function (e.g., removal of main bioturbators having long term consequences for benthic-pelagic processes; Lohrer et al. 2004; Huettel et al. 2014). Benthic habitat structure provides a number of important services for fish species, including nursery grounds and refuge from predation (Beck et al. 2001; Dahlgren et al. 2006). A range of sensitive marine benthic habitats have been identified for New Zealand (MacDiarmid et al. 2013), and any known links between these (and other) habitats and fish species have been documented (Morrison et al. 2014a; Morrison et al. 2014b). Knowledge of the effects of fishing on benthic communities has been used to evaluate the likely performance of different management approaches (Ellis & Pantus 2001), and both simple (Mormede & Dunn 2013) and more sophisticated spatially explicit models (Lundquist et al. 2013) have been developed within New Zealand to assist benthic community Ecological Risk Assessments.

Within New Zealand, trawls and dredges are used to catch a high proportion of commercial landings, and such methods represent the only effective and economically feasible way of catching some species. Bottom trawling is an important or predominant method in many of our key fisheries (e.g., hoki, orange roughy, snapper, squid). Because of its use in so many major inshore, middle depths and deepwater fisheries, bottom trawling is pervasive throughout much of the EEZ shallower than about 1200 m, and is probably the predominant disturbance agent deeper than about 100 m (Cryer et al. 2002). New Zealand oyster and scallop populations are exploited with dredge fisheries in a number of areas, on habitats ranging from mud to gravel, in relatively shallow depths (to a maximum of about 80 m, where these species occur). Although wind generated waves are also a source of widespread disturbance in

shallow waters around New Zealand, seabed disturbance by waves in areas deeper than 50 m is considered to be rare (Green et al. 1995). Fishing is considered the greatest threat to slope habitats (defined as 200 – 2000 m), vents, seeps and seamounts (less than 2000 m depth), while ocean acidification is considered the greatest threat to seamounts and other habitats deeper than 2000 m (MacDiarmid et al. 2012). Despite this there have been just two previous New Zealand empirical studies (one coastal, the other on the upper slope) at the broad spatial scale at which fisheries operate (Thrush et al. 1998; Cryer et al. 2002). Although we can infer *a priori* that disturbance by fishing constitutes a risk to ecological and production processes, our knowledge of the detailed consequences of change and the likely rates of recovery of specific New Zealand systems remains limited.

This project builds on a series of other studies targeting information gaps in support of comprehensive management of environmental effects of fishing. Previous work has generated maps of the distribution and frequency of bottom trawling effort (Baird et al. 2011; Black et al. 2013; Baird et al. 2015), developed broad-scale classification of the marine environment based on physical and biological data (Snelder et al. 2006; Leathwick et al. 2012), and collected new data on benthic communities in specific areas of interest (Bowden 2011). Together, these three major resources provide the kernel of a broad-scale study to assess the extent to which bottom trawling and dredging have modified our extensive, soft-sediment benthic habitats. The *Strategy for Managing the Environmental Effects of Fishing* (Ministry of Fisheries 2005) identifies that species or habitats at risk from fishing should be identified on the basis of reviews of current information (mostly from work on the overlap between habitat distributions and fishing effort) combined with targeted research (this project), environmental impact assessment (plans currently under development by MPI), and input from tangata whenua and stakeholders.

Overall objective:

To assess changes in seafloor communities and quantify key processes affected by disturbance from bottom fishing.

Specific objectives:

1. To design and test sampling and analytical strategies for broad-scale assessments of habitat and faunal spatial structure and variation across a variety of seafloor habitats.
2. To design and carry out experiments to assess the effects of bottom trawling and dredging on benthic communities and ecological processes important to the sustainability of fishing at scales of relevance to fishery managers.

This report addresses the design, analytical and study site location aspects in the following chapter. Separate chapters then describe the two case studies conducted using data collected specifically for this project; one in Tasman/Golden Bay, the other in the South Canterbury Bight; one study on Chatham Rise and Challenger Plateau, using Ocean Survey 20/20 data and research trawl bycatch data; an analysis of scampi fishery bycatch data in several regions, and consideration of data sets from Spirits Bay and the Hauraki Gulf. The results from the various studies are drawn together in a final chapter, summarising the implications of the effects of fishing on benthic communities at the EEZ scale.

2. SAMPLING STRATEGIES

Dedicated field sampling studies have been widely used to examine the effects of fishing on benthic communities (Watling et al. 2014), but such studies (particularly in offshore areas) require considerable ship time and expense, and it is highly unlikely that it will ever be practical to undertake such studies at the scale of the EEZ. Within this study we aim to investigate the effects of fishing on benthic communities with dedicated benthic sampling data of a deepwater case study area collected during the Oceans Survey 20/20 voyages of the Chatham Rise and Challenger Plateau (Bowden 2011), and two coastal case study areas undertaken within this project. We also took opportunistic advantage of previous studies that have collected data for other purposes (Greenstreet et al. 1999; Cryer et al. 2002), enabling analysis at minimal additional cost.

Within this study, we have aimed to undertake a suite of dedicated and opportunistic approaches, with a view to determining how consistent interpretations are from the different approaches. This will inform future decisions on the approaches employed in examining the effects of fishing in other New Zealand areas. A number of decisions therefore had to be made over the types of study to conduct in the coastal case studies, the sampling approaches to use, the opportunistic data to examine and the locations for the additional coastal case studies. Early in the project, a small workshop was held between a range of NIWA and MPI staff to discuss these issues, and the conclusions of this workshop are incorporated into the following paragraphs.

2.1 Experimental approach

A recent review identified 235 publications offering original research on the impacts of otter and beam trawls on the seabed, with a further 130 publications documenting effects of other mobile fishing gears (largely scallop, mussel and hydraulic dredges, but also intertidal hand raking) (Watling et al. 2014). These studies can largely be divided into the before/after, control/impact (BACI) design of Bernstein & Zalinski (1983) and Stewart-Oaten et al. (1986), or gradient based studies examining the effects of fishing at the scale of the fishery.

BACI approaches (ideally) involve manipulation (experimental trawling) and replicated sampling over time (Underwood 1992) (although many early studies only examined effects immediately after fishing, extending out to a few weeks). These approaches have been adopted widely in the literature about the effects of fishing, and avoid problems of spatial confounding (pseudoreplication; Hurlbert 1984) when multiple control and/or treatments are available (although see Eberhardt & Thomas 1991; Hewitt et al. 2001; Gray et al. 2006 for potential problems and alternative experimental designs). The detection of long term effects (on the timescale of months to years) relies on the ability to sample at, and prevent additional disturbance at, sites over a number of years, and longer term studies are therefore more expensive and logistically challenging than immediate effects or short term studies, and far less common in the literature. Interpretation of results from many of the effects of fishing studies conducted worldwide have been hampered by the lack of true control sites. Some studies have used opportunities provided by closed areas (Tuck et al. 1998), but in general, unfished areas are thought to be unfished because they are different from the fished grounds (Hall 1994). Other studies have used areas adjacent to wrecks (or other limitations on access, e.g., cable protection areas) as “pseudo-controls”, on the assumption that the presence of the wreck has prevented fishing activity (Ball et al. 2000), although the presence of the wreck may also affect the adjacent benthic communities in other ways (e.g., increased predation by resident fish).

Access to high resolution fishing effort data is relatively recent development in many areas (e.g., satellite vessel monitoring systems (VMS) data have only been available to European researchers since the mid 2000s). Therefore many fishing impact studies contain the important caveat that the study area may have been markedly affected by previous fishing activities. If the bulk of community change occurred prior to an experimental study being conducted, it may not be possible to detect impacts from fishing, either because the community is resistant to further effects, or because the effects are trivial compared to those caused previously.

These types of studies are often only able to compare experimentally fished with control sites, with logistical issues over the size of experimental areas generally preventing multiple levels of fishing intensity being examined. BACI designs are also considered to be relatively insensitive in detecting effects (Ellis & Schneider 1997). In addition to the difficulties of identifying appropriate control sites and design sensitivity, extrapolation of results from local scale short term manipulative experiments involving experimental trawling to the scale of the fishery is far from straightforward (and open to criticism). This is because experiments occur on much smaller scales in space and time than broad scale fishing effects, and so are conducted over more homogeneous habitats, whereas fishing occurs across a range of habitats, and chronic effects of fishing on benthic communities may accumulate over long periods of time. Also, recovery rates of benthic species are dependent on area and availability of sources of recruitment, and isolated small disturbances do not replicate the typically larger scale of fishing disturbances, which has slower recovery than experimental plots (Hall et al. 1994; Thrush et al. 1996).

The lack of control sites, and the increased availability of VMS data, has led to the increased use of statistical gradient analysis approaches, where habitats or communities are compared over a gradient of environmental drivers and fishing intensity to determine effects, with analysis using regression and ordination techniques. By using analytical approaches that incorporate spatial and environmental factors, these can be accounted for, allowing the analysis to focus on changes in community structure in relation to fishing pressure. These gradient based approaches were pioneered in New Zealand both in fisheries (Thrush et al. 1998; Cryer et al. 2002) and other anthropogenic impact studies (Ellis et al. 2000; Hewitt et al. 2005a). While New Zealand deepwater fisheries have had VMS since 1994, these data are only available for use by the Ministry for Primary Industries compliance team. However, tow by tow reporting of fishing location has been available since 1990 for deepwater fisheries, and since 2007 for inshore trawlers within the standard catch and effort reporting systems, and these data can be used to identify appropriate gradients of fishing pressure over which to investigate benthic community patterns (Baird et al. 2011; Black et al. 2013; Baird et al. 2015).

Given the difficulties in identifying appropriate control sites, the logistical complexities and additional costs involved in repeated sampling of study areas over time, and the difficulties of extrapolating results of small scale experimental manipulations to the scale of the fishery, such an approach was not considered appropriate to meet the specific aims of this project. The availability of high resolution fishing effort data (at least for trawl fisheries) and the scale over which gradient studies can be conducted make them a more appropriate alternative. These gradient analysis approaches are considered to offer the best opportunity to provide statistically powerful analysis of the effects of fishing at the appropriate scale to inform fishery managers. Gradient studies are also subject to poor knowledge of fishing effort patterns prior to the availability of high resolution fishing data, which confounds the effects of recent and historical fishing, and potentially makes actual effects harder to detect.

2.2 Sampling methodologies

A range of sampling methodologies are available for investigating benthic communities and habitats, each with their own advantages and disadvantages, with some more appropriate than others for sampling particular faunal components. Soft sediment organisms create much of their habitat's structure, and also have crucial roles in many population, community and ecosystem processes (Thrush & Dayton 2002; Lohrer et al. 2004). Therefore physical sampling of benthic communities, focusing on both infaunal and epifaunal species is important to confirm taxonomic identifications. However, intensive dedicated benthic community sampling can be very time consuming and expensive (both in conducting the field work and processing and identifying samples), and so alternative approaches have been considered in the comparative studies.

While acoustic and photographic imaging techniques can provide a broad scale perspective of habitat and epifaunal structure, it is important to integrate these approaches with traditional benthic (grab and sled) sampling to provide details of the species groups and functions or roles filled by these species. Fishing activities have been observed to influence marine ecosystems in a number of ways, including effects on infaunal species. The inclusion of infaunal data in the analysis allows for complete

examination of changes in macrofaunal community structure and biodiversity. As sled and grab samples collect animals which can be used in additional analyses, these allow for examination of size structure, production, function, and trophic structure, allowing the implications of changes associated with fishing disturbance to be fully appreciated.

Opportunistic use of existing data

While the combination of acoustic, photographic and benthic sampling approaches provide a powerful approach to the examination of differences in habitats, communities and the roles filled within them, the sampling is demanding in terms of dedicated sea time and equipment, and analysis of existing data can provide valuable additional information (Greenstreet et al. 1999; McConnaughey et al. 2000; Cryer et al. 2002). Opportunistic data sets may not sample the full benthic habitat or community to the extent that the integrated suite of approaches (described above) would, to be useful they should provide a consistent sampling method over large spatial scales that can be examined in relation to depth, location, habitat measures (where available) and fishing activity. For New Zealand waters, potential opportunistic data sets include existing benthic samples (Nelson & Gordon 1997), research trawl survey benthic bycatch, MPI observer benthic bycatch, and seabed photographs.

Several thousand benthic samples have been taken on New Zealand's continental shelf (Figure 1)(Nelson & Gordon 1997), from various biodiversity surveys sampling with benthic grabs and sledges. Much of these data are stored in the NIWA maintained *Specify* database. While such data have been used to describe basic patterns in broad scale macroinvertebrate assemblages (McKnight 1969), many of the older samples were not fully analysed, and the records typically only document presences (rather than abundances) and do not record the full sampled community. These data were examined in detail for coastal regions (down to 250 m) by Baird et al. (2015) for the purpose of examining patterns of sensitivity to, and recoverability from the effects of fishing, and were considered to be inadequate for the purpose. The recent comprehensive benthic faunal community study within the Oceans Survey 20/20 Bay of Islands study (Morrison et al. 2010) is included within this dataset, but there is very little commercial fishing activity in the Bay of Islands area (Baird et al. 2015), reducing its value for effects of fishing studies. Given that most the records do not contain full community data, or even consistent quantitative abundance estimates, they were therefore not considered useful for the present study.

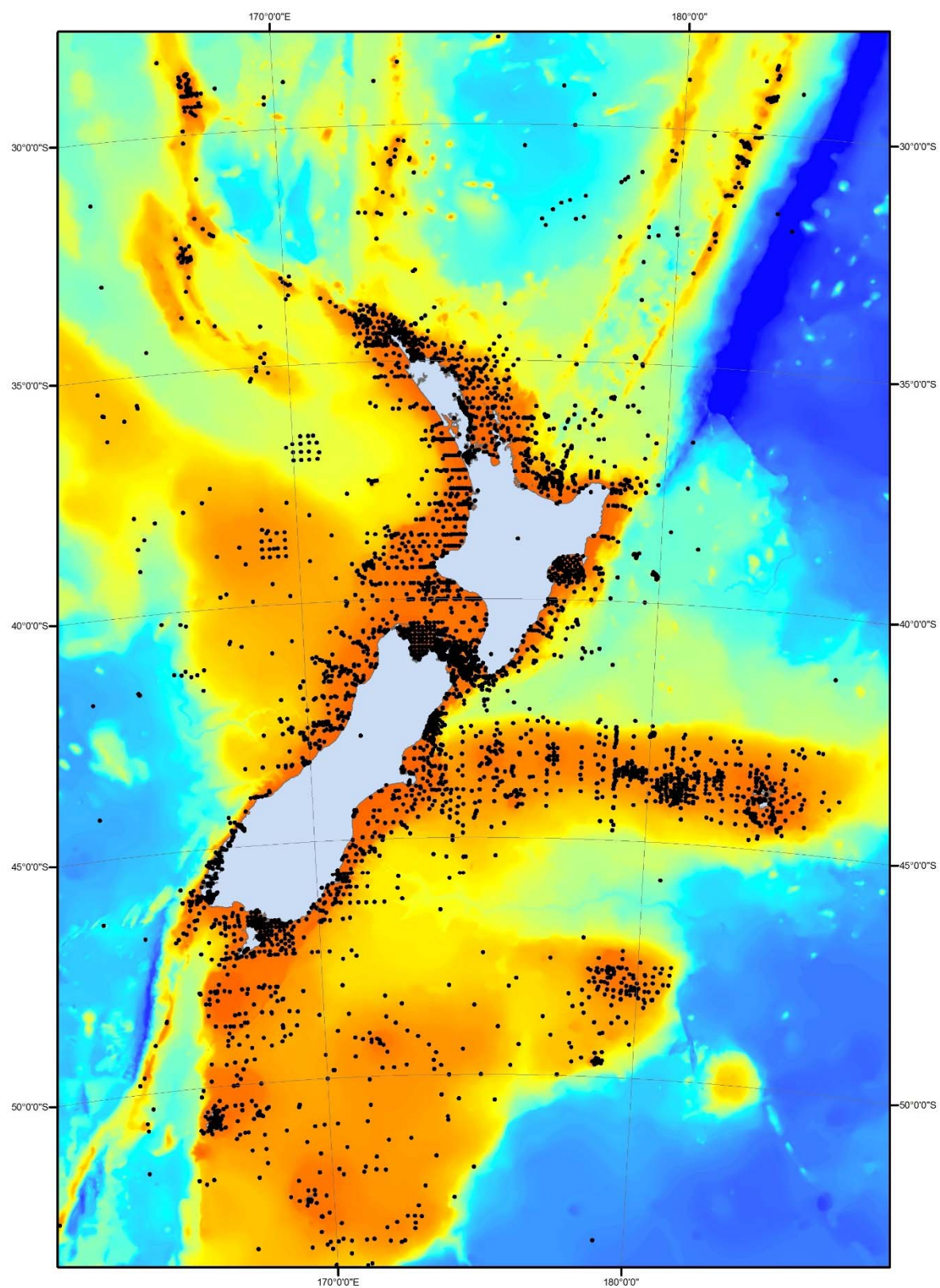


Figure 1: Distribution of benthic samples around New Zealand recorded in the *Specify* database.

Standardised trawl surveys are widely used in fisheries monitoring to provide abundance indices of commercial fish species, either as a monitoring tool in itself, or as an input into a stock assessment model. Since the early 1960s, trawl stations have sampled across wide areas of the seabed shallower than 1000 m (Figure 2), although in more recent years, regular survey coverage has been limited to the Chatham Rise, Sub-Antarctic, east and west coasts of the South Island, and the main scampi stocks. Trawl survey benthic bycatch has been routinely recorded on these trawl surveys for a number of years, and are recorded in the MPI *Trawl* database. While trawl gear may vary between survey series, and is only likely to sample larger epifaunal species in a consistent manner, this database provides a sample of the benthic community (sampled over the trawl track) which can be examined in relation to environmental parameters and fishing pressure. Baird et al. (2015) had concerns over the use of trawl survey bycatch data in examining patterns of sensitivity to fishing disturbance, although this was across surveys (using different fishing gear). We consider the use of trawl survey bycatch data here (within surveys using the same gear type) to be appropriate. Cryer et al. (2002) examined the effects of fishing on benthic communities from trawl survey bycatch (see Section 6.1), but that study involved a taxonomic specialist identifying the complete benthic invertebrate catch. During routine survey operations, benthic fauna are identified by experienced staff using identification guides (e.g., Tracey et al. 2011a) with some species returned to the laboratory for expert identification, but the level of taxonomic accuracy and resolution are generally unlikely to match that attained by Cryer et al. (2002).

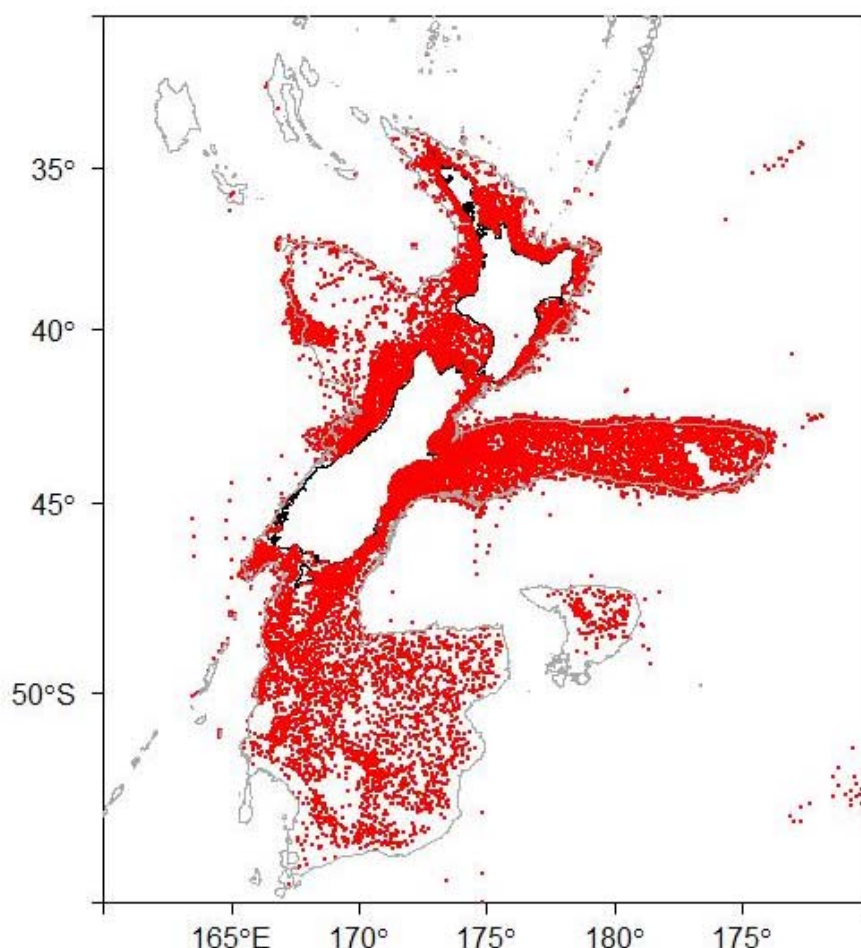


Figure 2: Distribution of trawl survey stations around New Zealand. Each red symbol represents a station recorded in the MPI *Trawl* database, with data going back to the early 1960s. Grey line represents the 1000 m depth contour.

In addition to standardised trawl surveys, data on the catch composition of commercial fishing activities (including bycatch) are also recorded by MPI observers on commercial vessels. While collected on a less strategic basis than surveys, with locations being determined by fishing operations (and generally providing a good representation of the main fishing areas), data are available for deepwater fisheries back to the early 1990s (observer coverage in inshore fisheries being at a far lower level). However, while these data are routinely used to estimate bycatch and discards in deepwater fisheries (e.g., Anderson 2012), or distributions of specific species groups (Tracey et al. 2011b), the resolution of taxonomic identification for some groups has changed markedly over time, and the data are not considered sufficiently consistent (in identification or sampling gear) to be used as the basis for benthic community indicators (Tuck et al. 2014).

Seabed video and photographic surveys have been widely used as non-destructive approaches for sampling epifaunal communities in biodiversity and effects of fishing studies (e.g., Thrush et al. 1998; Bowden 2011; Tuck & Hewitt 2013; Lambert et al. 2014), and have been employed in the case study surveys. A series of photographic and video surveys have been used to survey deepwater seamounts around New Zealand, but of more relevance to this particular study (focusing on soft sediments) are the photographic surveys used to provide an index of scampi burrows (e.g., Tuck et al. 2015a). Other historic seabed photographic datasets are known to exist but they have not been appropriately documented or digitized, and have not been considered for analysis here. Photographic survey approaches for scampi were first used in New Zealand in 1998 (Cryer et al. 2005), with coverage expanding into all the main scampi fisheries over time. Since 2010 within the MPI 10-Year Deepwater Fisheries Research and Monitoring Programme the four main scampi grounds are scheduled to be surveyed every three years.

On the basis of discussions at the project workshop, it was concluded that both trawl survey benthic invertebrate bycatch and seabed images from the scampi surveys would provide useful data sets, to be examined for some areas. The overall approach adopted was therefore to apply grab, sled and photographic approaches within the case study areas, and also examine any appropriate opportunistic data available for the same regions. Opportunistic trawl bycatch and seabed image data were also compared with each other in areas where both were available.

By adopting a hierarchical approach to sampling (combining the opportunistic and integrated approaches for the most comprehensive analysis in some areas, and examining opportunistic data alone for others, where the combined approach was not possible) will allow the conclusions drawn from these other approaches to be considered in relation to those from the integrated photographic and benthic sampling techniques.

2.3 Statistical approaches

A range of statistical approaches have been employed in examining the data from the different case studies.

Principal component analysis (PCA) was used in the data exploration to examine the ordination of sites relative to environmental variables, to identify any strong environmental gradients. Pairwise correlations were examined between explanatory variables, and when high correlations were identified, some variables were excluded from the subsequent analysis.

Non-metric multidimensional scaling (MDS) of the benthic communities was used to provide a preliminary examination of the spread of sites across ordination space relative to the fishing effort or other variables. MDS constructs a map of the samples (typically in two dimensions) which attempts to satisfy the conditions imposed by a rank dissimilarity matrix of the species compositions (positioning samples relative to each other to satisfy their relative dissimilarity), and can be used to identify how samples compare, and relate to other variables. Where differences in community composition show a good spread in ordination space, rather than being driven by a few distinctly different sites or clusters, analyses of drivers of community composition using continuous variables is likely to be robust.

The relationships between the benthic community at each site, environmental drivers and fishing pressure were examined using Distance-based linear modelling (DistLM) on normalised environmental variables (Anderson 2001; McArdle & Anderson 2001) within PERMANOVA+ for PRIMER (Anderson et al. 2008). DistLM partitions variation in a data cloud, as described by a resemblance matrix, according to a multiple regression model. Importantly, it supports the use of a number of different distance measures, including the frequently used Bray-Curtis similarity measure, and can be used in backwards selection mode, employing a range of model selection criterion. While both Redundancy Analysis (Van Den Wollenberg 1977) and Canonical Correspondence Analysis (ter Braak 1986) also partition variance in a data cloud, according to a multiple regression model, these two analyses are confined to the use of Euclidean and chi-square distances respectively, which are not used quite so frequently in analyses of community data. Moreover, there is no software package that simply allows backwards selection of variables. Instead forward selection is the available option, despite backward selection being more suitable when interactions and some correlations exist between explanatory variables. However, to ensure that results gained were not wholly driven by analysis type, we did analyse the datasets using Canonical Correspondence Analysis (CCA) techniques as well. Because these results matched those from DistLM, we do not present them here.

The approach adopted was to initially examine the effects of fishing effort variables, then add environmental variables, and finally add spatial variables. Given the focus of the study on the effects of fishing on benthic communities, this allows for examination of the effects of fishing terms prior to the inclusion of any correlation with environmental or spatial drivers, but still accounts for the effects of these other terms in the final model selection. Correlation with terms introduced into the model at a later stage may mean that terms initially retained may be dropped from the full model, but consideration of their individual explanatory power in relation to the variance of the response variable is still relevant. DistLM marginal tests for all terms considered within each model are presented in the Appendices.

While DistLM offers a number of advantages over alternative approaches (as discussed above), and is able to attribute proportions of the total variability to factors, it does not provide a plotting method to examine the effects of individual factors on individual species. To identify the species particularly sensitive to fishing (either positively or negatively), we have used Constrained Analysis of Principal Coordinates (Anderson & Willis 2003) to examine the correlations of individual species abundances with fishing effort, having taken into account (partialled out) the effects of the other retained explanatory variables. These analyses were conducted with the *capscale* function within the R library *vegan*. Constrained Analysis of Principal Coordinates (CAP) is an ordination method similar to RDA that allows the use of non-Euclidian dissimilarity indices.

The various univariate metrics considered (species richness, diversity, evenness, biomass and productivity) were also examined in relation to the fishing pressure and environmental variables within a generalised linear modelling framework, with minimum adequate model selection on the basis of AIC, providing an alternative approach to investigate the effects of individual terms on the response variable. Terms retained as significant by this and the DistLM approaches may differ owing to the correlation between terms and differing model selection criteria.

The epifaunal (image) and infaunal (grab) data were analysed separately, but in both cases, both raw data and square root transformed (to reduce the influence of the most dominant species) data were used. The analyses based on square root transformed data only are presented here as initial analyses demonstrated that similar results were found. This suggests that the results were not just driven by changes in the most abundant taxa. Indicators of biodiversity (number of taxa, Pielou's evenness and Shannon-Weiner diversity) were also calculated, at the site level, for both epifaunal and infaunal data separately. Number of taxa is a simple measure of the total number of species recorded at a site. The Shannon-Weiner diversity index accounts for both abundance and evenness of the species present, while Pielou's evenness index reflects how evenly distributed species are as a ratio (calculated as observed Shannon-Weiner diversity / maximum possible Shannon-Weiner diversity if all species present were equally abundant).

Backwards selection was used to select the most important variables. Following this, the effect of the fishing variables alone was determined, then the increased amount of variability explained by the other variables. Finally, spatial variables were included to determine whether there was any significant spatial structure left in the data.

In addition to the species based community analysis approaches described above, we also examined abundance within functional groups (defined on the basis of life history characteristics; Lundquist et al. (2013)) or key species across gradients of fishing pressure. Lundquist et al. (2013) allocated fauna collected within the Tasman and Golden Bays and Chatham Rise / Challenger Plateau case studies to eight functional groups using a species functional traits database developed over a series of NIWA commercial and core funded research projects. Abundance within these functional groups was then modelled in relation to the effects of fishing, and validated from the field data. Functional groups 4 (substrate destabilisers – surface dwelling, mobile, deposit feeding) and 6 (emergent epifauna – long-lived, sedentary, habitat-forming species) from this functional group approach are considered to be the most likely to be affected by fishing, and have been examined in relation to fishing effort gradients. Where detailed life history data were not available (for some data sets), key individual species have been examined in a similar way (as proxies for functional groups). Habitat complexity is often associated with increased biodiversity (Hewitt et al. 2005b; Mormul et al. 2011; Buhl-Mortensen et al. 2012), and so in the analysis of infaunal community data (Tasman and Golden bays, and South Canterbury Bight studies), the abundance of Functional group 6 (emergent epifauna – long-lived, sedentary, habitat-forming species) was also included as an explanatory term.

2.4 Design decisions

For the case studies where new data were being collected, key design decisions were made on the basis of existing data from similar studies.

Power, precision and sample size

Power/sample size relationships were used to suggest the number of sites required, and precision/sample size relationships were used to determine the number of within-site replicates needed to achieve good estimates of site means within the project budget. Formal hierarchical analyses of power were not used for the following reasons:

- The usefulness of these analyses depends heavily on the data available for the power analysis having the same variance structure across spatial scales as the data to be collected. This is impossible to ensure when data from the potential survey area are either not available, or were not collected at the same spatial scale or over the same environmental gradients. Moreover, changes in variance structure are common over time.
- They are, at present, not readily available for multivariate data.

Using less formal techniques allows us to incorporate available information from other areas, integrate cost-benefit analyses and assess multivariate precision/sample size relationships. Data for investigating these less formal relationships were available from three sources:

- video (2 replicates per site) and grab (5–7 replicates per site) data from a study of fishing effects in the Hauraki Gulf (Thrush et al. 1998);
- video (1 replicate per site) and grabs (12 replicates per site) from Separation Point (Handley et al. 2014);
- video (1–2 replicates per site) from Chatham Rise and Challenger Plateau (Bowden & Hewitt 2011).

Curves of increasing precision (standard errors about the mean) with increasing sample size were determined for data having within-site replication (i.e., grab data from the Hauraki Gulf and Separation Point) as described in Bros & Cowell (1987) and Hewitt et al. (1993).

Precision for community data within sites for the Hauraki Gulf grab data was calculated following Anderson (2001), as a pseudo multivariate standard error (square root (q/n)) where q is average square Bray-Curtis dissimilarity and n is sample size, for increasing sample size from random draws of data. Number of sites and effect sizes for detecting differences between categories (e.g., differences between different levels of fishing effort) can be determined, assuming equal numbers of sites in all comparisons, using:

$$\sigma_m = \sqrt{\left(\sum_1^k (m_i - m)^2 / k\right)}$$

$$f = \sigma_m / \sigma$$

where: where m is the overall group mean, m_i is the mean for group i , f is effect size, σ is the standard deviation of the group or class mean, σ_m is the standard deviation of the average of all group means, and k is number of groups (i.e., habitats in this analysis). For the Chatham Rise and Challenger Plateau video data, estimates of the power of the analysis to detect differences between observed categories were then derived for effect sizes ranging from 0.5 to 0.8, based on $\alpha = 0.05$ for a two tailed test, from the tables in Cohen (1988).

However, this project intends to use gradient (regression) analysis as opposed to categorical analysis. This involves testing that fishing effort makes no unique contribution to the overall R^2 found for the regression (this is equivalent to the null hypothesis that $\beta_{y,F}=0$, i.e., the slope related to fishing effort = 0). Effect sizes for this test were determined from video and grab Hauraki Gulf data as:

$$f^2 = (R_{y,a,F}^2 - R_{y,a}^2) / (1 - R_{y,a,F}^2)$$

where $R_{y,a,F}^2$ is the proportion of variance explained by fishing effort and other environmental variables and $R_{y,a}^2$ is the proportion explained by other environmental variables only. Power then becomes the ability to detect a relationship between the dependent variables (in particular fishing effort) and the response variable.

Within-site replication

There was limited information on which to base a decision about replication of videos within sites. Clustering of data from the 2 km long video transects on the Chatham Rise and Challenger Plateau, generally assigned replicates to the same community association (Floerl et al. 2012). Data from video transects within sites from the Hauraki Gulf data generally were more similar than between sites (J. Hewitt NIWA, unpubl data). Given that deploying the video gear is time consuming, in the present studies it was decided to use a single video transect within each site covering a large area of ground (about 200 m). This length of transect is between that used by the Thrush et al. (1998) study and that used at Spirits Bay (Tuck & Hewitt 2013), both of which have successfully detected fishing effects.

Curves of increasing precision with increasing sample size for grab data are shown in Figure 3 – Figure 5.

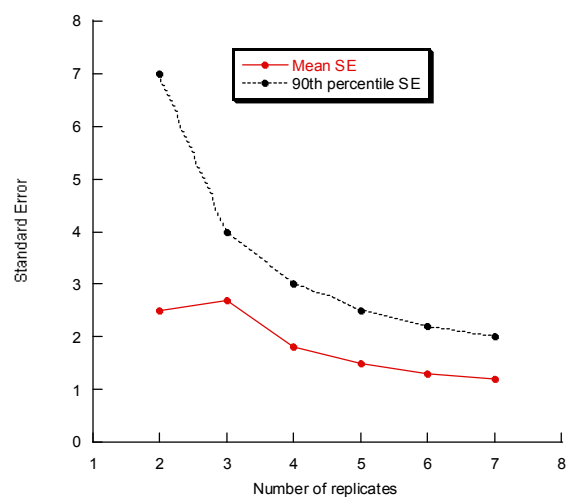


Figure 3: 90th percentile and mean standard errors observed from random draws of number of taxa from Hauraki Gulf grab data.

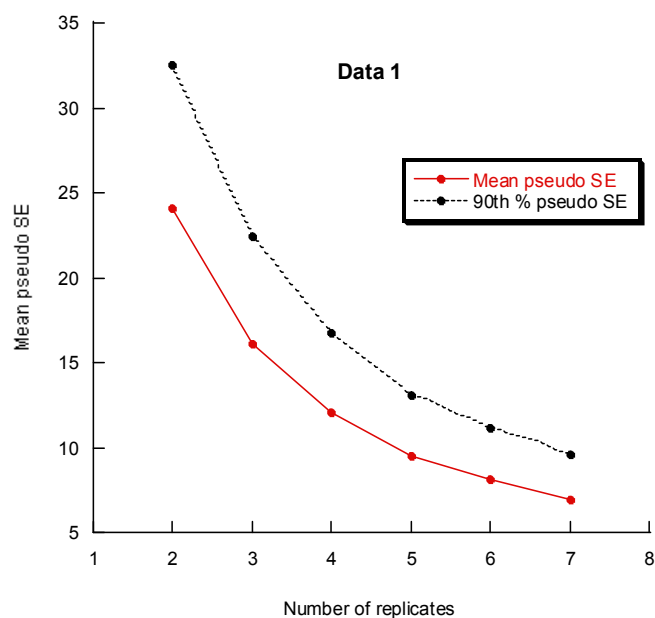


Figure 4: Estimated precision for community data from random draw of grabs within a site from the Hauraki Gulf data using a pseudo estimate based on Anderson (2001).

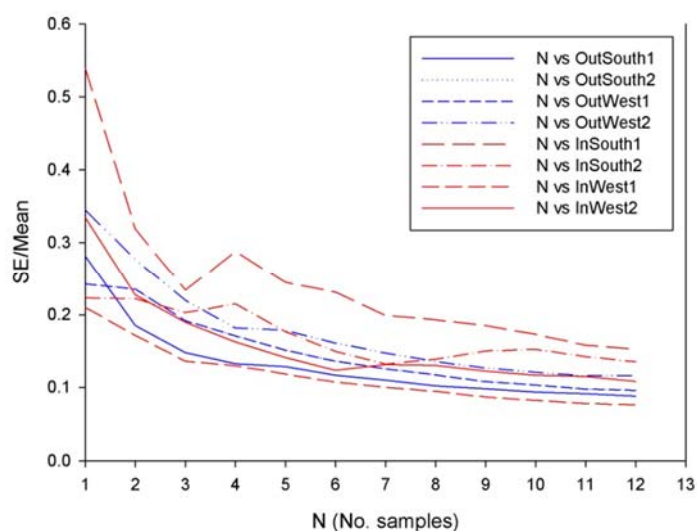


Figure 5: Mean standard errors observed from random draws of number of taxa from grab sampling at different sites in Separation Point.

While the maximum increase in precision about the mean had occurred by three replicates for samples from both the Hauraki Gulf and the Separation Point studies, curves generally did not show a strong levelling off and some increase in precision was still occurring after seven replicates. Therefore, for the Tasman Golden Bay case study, the decision was made to take ten replicates and process seven initially, modifying the analytical process depending on results. On the basis of the results from this initial case study, the approach employed for the South Canterbury Bight case study was to maximise the number of sites with less emphasis on within-site replication, and so 3 grabs and 1 sled tow were sampled at each location.

Number of sites

Using video data from the Chatham-Challenger surveys, Bowden & Hewitt (2011) calculated power curves for distinguishing differences between habitats (Figure 6). The range of effect sizes was based on a comparison of the magnitude of differences between values of the assemblage metrics between *a posteriori* habitats, and between *a priori* sampling strata. Effect size can range from 0 (no similarity between groups) to 1 (complete similarity) and for the Chatham-Challenger data were found to range from $f = 0.5$ to $f = 0.8$. The results of this analysis were plotted as the number of sites that would need to be sampled within each habitat in order to differentiate reliably between all habitats present in the survey area. Results were plotted separately for effect sizes from $f = 0.5$ to $f = 0.8$ (Figure 6).

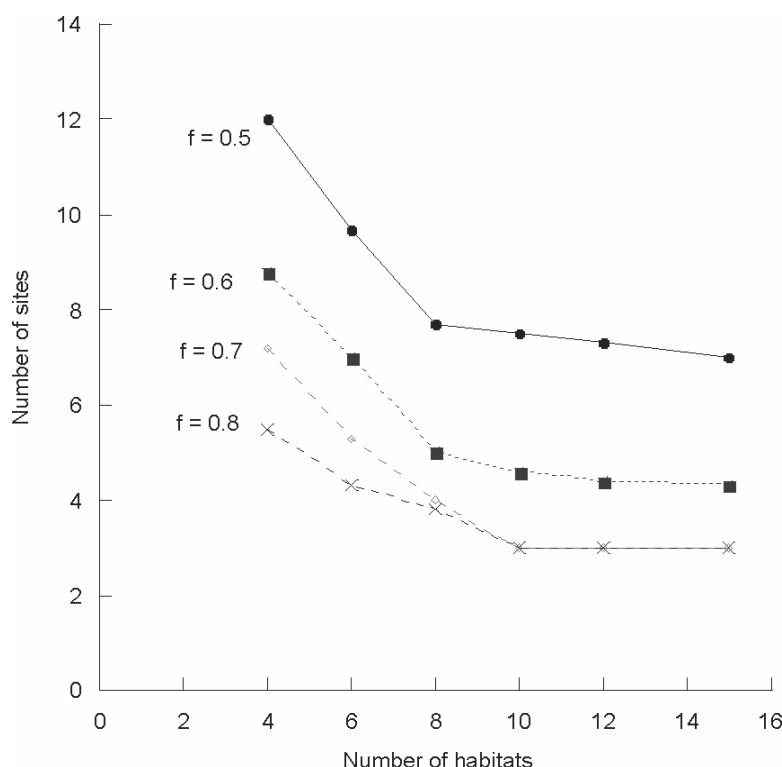


Figure 6: Power analysis: the number of sampled sites required per habitat to achieve an 80 % probability of detecting differences between habitats. Values are shown for four effect sizes ($f = 0.5$ to $f = 0.8$) as proportional differences between the mean values of assemblage metrics between habitats.

Utilising these results for this project, if there were a total of 10 habitats and each of these were, on average, 50 % different from the others for a given assemblage metric (i.e. effect size, $f = 0.5$), more than 7 videoed sites would need to be sampled within each habitat to achieve an 80 % probability of differentiating between them. If we translate this into trying to detect differences between high, medium, low and no fishing effort in the present study, we would need 12 sites in each category. However, this is an overestimate as ANOVA-type techniques, such as power analysis, are notoriously inefficient at detecting gradient effects, thus fewer than 12 videoed sites per fishing category would probably be needed.

When the Hauraki Gulf data were initially analysed by regression (Thrush et al. 1998) the overall proportion of variability ($R^2_{y,a,F}$) observed varied from low (0.1) to high (0.5). Power curves derived from these data for multiple regressions (Figure 7) demonstrate that, for $\alpha = 0.05$ and number of dependent variables in the regression analysis equalling 4, 80% power to explain proportions of variance (R^2) between 0.3 and 0.4 could be achieved with as few as 30 sites. Conversely for very low R^2 (0.1), more than 60 sites would be required to even achieve 50% power. Between 30 – 40 sites would give power to detect reasonable effects on diversity (R^2 varying between 0.2 – 0.3).

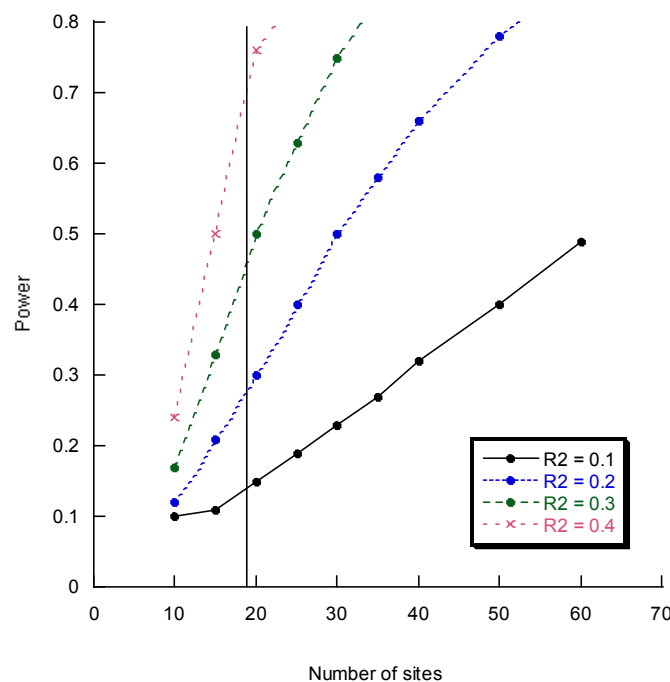


Figure 7: Power curves for the relationship between power and sample size based on 4 variables and $\alpha = 0.05$. Line indicates the 18 sites used in the Hauraki Gulf analysis by Thrush et al (1998).

2.5 Study locations

Studies examining the effects of fishing on soft substrate benthic communities have been conducted previously in New Zealand (Thrush et al. 1995; Thrush et al. 1998; Cryer et al. 2002; Tuck & Hewitt 2013), including the Oceans Survey 20/20 voyages (Bowden 2011) providing data for the deepwater (Section 5). The identification of appropriate locations for the two coastal case studies to be conducted within the research therefore considered previous studies, with a view to providing a good coverage across geographic and habitat ranges.

Previous New Zealand studies have focussed on exposed sandy habitats of Spirits Bay (25–70 m) (Tuck & Hewitt 2013), more sheltered sandy and muddy habitats within the Hauraki Gulf (15–30 m) (Thrush et al. 1995; Thrush et al. 1998) and deeper (200–600 m) muddy habitats in the Bay of Plenty (Cryer et al. 2002), all off the North Island. Also, during the course of the current study, further relevant research has been conducted in the Hauraki Gulf (Morrison et al. 2016).

Strong consideration was given to a study examining the effects of oyster dredging on the gravelly habitats of the Foveaux Strait. While official reporting only documents fishing effort at very broad statistical areas, the fishers maintain finer scale (1 n.m. $\times$ 1 n.m.) catch and effort reporting (Michael 2009), which would provide data on fishing effort patterns. However, the very patchy nature of the fishing effort (even within these fine scale grids), the strong physical drivers of community structure in Foveaux strait and the potential that the long history of fishing in the oyster fishery had already had substantial impact on benthic communities (Cranfield et al. 1999) led to the conclusion (also shared by MPI and an international reviewer) that the logistical difficulties of conducting a statistically robust study in the area would be beyond the resources of the project.

Areas covered by the trawl surveys on the east (Beentjes et al. 2013) and west (MacGibbon & Stevenson 2013) coasts of the South Island provided the opportunity for the examination of bycatch data, and so were given priority. Within the Tasman and Golden Bays region near Nelson a range of previous studies have examined benthic communities (Grange et al. 2003; Hewitt et al. 2004; Handley et al. 2014),

particularly focussing on areas closed to fishing within the region. While data from these studies were eventually excluded from the final analysis as the habitats within the closed areas (particularly Separation Point) were considered to be unique, the background data from the region was still considered to be useful. The decision was taken to conduct one of the case studies in the relatively sheltered sandy and muddy habitats of Tasman and Golden Bays, looking at the effects of inshore trawling, and scallop and oyster dredging (Section 3).

The South Canterbury Bight area was selected as the location for the second case study (Section 4), on relatively exposed sandy and muddy habitat between 20 and 130 m. This area is important for inshore commercial trawlers, targeting a range of demersal species. This area does not benefit from the same level of historical study as the Tasman and Golden Bays region, but the trawl survey does provide a useful opportunistic data set for comparison.

Given that Cryer et al. (2002) successfully used trawl survey bycatch (albeit identified by a benthic specialist) to identify effects of fishing in the Bay of Plenty scampi fishery, it was also considered useful to examine other bycatch data collected and identified by the same researchers but not analysed at the time (off the Wairarapa coast) and more recent survey bycatch and seabed photographs from both of these areas and the scampi fishery to the east of the Auckland Islands (Section 6).

Combining across all the case studies, opportunistic analyses and previous investigations, this report documents studies of the effects of fishing studies across the latitudinal range of New Zealand, in habitats ranging from relatively sheltered shallow bays and harbours to very exposed areas and deepwater communities down to over 1000 m (Figure 8, Table 1).

Table 1: Summary of New Zealand studies included within this report.

Area / study	Depth range (m)	Sediment types*	Fisheries†	Source
Spirits Bay	30 – 100	S, G	SCA, IT	Tuck & Hewitt (2013)
Hauraki Gulf	10 – 40	S, M	SCA, IT	Thrush et al. (1998)
Bay of Plenty	200 – 600	M	SCI	Cryer et al. (2002)
Bay of Plenty (2012 data)	300 – 450	M	SCI	Present study
Bay of Plenty (2008 data)	300 – 450	M	SCI	Present study
Wairarapa	220 – 450	M	SCI	Data collected by Cryer
Hawke Bay / Wairarapa	200 – 500	M	SCI	Present study
Nelson Bays	10 – 45	S, M	SCA, OYS, IT	Present study
Chatham Rise / Challenger Plateau	100 – 1800	M	SCI, MT, DT	Present study
Chatham Rise / Challenger Plateau	100 – 1800	M	SCI, MT, DT	Present study
Sth Canterbury Bight	40 – 120	S, M	IT	Present study
Auckland Islands	350 – 520	M	SCI	Present study

* Key to sediment types: M – mud; S – sand; G – gravel

† Key to fisheries in each study area: SCA – scallop dredge; IT – mixed inshore trawl; SCI – scampi trawl; OYS – oyster dredge; MT – mixed middle depth trawl; DT – mixed deepwater trawl.

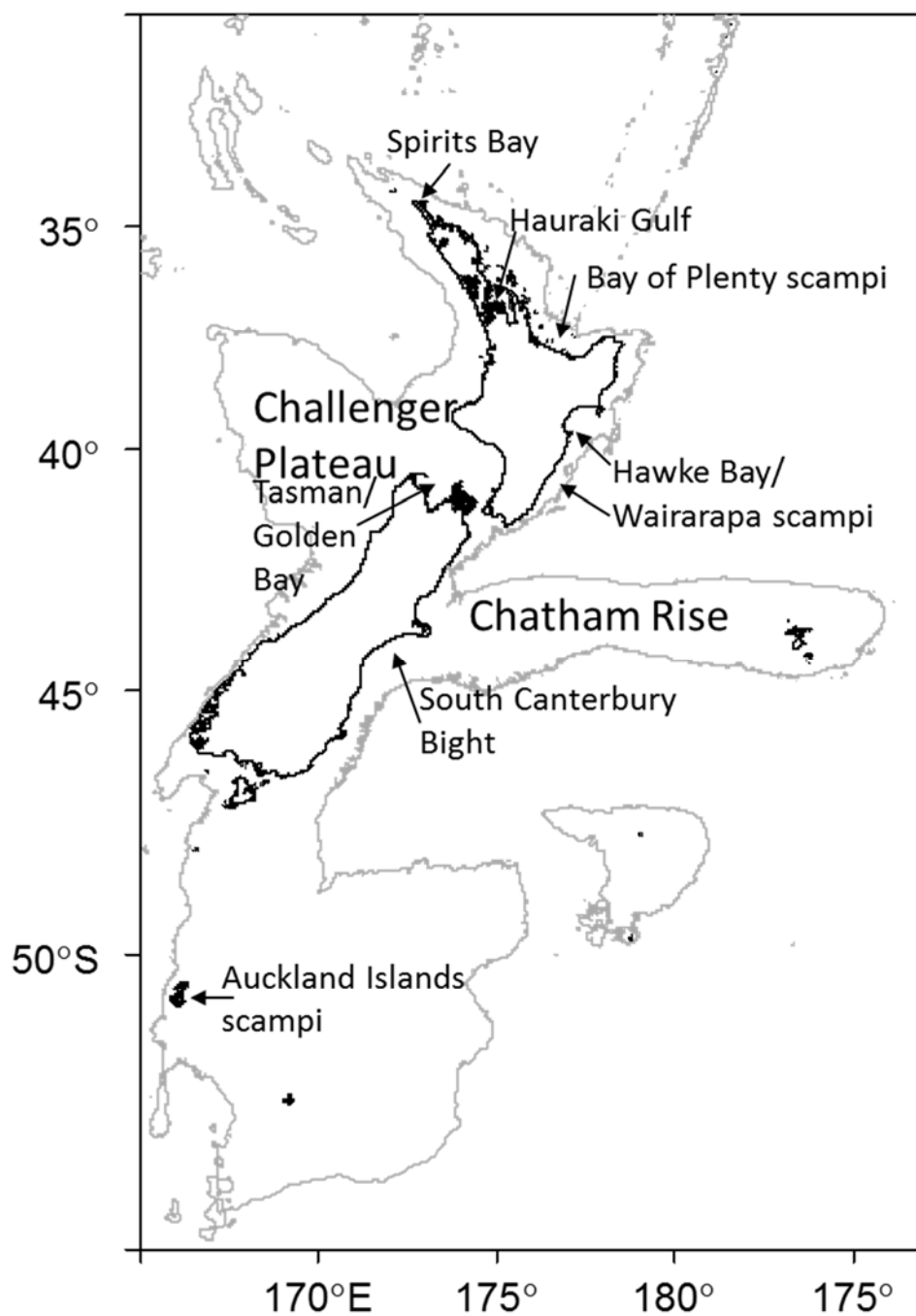


Figure 8: Locations of study sites included within this report. Grey line represents 1000 m depth contour.

3. TASMAN AND GOLDEN BAYS

3.1 Dedicated benthic community sampling

3.1.1 Methods

Study area

The study was conducted in Nelson's Tasman and Golden Bays where the seabeds of the two bays slope gradually out to 40 m depth (Figure 9). The Golden and Tasman Bay seabeds are intensively fished, by trawling and seining for finfishes including flatfish species, barracouta (*Thyrstites atun*), snapper (*Pagrus auratus*) and red cod (*Pseudophycis bachus*) and have also been intensively dredged for scallops (*Pecten novaezelandiae*) and oysters (*Ostrea chilensis*). Recreational fishers are also permitted to dredge for scallops and oysters.

Tasman and Golden Bays comprise one of the most intensively fished and managed areas of photic zone seabed in New Zealand. The scallop fishery was managed spatially by rotationally fishing sectors annually until 2005/06 when the Tasman Bay stocks declined and enhancement of juvenile stocks failed. After 2007, commercial harvest of scallops was largely restricted to Golden Bay, west of the exclusion zone, and the Marlborough Sounds (Mitch Campbell, Challenger Scallop Enhancement Co., pers. comm.). Dredge oysters populations are most abundant in Tasman Bay to the south east of Separation Point (Brown 2012).

Sedimentation rates in Tasman and Golden Bays are high compared to other shelf areas of New Zealand (van der Linden 1969). Sediments are introduced from (i) the west coast, and (ii) rivers within the bays (Figure 9). There is a prevailing northward drift of currents and sediment along the west coast of the South Island, with rapid deposition of sediments on Farewell Spit. Prevailing clockwise current rotation in Golden Bay transports the silt and coarse sand from the Aorere and Takaka Rivers and wind-blown sand originating from Farewell Spit. In Tasman Bay, the Motueka and Waimea Rivers discharge sandy silt and calcareous gravel material. The Aorere and Takaka Rivers, however, contribute four times the amount of sediment (2.7×10^6 tonnes.yr⁻¹) than the Moteuka and Waimea Rivers (0.7×10^6 tonnes/yr; Griffiths & Glasby 1985). Furthermore, Golden Bay sediments are sandier than those in Tasman Bay (Churchman et al. 1988). Sediments that move north past Farewell Spit and into Tasman Bay and Cook Strait also contribute to deposition. Coastal erosion is a very minor source of sediment (University of Auckland 1975).

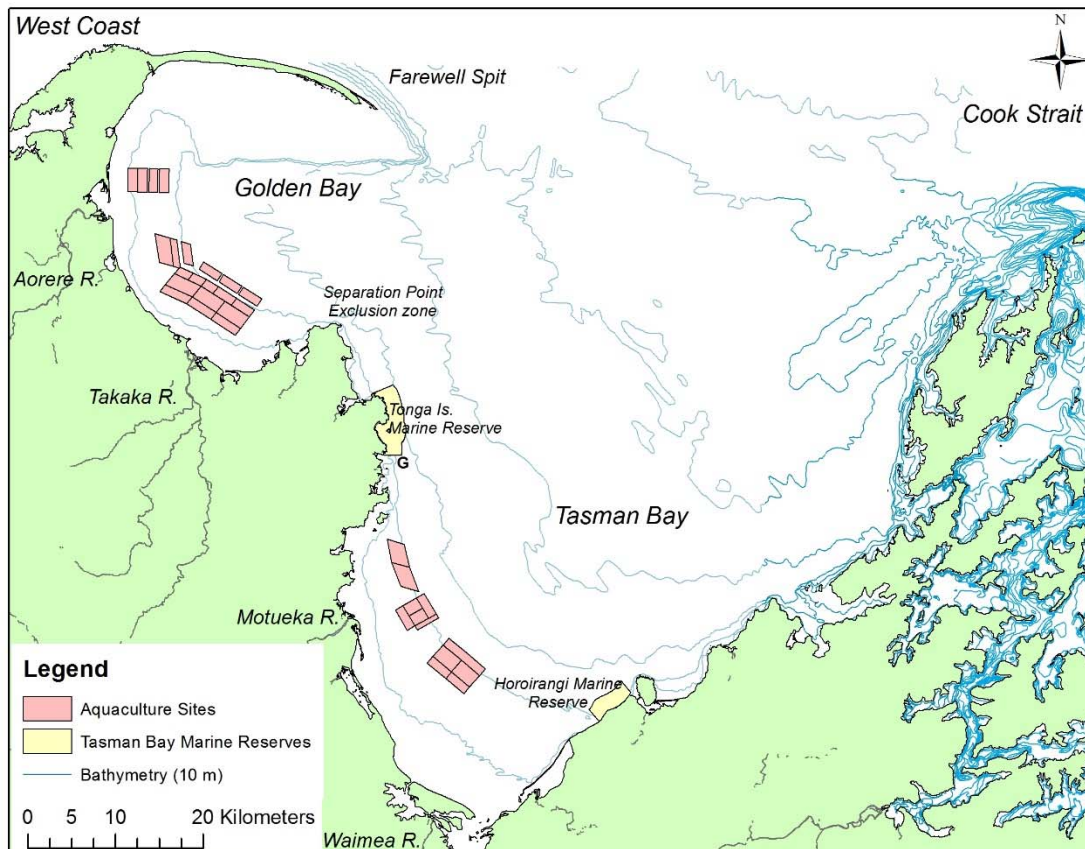


Figure 9: General location map of Tasman and Golden Bay study area.

Environmental and fishing data used

Environmental data

Hydrodynamic models were used to estimate sediment transport and other physical environmental parameters that might result in changes in benthic community structure, in addition to fishing. Current speed in Tasman and Golden Bays was modelled using a NIWA ROMS model, which is a widely used ocean/coastal model (Haidvogel et al. 2008; Warner et al. 2008; MacCready et al. 2009; Zeldis et al. 2011a). The model was set up on a rectangular 130×128 grid with spacing of 1 km as specified in Zeldis et al. (2011a). A preliminary analysis of the tides in the Nelson Bays model shows that it agrees with the NIWA EEZ tidal model (Walters et al. 2001). Validation of this model shows that the main factor limiting the accuracy of the residual currents generated by the hydrodynamic model is likely to be the limitations of the wind data sets from which the surface stress is calculated. While the model is imperfect, it currently provides the best available basis for simulating transport and mixing in the Nelson Bays system (Zeldis et al. 2011a). The outputs used for the gradient analysis were the mean (Figure 10) and maximum near bottom model layers.

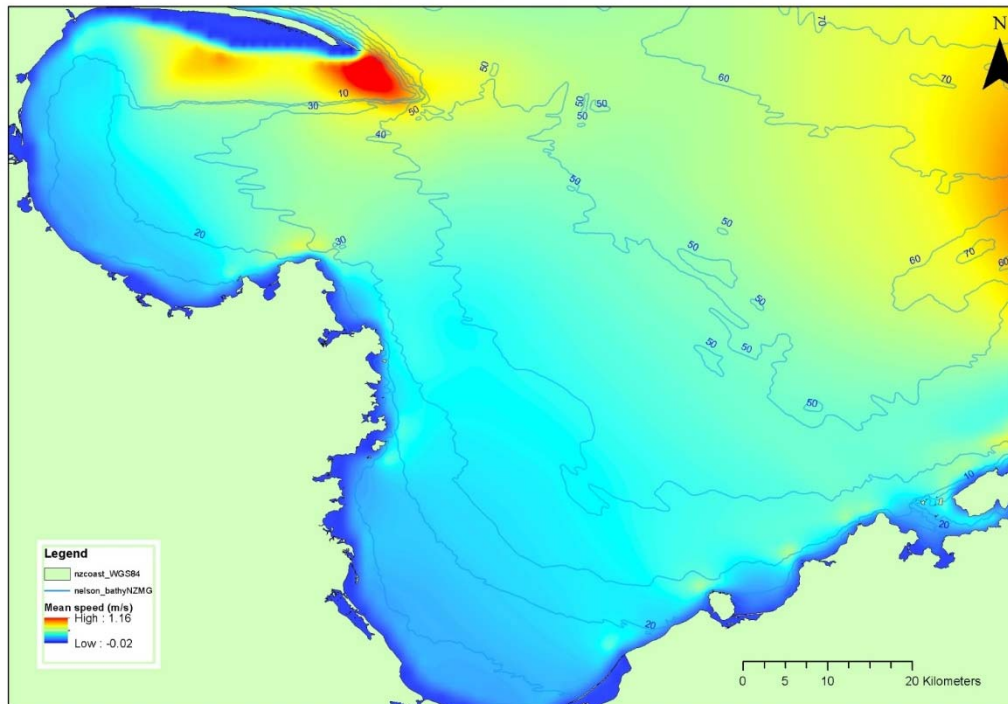


Figure 10: ROMS model output of mean near bottom current speed for the Tasman and Golden Bays study area.

Wave modelling was carried out using NIWA's operational forecasting system called NZWAVE_12 which incorporates wind inputs from the weather forecasting model NZLAM_12 where the "12" in both instances indicates a horizontal grid spacing of 12 km, and both these models are nested in coarser-scale global models. The finer resolution improves the prediction of wave generation by wind, as well as the propagation of waves into near-shore waters (Zeldis et al. 2011b). Output wave and wind statistics were available for the 24 months from March 2009 through to February 2011. A simulation of wave conditions in the Greater Cook Strait during January 2008 was then nested inside the NZWAVE_12 domain using a grid of approximately 1 km. The same 12 km resolution wind fields were used as for the operational forecasting, but the finer resolution allowed nearshore wave processes to be better simulated.

Salinity was simulated from a hydrodynamic model over half-hourly intervals between Aug 27 2009 and March 11 2010. The model used NZLAM output for surface winds (12 km resolution), with river inputs for Aorere, Motueka, Riwaka, Takaka and Wairoa. The resolution of hydro model was about 1 km in the horizontal and 10 layers in the vertical. For each surveyed site position, six salinity statistics were calculated: maximum, minimum, upper and lower quartiles, median and mean.

Fishing effort data

To model the spatial extent of bottom trawling, data were extracted from the Trawl Catch Effort Return (TCER) and Trawl Catch Effort and Processing Return (TCEPR) landing statistics database held by the New Zealand Ministry for Primary Industries (MPI). Fishers are required to record the start trawl positions but not end positions to degrees and nearest minute along with tow speed, duration and door-spread. The data were first plotted and cleaned of spurious records (about 1.6% of the data from within Tasman and Golden Bays) including records on the land and within reserves. We then estimated the area of seafloor potentially swept by each trawl (tow distance times swept width) from target catch data for flatfish species, barracouta, snapper, red cod, and red gurnard. One vessel recorded both start and end positions of tows which provided a check to determine the accuracy of our estimates of distance towed, and confirmed that the approach provided reasonable estimates.

Initial attempts at interpolating these data used the start positions only, to identify gradients of fishing pressure and inform site selection. The truncation of latitude and longitude coordinated within the database meant that a number of tows were recorded at the same start location, and for each start location, tows were summed to estimate cumulative area swept. A trawl “impact factor” was then derived taking into account the date of each trawl, with recent trawls given heavier weight by the formula:

$$\text{Impact factor} = (574 - (\text{trawl date} - \text{start date})) \times \text{estimated area swept}$$

where start date was 01/10/2007, and 574 was the number of days covered by the Ministry’s database preceding the date of our sampling (since 01/10/2007). The Impact factors were then interpolated in ArcMap v 9.3.1 (ESRI Inc. 1999-2009) using the Spatial Analyst Krig interpolation (Figure 11). The coastal overlay was used as a barrier with no smoothing. This approach has a number of limitations (including those introduced by the truncation of spatial coordinates, and failure to account for deviations from the straight line trawl track), and the effort maps generated were not used in the analysis of the effects of fishing on benthic communities.

In response to reviewers’ comments following presentation of the initial analysis of the data, an alternative approach was devised (similar to that later implemented by Baird et al. (2015)) to plot consecutive tows undertaken by each vessel for each trip number for each day recorded in the database. This involved plotting lines between consecutive trawl start positions, per vessel, per day (data excludes single trawls that fell on the same day = 7.8 % of original data) (Figure 12). The "Line Density" tool in ArcInfo v 10.0 (ESRI Inc. 1999-2010) was used to interpolate the trawl lines giving each line a weight based on the estimated area towed (calculated as above). This approach, while not a definitive map of trawls as could be expected from GPS tracking systems (e.g., www.vms.co.nz), appeared better than the previous interpolation derived from just the start positions, and shows some clear patterns in the trawl effort (Figure 12). This approach also overcame the issue of how to treat multiple tow start positions recorded at the same truncated coordinates.

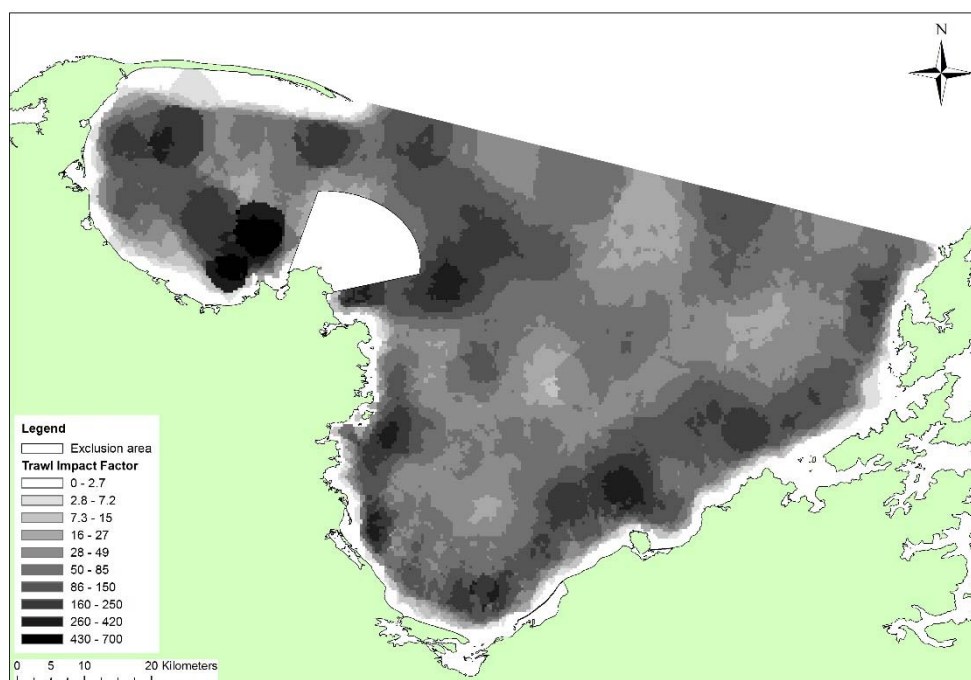


Figure 11: Krig interpolation of “trawl impact factors” derived from trawl start positions for the Tasman and Golden Bays study area.

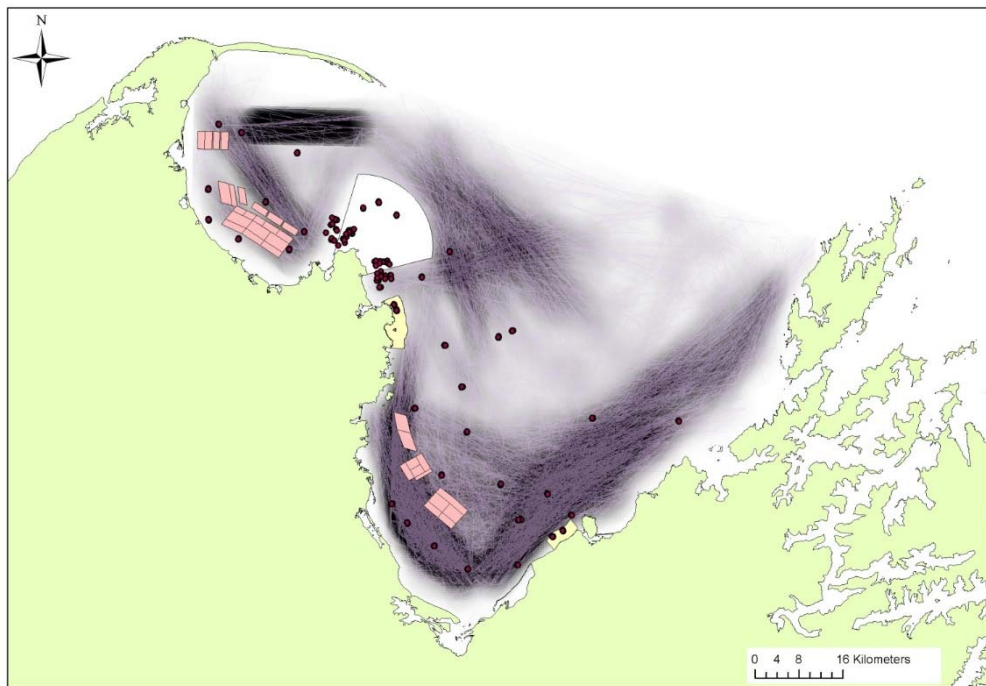


Figure 12: Estimated trawl paths (pink lines) atop line density interpolation (grey-black) derived from trawl start positions recorded on TCER and TCEPR reporting forms for the Tasman and Golden Bays study area. Pink polygons are aquaculture management areas. Circular symbols represent sampling locations.

Detailed estimates of the spatial extent of scallop and oyster dredge tows could not be obtained due to confidentiality of industry collected data and no legal requirement to file spatial data (above that at a sector level) for these fisheries. Instead, catch effort landing returns data were used to quantify scallop fishing effort (number of tows reported) by sector for the years preceding our sampling. Information was provided by the scallop industry on the areas within each sector that were fished each year, and on the basis of these data, relative levels of scallop fishing effort (tows.km⁻²) were calculated for each sector for each year. Sample stations within fished areas in each year were identified, and average effort estimated for each station in the three years prior to sampling (2007 – 2009 fishing seasons). There had been no scallop effort in Tasman Bay during this period, and so average effort for the previous three years (2004 – 2006 fishing seasons) was also calculated. Weighting more recent effort within the three year averages was examined, but resulted in very similar patterns, and was not pursued.

No appropriate spatially resolved oyster effort data could be obtained, as it is not reported spatially, and so we substituted oyster density as a proxy for oyster effort (research survey data from Brown (2012)). Oyster density data was krig interpolated (Figure 13).

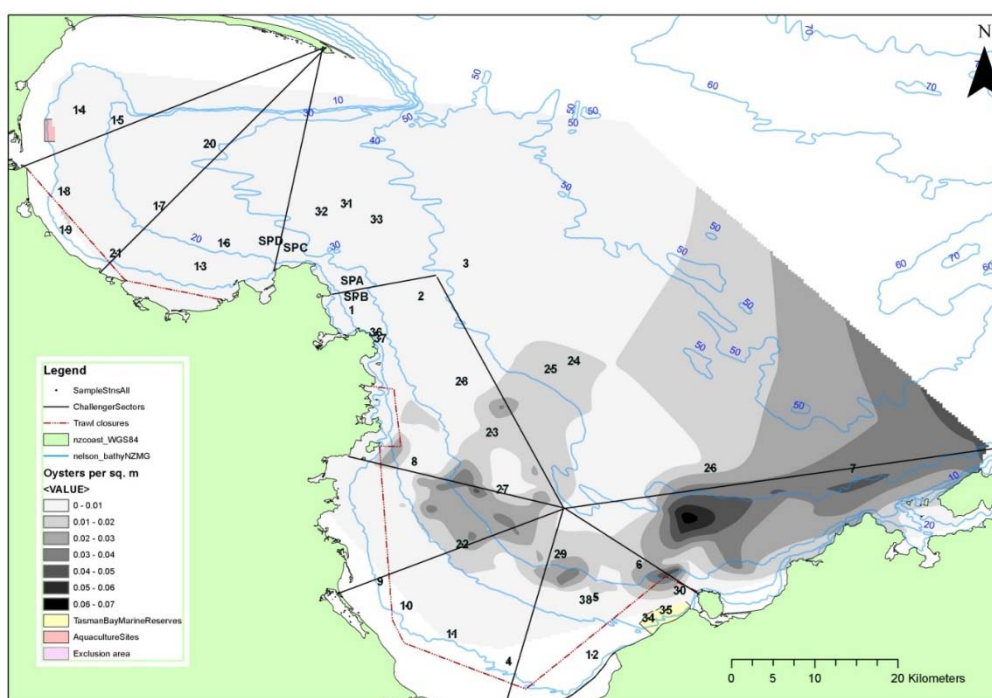


Figure 13: Krig interpolation of oyster density data from Brown et al. 2008 (grey shading) for the Tasman and Golden Bays study area. Values in blue indicate depth in metres and those in black indicate sampling locations. Black lines represent fishery sectors (used to stratify the survey), with the red dashed line representing the shallow extent of the survey.

Spatial correlates

In order to determine whether there was spatial differentiation across the study area likely to affect the analysis, spatial variables representing a polynomial surface were calculated, i.e., kilometres North, East and Northeast with 0,0 being in the southwest corner). Because the Northeast component could be confused with distance offshore in some locations this distance was explicitly calculated and also included in any spatial analysis. These spatial variables were not strongly correlated with any of the environmental variables, with the exception of maximum salinity and maximum wave heights which were correlated with East and North respectively (see Table 5).

Explanatory variables were categorised (Table 2) as:

- Expressions of fishing effort;
- Broad-scale environmental information derived from models;
- Smaller-scale environmental information collected during the survey;
- Spatial variables (correlates).

Table 2: Environmental variables available for use in the statistical analysis, and abbreviations used in later plots and tables for Tasman and Golden Bays case study.

Data	Metric	Abbreviation
Fishing effort	Average trawl effort previous 3 years	TEffort
	Average scallop effort previous 3 years	SEffort
	Average scallop effort previous 4 – 6 years	SE6yr
	Average oyster effort previous 3 years	OEffort
	Count of trawl / dredge marks from sidescan	Tmarks
	Categorical index of Tmarks	TMI
	Density of trawl / dredge marks from sidescan	xmarks
Broader scale environmental data derived from models	Salinity minimum	mnSal
	Salinity maximum	mxSal
	Salinity upper quartile	q1sal
	Salinity lower quartile	q3sal
	Salinity median	mdsal
	Salinity mean	xsal
	Maximum tidal current	mxCurrents
	Mean tidal current	xCurrents
	Average significant wave height	sigwave
	Maximum wave height	mxWaves
Site specific data collected during the survey	Sediment chlorophyll <i>a</i>	Chla
	Sediment organic content by loss on ignition	LOI
	Sediment dissolved organic matter	DOM
	Sediment % shell	Shell
	Sediment % sand	Sand
	Sediment % mud	Mud
	Sediment shear stress	Shear
	Sediment penetrability	Penetrometer
	Sediment % moisture	moisture
	Index of biogenic habitat reflectance from sidescan	xreflect
	Biogenic habitat % cover from sidescan	biogenic
	Index of biogenic habitat complexity from sidescan	Hcomplex
Spatial correlates	Northing	North
	Easting	East
	Distance in Northeast direction	NorthEast
	Distance offshore	Distance offshore

Pearson's R correlation analysis was used to identify strong (more than 0.8 or less than -0.8) correlations between variables, to help in selecting the most appropriate explanatory variables to include in the analysis.

Site allocation

Information used to locate sites for the Tasman and Golden Bays survey included depth, sediment type and fishing pressure. Depth was available as 10 m contours from LINZ hydrographic charts and sediment type from Mitchell (1987). Although Thrush et al (1998) also found estimates of chlorophyll *a* to be a useful predictor for benthic communities in the Hauraki Gulf, this information was not available. Estimates of current speed and wave height were not available at the time of site selection, but were included in subsequent analysis. Estimates of fishing pressure were obtained from two sources: (1) reporting of scallop effort (at the scale of the statistical reporting areas; 125 – 160 km²) in recent years, and (2) start points for commercial trawling from October 2007 (when record keeping on the TCE forms started) to 2009. The three closed areas; Separation Point, Tonga Island Marine Reserve, and Horoirangi Marine Reserve were assumed to have had no recent fishing effort, although all have been fished previously, and are likely to be in different states of recovery.

Depth through the area of interest ranged from 10 – 45 m and the area was divided into three depth bands (10 – 19.9 m, 20 – 29.9 m, 30 – 44.9 m) to stratify the sampling. Sediment throughout the depth range was predominantly mud. A few clay patches were present and frequently sand was present as a muddy sand or sandy mud mix. However, the clay patches were too confined and we did not have sufficient confidence in the differentiation between mud, muddy sand and a sandy mud to use sediment as a stratifier. Instead we decided to add the actual sediment type found at the sites as a covariable and not include it explicitly in the design. Four strata were used for each of the fishing pressure measures of which “absent” was one (Table 3).

Table 3: Fishing effort strata.

Strata	Number of trawl start points	scallop effort (number of tows)
Zero	0	0
Low	>0 – 50	>0 – 20
Medium	>50 – 200	>20 – 200
High	>200 – 800	>200

Polygons representing contiguous areas (sized over 1 km²) of depth and trawl intensity categories (e.g., depth 10 – 20 m, high trawl) were delineated, with the exception that areas of low or medium trawl start points lying between high density start points were excluded in case trawlers worked between these points. Polygons for sampling (at a site randomly located within the polygon) were then selected as follows. Polygons representing each combined depth and trawl start point density category were randomly selected within each of three spatial areas: Golden Bay and Separation Point, Tasman Bay northwest and Tasman Bay southeast, giving 22 polygons (as not all combinations occurred in each area). New polygons were then added to give 6 polygons in total for each of the scallop effort strata (total sites = 22 + 6 = 28). Nine extra polygons were then added to ensure that a good spatial spread across the area sampling of sediment plume from the Motueka River was sampled and matching sites within a marine reserve by at least one site nearby outside the reserve. At this stage we also added in polygons that were similar in strata allocation but had a different subdominant sediment type (total sites now 38). Finally 8 extra polygons were randomly selected to be sampled if extra time was available during field surveys (including 2 more sites in marine reserves). Table 4 summarises the initial 42 sites (the 38 that ended up being sampled plus the previously sampled Separation Point sites) and their strata information. The previously sampled sites (SPA – SPD; Figure 13 and Table 4) provided data on general fauna in the area, but were excluded from the analysis owing to concerns that they may reflect a completely different habitat to other sampled areas. The fishing effort levels at sampling stations are shown relative to the distributions of fishing effort (or proxy) for trawl and oyster fishing by 10 m depth band in Figure 15.

After a review process, it was decided to remove the Separation Point sites from the effects of fishing analyses as it was considered that the Separation Point marine reserve area may have been inherently different from the rest of the surveyed area before fishing pressure began. As part of this review process the fishing effort statistics were recalculated. Moreover, sediment characteristics were collected as part of the survey. The locations of the strata and 38 sampled stations are presented in Figure 14.

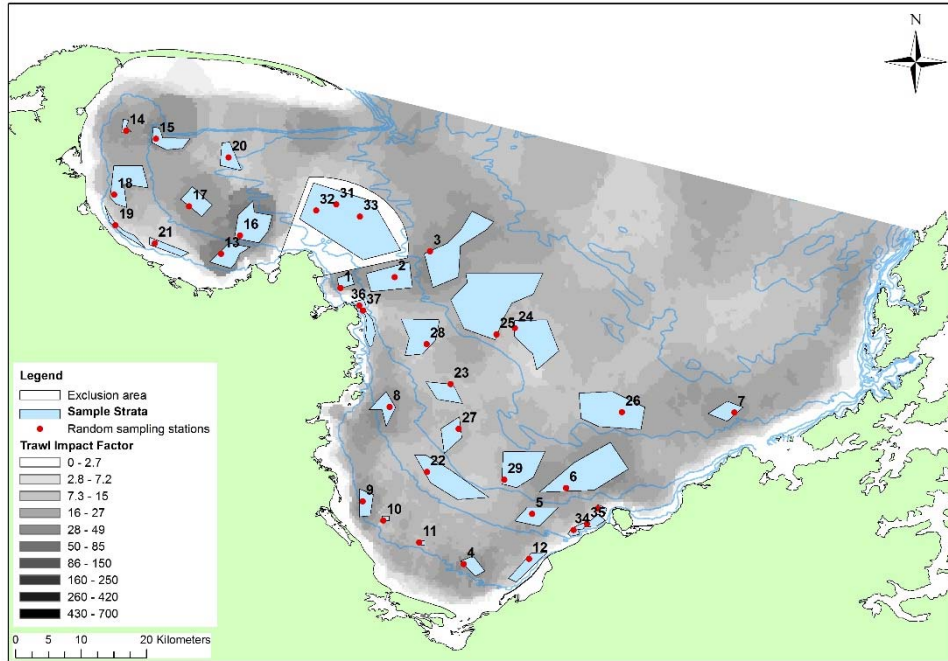


Figure 14: Sampling plan for the Tasman and Golden Bays study area: sites (numbered red circles) were allocated randomly within polygons representing homogeneous seabed areas (blue polygons) within each of three depth strata (blue isobaths), fishing pressure strata (grey shading interpolated from trawl start positions), and dominant sediment types (not shown).

Table 4: The 42 sites with their strata information. Although not used in the analysis, sediment type is provided to demonstrate that both mud (m) and sand/mud (s) types were sampled. Sites are ordered by spatial area, sediment type and fishing intensity. Site locations are presented in Figure 13 and Figure 14. Sites 1 – 38 sampled within current project. Sites SA – SD sampled in previous investigation, and excluded from this analysis. Depth bands are defined as 10–20 m (shallow), 20–30 m (med) and 30–45 m (high), trawl and scallop effort as categorical levels of variables according to Table 3, and spatial areas as Golden Bay and Separation Point (GB/SP), Tasman Bay northwest (TBNW) and Tasman Bay southeast (TBSE).

Site	Depth	Trawl effort	Scallop effort	Spatial area	Sediment type
1	shallow	med	low	TBNW	m
2	deep	med	low	TBNW	m
3	high	high	low	TBNW	m
4	shallow	high	med	TBSE	s
5	med	med	med	TBSE	m
6	med	high	med	TBSE	m
7	deep	med	zero	TBSE	m
8	med	high	low	TBNW	m
9	shallow	high	med	TBSE	m
10	shallow	low	med	TBSE	m
11	shallow	low	med	TBSE	s
12	shallow	low	med	TBSE	m
13	shallow	high	high	GB/SP	s
14	shallow	high	med	GB/SP	m
15	shallow	high	med	GB/SP	s
16	shallow	high	high	GB/SP	m
17	med	med	high	GB/SP	m
18	shallow	med	high	GB/SP	s
19	shallow	low	high	GB/SP	s
20	deep	low	high	GB/SP	s
21	shallow	low	high	GB/SP	s
22	med	low	zero	TBSE	m
23	deep	low	low	TBNW	s
24	deep	low	low	TBNW	s
25	deep	med	low	TBNW	s
26	deep	med	low	TBSE	s
27	deep	low	low	TBSE	s
28	deep	low	low	TBNW	s
29	med	med	med	TBSE	m
30	med	low	med	TBSE	m
31	deep	zero	zero	GB/SP	m
32	deep	zero	zero	GB/SP	m
33	deep	zero	zero	TBNW	m
34	shallow	zero	zero	TBSE	m
35	shallow	zero	zero	TBSE	m
36	med	zero	zero	TBNW	m
37	med	zero	zero	TBNW	m
38	med	med	med	TBSE	m
SPA	med	zero	zero	TBNW	m
SPB	med	high	high	TBNW	m
SPC	med	low	high	GB/SP	m
SPD	med	med	low	GB/SP	m

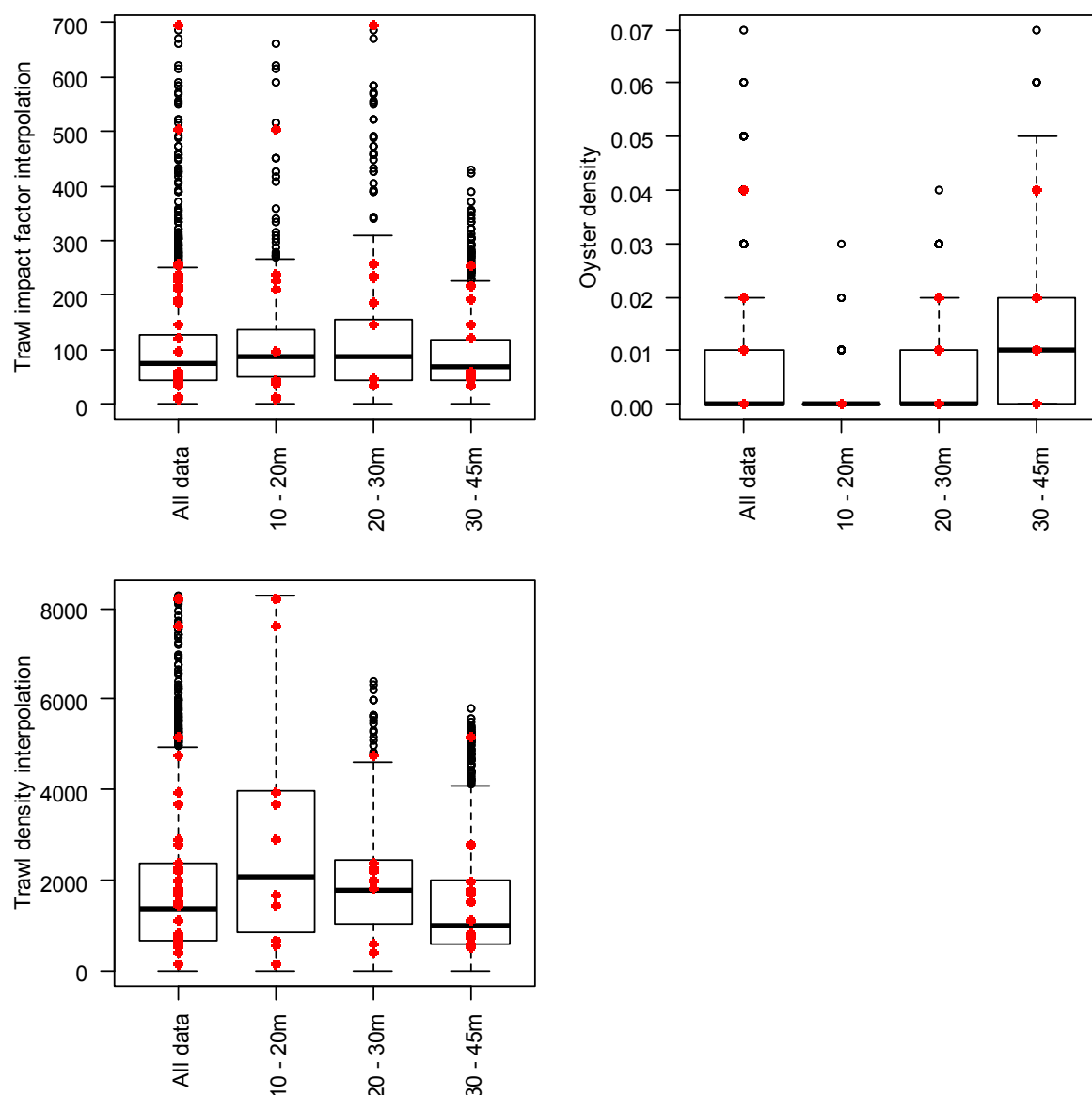


Figure 15: Boxplots showing distribution of fishing effort data extracted by grid cell for the whole Tasman and Golden Bays study area (All data) and individual depth bands for the measures of trawl fishing effort, and the proxy for oyster effort (oyster density). Red symbols overlaid represent the effort levels at each of the sampling locations within each depth band.

Sampling

Sidescan sonar transects

Trawling and dredging have previously been shown to disturb the seafloor, creating furrows or scars from dredges, trawl doors, ropes, bobbins and chains as they are dragged across the seafloor (e.g., Collie et al. 2000; Kaiser et al. 2006). These physical changes are visible to imaging systems such as sidescan sonar (Humborstad et al. 2004; Malik & Mayer 2007). At each site, we ran two replicated 60 m wide, 200 m long sidescan swaths using a high-frequency (675 kHz) Tritech towfish connected to a Garmin 72 GPS receiver. Vessel speed was maintained between 1 and 2 knots. The recorded files were then subsequently analysed using Seanet Pro V1.1.6 software (Tritech International Ltd., UK.). For every 30 seconds of sidescan track, we recorded: the number of trawl/dredge marks, estimated percent cover of biogenic habitat, and also two indices: biogenic reflectance, and habitat height/complexity judged by the degree of shadow cast on the tracks. Three-dimensional material that extends above the surface of the seabed, casting shadow recorded by the sidescan, was assumed to be biogenic in nature. Similarly,

highly reflective material (later observed to be mostly mollusc shells on video) was recorded as biogenic reflectance. The soft sediments of Tasman and Golden Bays are typically featureless, but in unfished habitats like inside the Separation Point Exclusion zone and inside Tonga Island marine reserve (Figure 9), bivalve shells and bryozoans form biogenic habitats that are associated with high biodiversity (Bradstock & Gordon 1983; Grange et al. 2003; Hewitt et al. 2005b). The variables derived were (i) count of trawl / dredge marks, (ii) estimated percentage cover of biogenic habitat, (iii) an index of trawl / dredge marks (0 = none, 1 = old, 2 = new; highest value taken per 30 seconds), (iv) an index of biogenic reflectance (0 = none, 1 = low, 2 = high), and (v) an index of habitat height / complexity (0 = smooth, 1 = medium, 2 = coarse).

Epifaunal sampling

Two 200 m sled based video tows were used to ground-truth each of the sidescan tracks above, and to identify macrofauna, habitats, and diatom mats. The camera was an Outland Technology 3.6 mm × 600 TV lines mounted on a custom built sled with an altitude of 500 mm and 700 mm lens to seafloor distance, with a field of view 700 mm wide. For illumination, the sled was equipped with two 700 lux 18 watt halogen lights and two lasers mounted 500 mm apart to set the width that organisms were recorded from. Within each run, all visible epifauna were identified and counted.

In addition, a modified epi-benthic sled was used to sample large (over 10 mm) sized organisms living at the sediment surface along the same transects as above. From prior experience with this type of sled in soft sediments conditions like Tasman Bay, we expected them to quickly fill with mud, acting as a sub-surface dredge, rather than sampling epi-benthic species. We therefore modified the sled fitting it with wider skids and steel tines spaced 12 mm apart to pick up organisms living at the substratum interface (Figure 16). Organisms were identified and frozen for potential stable isotope analysis.

Biomass of visually sampled epifauna was estimated by raising average specimen weights from the epi-benthic sled to observed densities. Epifaunal community productivity was estimated by applying previously determined Productivity / Biomass ratios (Lundquist & Pinkerton 2008; Pinkerton et al. 2008) to biomass estimates. Both biomass and productivity were examined in terms of total quantity, and total of typical inshore fish prey, on the basis of Stevens et al. (2011).

Infaunal sampling

Within each 200 m × 120 m sidescan swathe, seven random grab samples were collected using a Van-veen grab (0.069 m²) (Figure 17) recording GPS positions of each grab. Only grabs greater than 70% full were retained and further processed. Infauna were sieved on a 0.5 mm sieve and fixed in ethanol, but first processed to 1 mm in the laboratory. Individual species abundance and biomass were recorded, with biomass used in a similar way to the epifaunal data to estimate infaunal productivity. The 0.5 to 1 mm size fractions were not included in the multivariate analysis of community composition in relation to fishing pressure, but were analysed within MPI project ZBD200925, examining functional traits. Before sieving the samples in the field, the following sub-samples and measurements were recorded:

- Stable_isotope: A 5 ml syringe was used to sample surficial sediments which were placed into labelled acid washed test tubes (these have been stored for future analysis).
- Chlorophyll-*a*: A sub sample of surficial sediment was taken to 1 cm using a 45 mm pottle and spatula.
- Sediment chemistry: A 5 cm diameter, 5 cm depth tube corer was used to sample sediments for analysis for organics content by loss on ignition.
- Sediment grain size: A 5 cm diameter, 5 cm depth tube corer was used to sample sediments for sediment grain-size analysis.
- Shear strength: A standard soil shear gauge was used to assess the shear strength of the top 3 cm of sediments.



Figure 16. Modified epi-benthic sled showing wider skids and steel tines designed to pick up large (over 10 mm) organisms living at the surface of the sediments.



Figure 17: Van-Veen grab full of sediment before samples had been retrieved.

Sediments analysis

Sediments were weighed wet, and then oven dried to determine moisture content. After drying, they were wet sieved through 2 mm and 1 mm sieves and the filtrate collected on a 63 μm sieve. The 1 mm and 2 mm fractions were then re-dried to constant weight and re-weighed to calculate % sand and shell gravel respectively.

3.1.2 Results

Explanatory variables

Boxplots of the explanatory variables were drawn to determine which were likely to need log transformations before analysis (Figure 18). The extreme non-normality of oyster, scallop and trawl effort as well shear stress, total number and average number of marks suggested that log-transformations would be useful. Natural log transformations were used for all but scallop and trawl effort as their ranges were less than 10; for scallop and trawl effort $\log_{10}$ transformations were used.

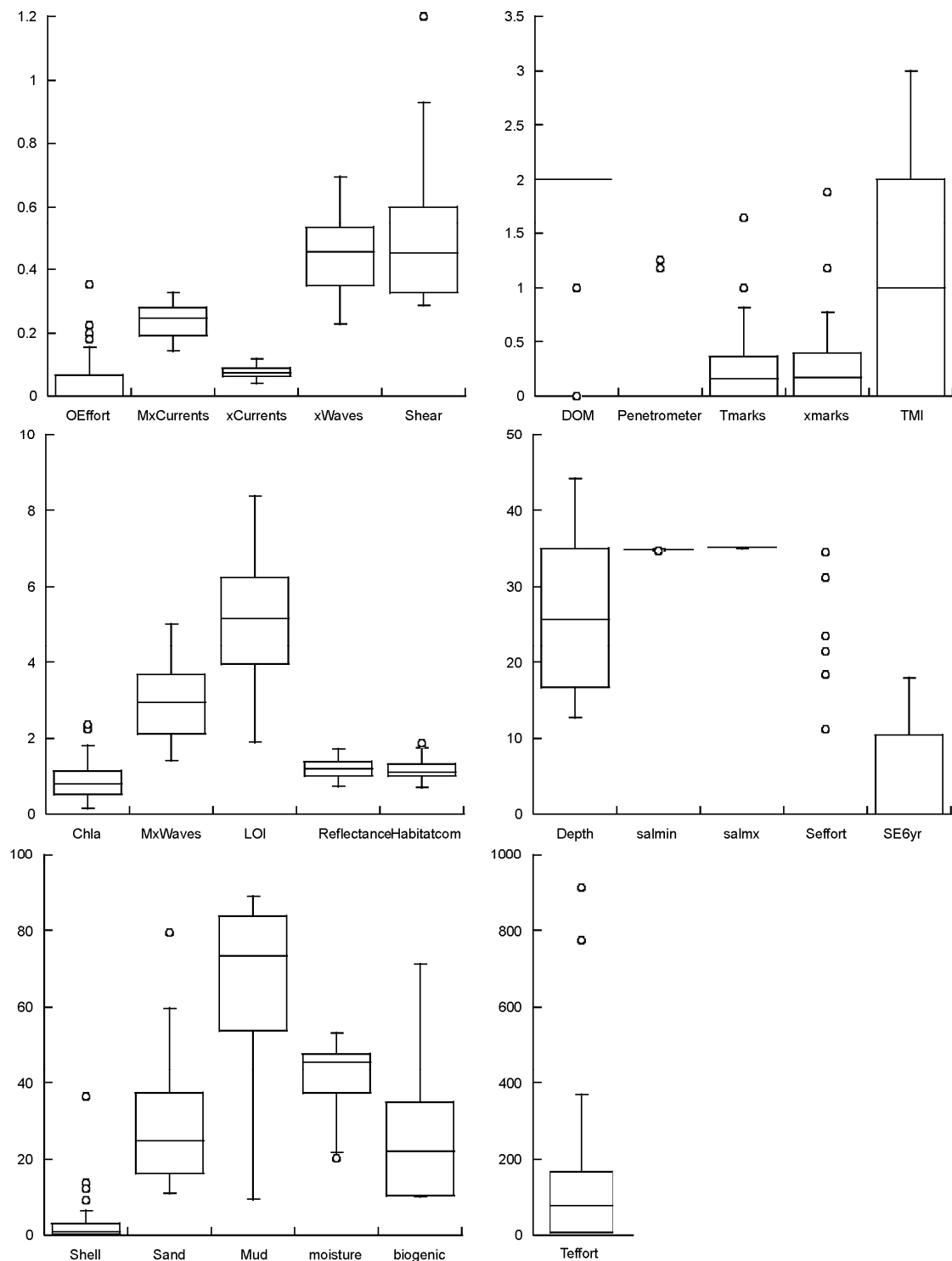


Figure 18: Boxplots of untransformed environmental variables grouped by maximum values. Because of their small range only two of the salinity variables are plotted. Mx - maximum, x - average, T, S or Oeffort - trawl, scallop or oyster effort averaged over previous 3 years, SE6yr - scallop effort averaged over previous 4–6 years, TMI - trawl mark index, LOI - % substrate mass loss on ignition (organic matter content). Variable name abbreviations provided in Table 2.

Pearson's R correlation analysis was conducted to determine correlations between dependent and independent variables (Table 5). In summary,

- None of the effort data were well correlated with each other or any other variables.
- The information gained from the video on dredge marks (TMI, xmarks and tmarks) were all well correlated with each other)
- The information gained from the video and sidescan sonar on epifaunal habitat structure (reflectance, biogenic and habitat complexity) were all well correlated
- Salinity variables mainly were correlated with each other, although median, mean and upper quartile salinity were also correlated with average significant wave height and depth
- Average significant wave height was correlated with depth
- Average current speed was correlated with maximum current speed and average significant wave height
- LOI was correlated with % sediment moisture
- Correlations between the spatial variables and maximum salinity and maximum significant wave height reflect the patterns of exposure to oceanic conditions in the region.

Where sets of variables were available (e.g., median, minimum, mean, interquartiles, maximum and minimum salinity), we decided to use in the statistical analysis variables with the fewest strong correlations: maximum, lower quartile and minimum salinity, depth, total mark counts, maximum wave height, maximum currents, chlorophyll *a* and mud content. Representations of sediment organic content (LOI and DOM), shell content and epifaunal habitat structure were not included as these are variables likely to be influenced by the disturbance of the sea floor created by fishing effort, both in the short and long-term.

Table 5: Pearson's R correlations more than 0.8 observed between environmental variables. Variable name abbreviations provided in Table 2.

	mdsal	minsalsal	mxsal	q1sal	xsal	q3sal	mxCurrents	sigwave	mxWaves	Tmarks	xmarks	xreflect	Hcomplex	LOI
mxCurrents							0.85							
sigwave	0.93						0.85							
xmarks										0.99				
TMI										0.82	0.83			
Hcomplex												0.85		
moisture														0.85
biogenic												0.86	0.83	
q1sal		0.89												
xsal	0.95			0.93				0.86						
q3sal	0.95				0.91			0.91						
Depth	0.94				0.89	0.96		0.93						
East									0.92					
North			0.82											

Principal component analysis (PCA) of the environmental variables, after normalisation, was conducted to investigate how the sites were positioned relative to the variables and whether any of the environmental variables combined to produce strong gradients amongst the sites. The first axis represented 31% of the variability, with the second axis representing a further 20% (Figure 19). Altogether the first four axes accounted for 74% of the variability, suggesting that the environmental variables did not combine to produce strong gradients.

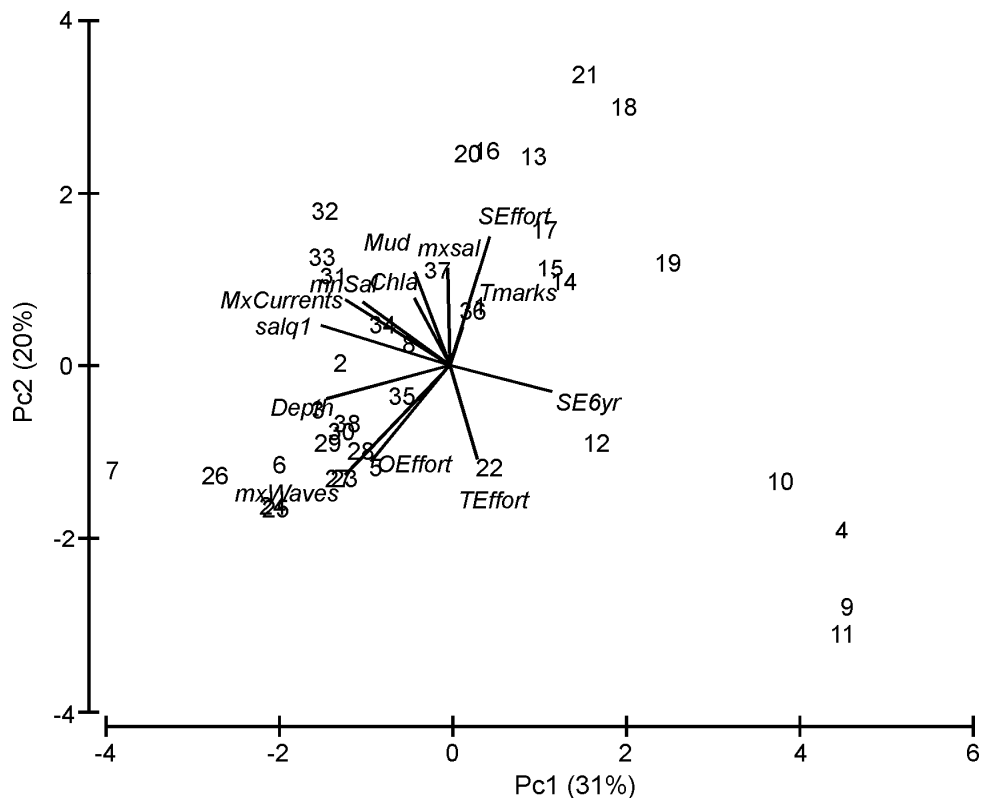


Figure 19: Tasman and Golden Bays study sites, and environmental drivers, along the first two axes of the principal component analysis (accounting for 51% of variation). Sites closest together are more environmentally similar. The length of the line representing the environmental variable indicates importance to differences between sites. Variable name abbreviations provided in Table 2. See Figure 14 and Table 4 for an explanation of site numbers.

Effects of fishing on the benthic community Epifaunal communities

An initial non-metric multidimensional scaling analysis of the epifaunal communities showed a good spread of sites across the ordination space (relative trawl and scallop effort shown) and a satisfactory representation in two dimensions (Figure 20). Sites with higher levels of fishing effort were generally located towards the lower part of the plot.

Backwards selection with DistLM removed only three of the environmental variables (chlorophyll *a*, total marks, scallop effort averaged over 4 – 6 years previous) leaving the other variables to explain 47% of the variability (Figure 21, Table 6). Depth and maximum salinity were the strongest drivers on the first two axes. The three fishing effort variables explained 18%, but the variables all drove the communities in different directions (Figure 22). Addition of variables representing spatial variability increased the total amount explained only to 50% and only kilometres North and NorthEast were selected as important.

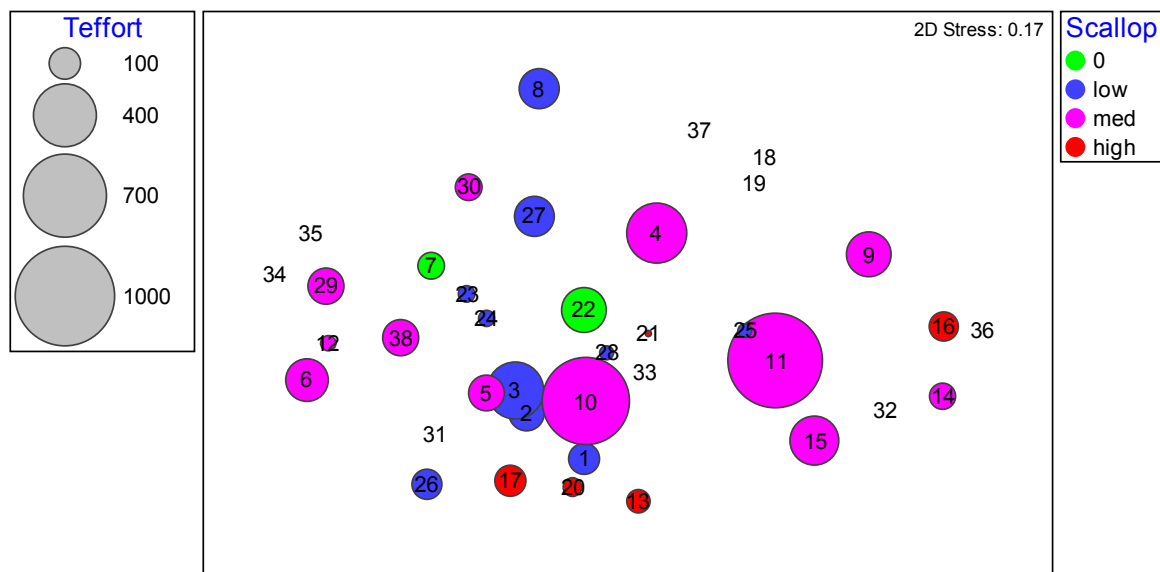


Figure 20: Non-metric multidimensional scaling analysis of the Tasman and Golden Bays epifaunal communities sampled from video data, indicating relative levels of scallop and trawl fishing effort. Numbers represent site labels. Sites that are closest together are most similar. Site labels represent sites documented in Table 4.

Table 6: DistLM results: Explanatory power (% of total community composition variance explained) of environmental variables for Tasman and Golden Bays epifaunal data based on fishing effort variables only, extra explanatory power for best model based on environmental variables and the best model including spatial variables. Terms marked * are removed from the best model as additional terms are made available, while grey highlighted variables become important as additional terms are made available, suggesting a range of interactions. Variable name abbreviations provided in Table 2. Marginal tests are provided in Appendix 1. Direction of correlation between fishing effort variables and univariate statistics indicated by arrows.

	Fishing effort	Environmental model	Plus spatial variables	Best model
Community	0.18 SEffort, OEffort, SE6yrs, Tmarks	0.29 mxWaves, Depth, Mud, mnSal, q1Sal, mxSal	0.03 North	0.50
#taxa	0.23 OEffort↓*, SEffort↑, SE6yrs↑*, Tmarks↑	0.23 mxWaves*, Mud, mxSal*, Chla, q1Sal	0.03 North	0.48
Shannon-Weiner	0.05 SEffort↑*, TEffort↑	0.22 mxCurrents, mxWaves*, Depth, mnSal,	0.02 North	0.29
Pielou's evenness	0.06 OEffort↑*, TEffort↓, SE6yrs↓,	0.38 Depth, Mud*, mnSal, mxCurrents, mxWaves*, Chla	0.01 North	0.44
Total biomass	0.06 OEffort↓*, Tmark↓s, SE6yrs↑	0.29 mxSal, Mud, mxWaves*, Depth	0.01 East	0.37
Fish prey biomass	0.06 OEffort↓*, Tmarks↓, SE6yrs↑	0.33 mxSal, mnSal*, q1Sal*, Depth, Mud, Chla, mxWaves	0.04 East	0.43
Productivity	0.07 OEffort↓*, Tmarks↓, SE6yrs↑	0.31 mxSal*, mnSal*, q1Sal*, Depth, Mud, Chla, mxWaves	0.01 East	0.39
Fish prey productivity	0.07 OEffort↓*, Tmarks↓, SE6yrs↑	0.31 mxSal*, mnSal*, q1Sal*, Depth, Mud, Chla, mxWaves, mxCurrents	0.04 East	0.42

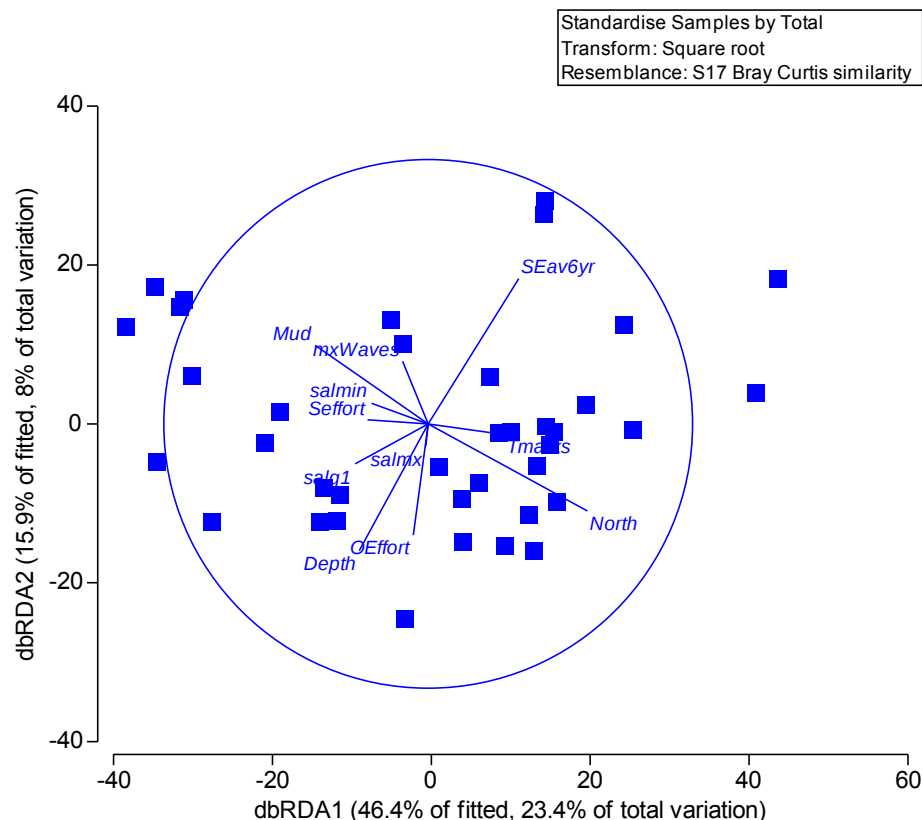


Figure 21: Distance-based redundancy analysis showing the position of sites and effort and environmental drivers selected as best at explaining between site dissimilarities in Tasman and Golden Bays epifaunal communities in two dimensions. Variable name abbreviations provided in Table 2.

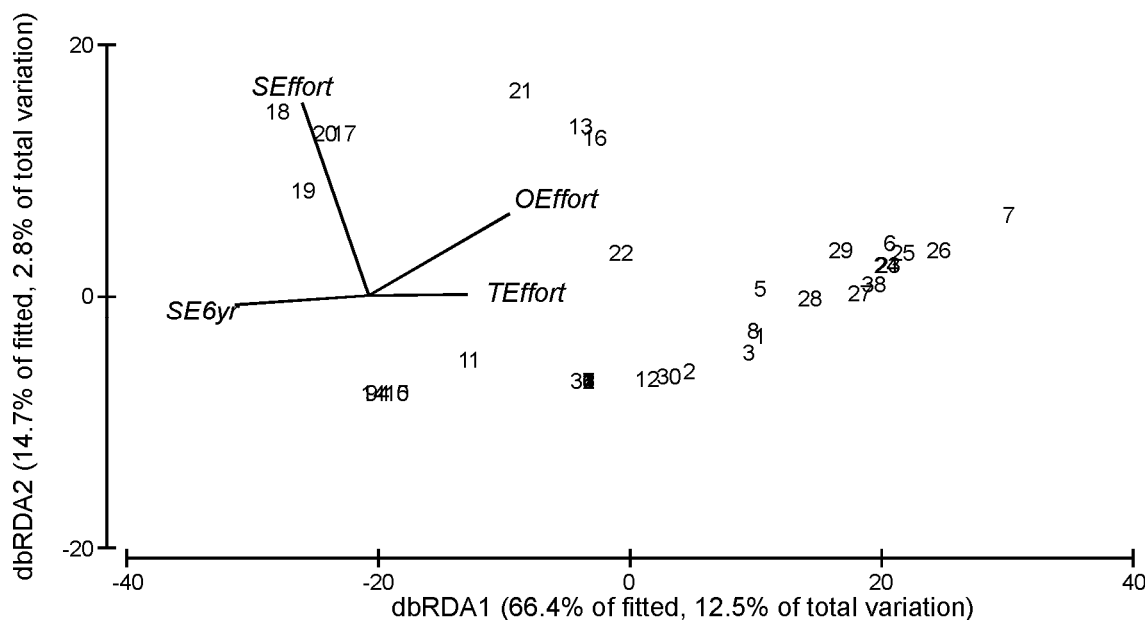


Figure 22: Distance-based redundancy analysis of the Tasman and Golden Bays epifaunal communities showing the position of sites and the four fishing effort variables in two dimensions. Variable name abbreviations provided in Table 2.

Examining the effects of the fishing on individual epifaunal species (using CAP with effort conditioned on other retained variables) identified a number of species that were negatively correlated with the fishing effort terms. *Atrina zelandica*, sponges and colonial ascidians were negatively correlated with trawl effort, the sea urchin *Pseudechinus albocinctus* and carpet star *Patiriella* sp. were negatively correlated with oyster effort, and unidentified hermit crabs and the gastropod *Maoricolpus* sp. were negatively correlated with scallop effort.

Epifaunal diversity, biomass and productivity

Oyster effort, recent scallop effort and trawl marks accounted for 23% of the variance in species richness when only fishing variables were offered to the model, but as environmental variables were introduced, oyster effort and recent scallop effort were dropped, while longer term scallop effort was retained. Species richness showed a negative relationship with oyster effort, but a positive relationship with the other effort measures. The final model also included terms for salinity, mud, chlorophyll and distance north, and accounted for 48% of the total variance (Table 6).

The Shannon-Weiner index was slightly less well explained by our variables (Table 6). The best model explained 29% and the retained variables in the final model included trawl effort, minimum salinity, depth, maximum currents and distance north. Scallop effort explained the most variance (of the fishing terms) in marginal tests, but was not retained in the final model. The Shannon-Weiner index showed a positive relationship with scallop and trawl effort. Pielou's evenness shows a positive correlation with oyster effort, but this term was only retained in the fishing effort only model. Both trawl and scallop effort (both negatively correlated with evenness) were retained in the final model for Pielou's evenness, along with minimum salinity, depth, chlorophyll, maximum currents and distance north, accounting for 44% of the total variance.

Both the total and fish prey epifaunal biomass and productivity showed similar patterns in relation to fishing and environmental drivers (Table 6). Each of the measures of biomass or productivity were negatively correlated with the measures of trawl and oyster fishing effort, but positively related to measures of scallop effort, with only oyster effort retained when just fishing effort variables are offered to the model, although this is dropped and other fishing variables are retained as more environmental variables are offered, suggesting that interactions are present. The minimum adequate models selected when all variables are offered account for between 37–43% of the variability with trawl marks and scallop effort consistently included in the final DistLM models. The effects of fishing pressure and environmental variables on biomass and productivity were also examined within a generalised linear modelling framework. Using a forward stepwise model selection approach, and retaining terms increasing AIC, fishing effort terms were retained for models examining total biomass (scallop effort and trawl marks) and fish prey biomass (scallop effort, trawl marks and oyster effort), but only environmental and spatial terms were retained for models examining estimated productivity. On the basis of the minimum adequate models generated (comparing model estimated biomass from the original data set with predictions with no fishing), across the sites studied, fishing has reduced the epifaunal biomass by 0–62% (median across sites studies of 21% decrease), and effects on epifaunal fish prey biomass range from a 13% increase to an 86% reduction (median of 10% decrease).

Infaunal communities

An initial non-metric multidimensional scaling analysis of the infaunal communities showed a good spread of sites across the ordination space and a satisfactory representation in two dimensions (Figure 23). Sites with higher levels of fishing effort were generally located towards the lower left part of the plot.

The model including only fishing variables accounted for 21% of the community variance. Inclusion of the abundance of erect structure forming epifauna (FG6) improved the model further, and environmental variables accounted for a further 22% of the variance. The final model included terms for trawl effort, oyster effort, recent scallop effort, FG6, salinity, mud, maximum waves, and distance north and east, and accounted for 53% of the total variance. (Figure 24, Table 7). Mud and distance east were the strongest drivers on the first two axes, although the abundance of erect structure forming

epifauna (FG6), scallop and trawl effort, and maximum wave height had more explanatory power than the remaining retained variables. The four fishing effort variables (retained in the final model) tended to drive the communities in different directions (Figure 25). Addition of variables representing space increased the total amount explained to 53%.

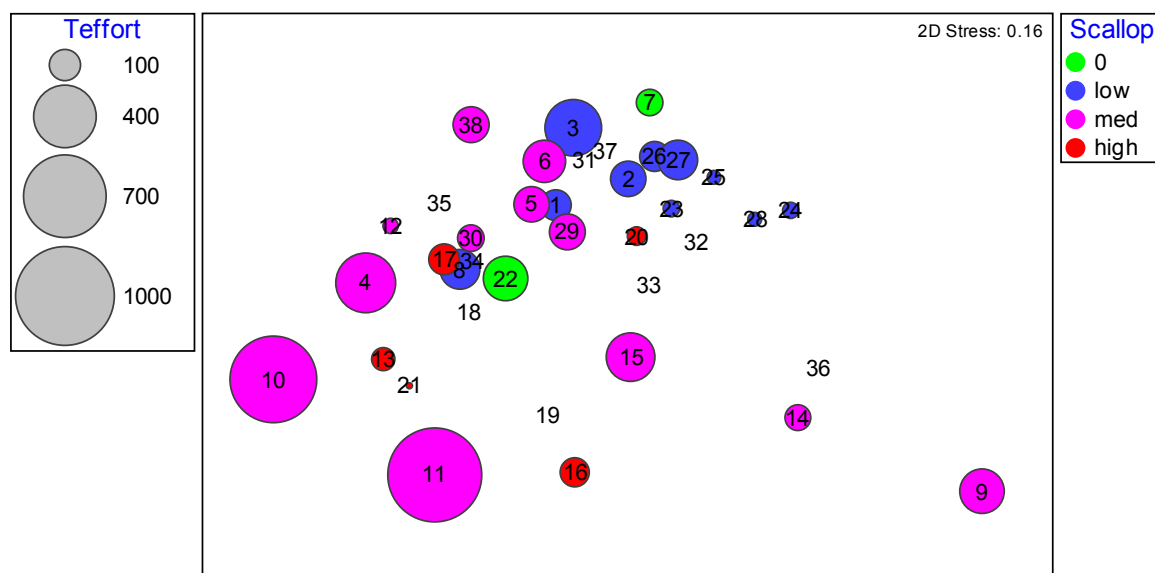


Figure 23: Non-metric multidimensional scaling analysis of the Tasman and Golden Bays infaunal communities. Sites that are closest together are most similar.

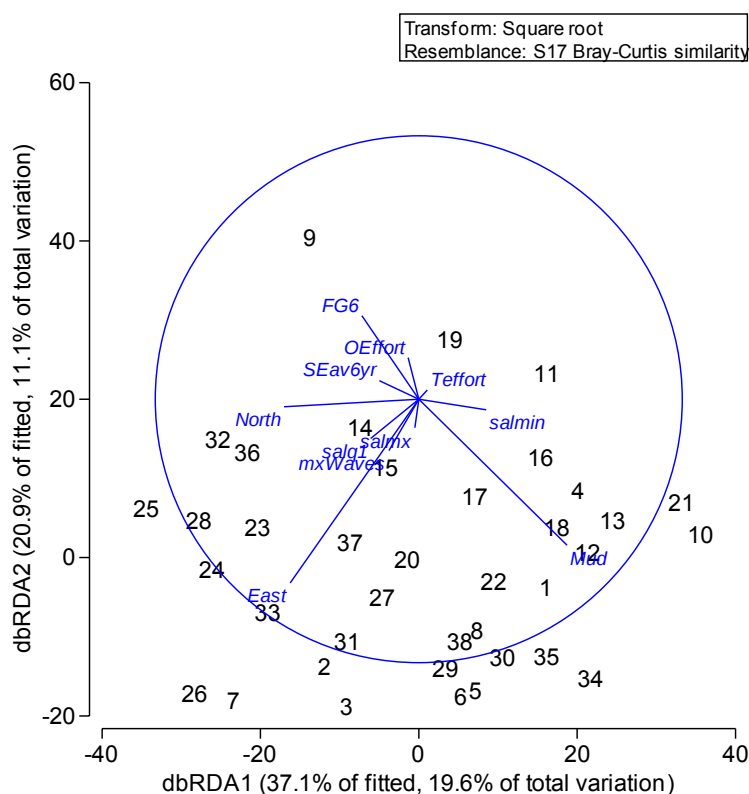


Figure 24: Distance-based redundancy analysis showing the position of sites and environmental drivers selected as best at explaining between site dissimilarities in the Tasman and Golden Bays infaunal communities in two dimensions. Variable name abbreviations provided in Table 2.

Table 7: Explanatory power of environmental variables for the Tasman and Golden Bays infaunal data based on fishing effort variables only, extra explanatory power for best model based on the abundance of erect structure forming epifauna (FG6), environmental variables and the best model including spatial variables. Terms marked * are removed from the best model as additional terms are made available, while grey highlighted variables become important as additional terms are made available, suggesting a range of interactions. Variable name abbreviations provided in Table 2. Marginal tests are provided in Appendix 1. Direction of correlation between fishing effort variables and univariate statistics indicated by arrows.

	Fishing effort	FG6	Environmental model	Plus spatial variables	Best model
Community	0.21 TEffort, SEffort, OEffort, SE6yrs*	0.04 FG6	0.22 mxWaves, Depth*, Mud, mnSal, mxSal, q1Sal	0.05 North, East	0.53
#taxa	0.43 TEffort↓, SEffort↓*, OEffort↑, SE6yrs↓	0.02 FG6	0.03 mxSal, q1Sal, mxCurrents	0.08 East	0.54
Shannon-Weiner	0.33 TEffort↓, SEffort↓, OEffort↓, SE6yrs↑, Tmarks↓	0.03 FG6	0.03 mxWaves*, q1Sal, Mud	0.05 East	0.43
Pielou's evenness	0.33 TEffort↑, SEffort↑, OEffort↓, SE6yrs↑, Tmarks↑	0.03 FG6	0.06 mxSal*, q1Sal*, mnSal	0.01 North	0.41
Total biomass	0.42 OEffort↓*, SE6yrs↑*, Tmarks↑, TEffort↑	0.03 FG6	0.27 mxSal*, q1Sal, Depth, Chla, Mud	0.01 North	0.73
Fish prey biomass	0.40 OEffort↓*, SE6yrs↑*, Tmarks↑, TEffort↑	0.02 FG6*	0.30 q1Sal, Depth, Mud, Chla, mxSal, mxWaves, mnSal, mxcurr	0.06 North, East	0.78
Productivity	0.46 TEffort↑, OEffort↓*, SE6yrs↑*, Tmarks↑	0.01 FG6	0.25 mxSal, Depth, Mud, Chla	0	0.72
Fish prey productivity	0.43 TEffort↑, OEffort↓*, SE6yrs↑, Tmarks↑	0	0.32 mnSal, mxSal, q1Sal, Depth, Chla, Mud, mxCurrents	0.02 North, East	0.79

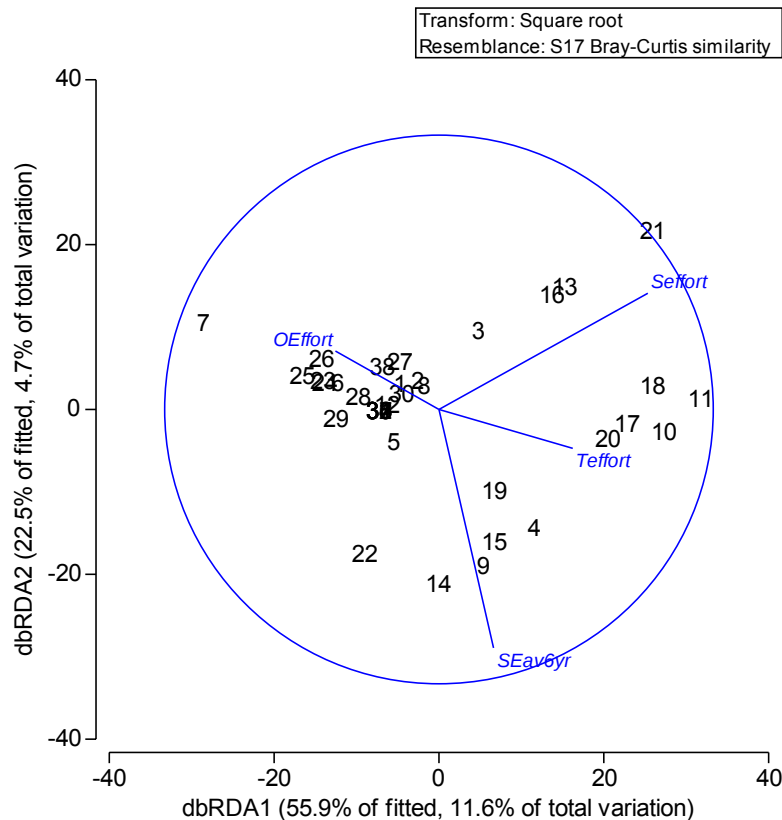


Figure 25: Distance-based redundancy analysis of the Tasman and Golden Bays infaunal communities showing the position of sites and the four fishing effort variables in two dimensions. Variable name abbreviations provided in Table 2.

Examining the effects of the fishing on individual infaunal species (using CAP with effort conditioned on other retained variables) identified a number of species that were negatively correlated with the fishing effort terms, although correlations were generally not as strong as those observed for epifaunal species. The bivalve *Corbula zelandica*, and the tube building polychaetes Serpulidae, Trichobranchidae and Maldanidae were negatively correlated with trawl effort, the bivalves *Ennucula strangei* and *Theodora lubrica* were negatively correlated with oyster effort, and the gastropod *Nozeba emarginata*, Maldanidae and generic hermit crabs were negatively correlated with scallop effort.

Infaunal diversity, biomass and productivity

Number of taxa was best explained (54%) by a number of variables, including three fishing variables (trawl effort, oyster effort and longer term scallop effort), and showed a negative relationship with trawl and scallop effort, but a positive relationship with oyster effort (Table 7). The abundance of erect structure forming epifauna (FG6), the first quartile and maximum salinity, and maximum currents were also important variables. The three retained fishing variables in the final model explained 34% of the variability. Inclusion of spatial variables into the explanatory dataset increased the variance explained to 54% and resulted in additional environmental variables becoming significant suggesting some interactions between those variables and the spatial variables occurred.

The Shannon-Weiner diversity index was slightly less well-explained by our variables (Table 7). The best model explained 43% of the variability, and the important variables included Oyster, Trawl and Scallop effort, trawl marks, FG6, the first salinity quartile and % mud. The Shannon-Weiner diversity was negatively related to all effort variables except recent scallop effort, and fishing effort explained 33% of the variability. Including FG6, environmental and spatial variables increased the explanatory power, with the best model accounting for 43% of the variability. The best model explained 41% of the variability in Pielou's evenness; significant environmental variables included the fishing effort

variables, FG6 and salinity variables, although some of these were dropped from the model when spatial variables were offered, suggesting interactions. Evenness was positively related to trawl and scallop effort, but negatively related to oyster effort. The combined fishing effort variables explained 33% of the total variance.

Total and fish prey infaunal biomass and productivity show similar patterns in relation to fishing and environmental drivers (Table 7), but patterns were different to those observed for epifauna. Biomass and productivity measures showed a negative correlation with oyster effort, but positive correlations with the other effort terms. Oyster effort was retained within models when only fishing terms were offered, but was dropped as environmental terms became available, although other fishing terms were retained or introduced. The minimum adequate models selected when all variables are offered account for between 72–79% of the variability with trawl effort, scallop effort and trawl marks consistently retained in DistLM models. The abundance of erect structure forming epifauna (FG6) was retained in most final models, but only explained 1–3% of the variability. The effects of fishing pressure and environmental variables on biomass and productivity were also examined within a generalised linear modelling framework. Using a forward stepwise model selection approach, and retaining terms increasing AIC, fishing effort terms were retained in all models (trawl effort and trawl marks in models examining biomass, and these terms along with scallop effort for models examining productivity). On the basis of the minimum adequate models generated (comparing model estimated biomass from the original data set with predictions with no fishing), across the sites studied, fishing has increased the infaunal biomass (total and fish prey) by 0–180% (median across sites studies of 20% increase). The effects on infaunal productivity (total and fish prey) range from a 24% reduction to a 69% increase (medians of 9–12% increase).

Functional traits

The three different measures of fishing effort make it difficult to combine into a single analysis, but individually, all three bivariate scatter plots (Figure 26) of habitat-forming emergent epifauna (Functional group 6) abundance against effort show a declining ceiling factor effect, although a quantile regression of the 90th percentile only detected significant negative relationships for trawl ($p = 0.0375$) and oyster ($p = 0.0482$), with scallop effort being non-significant ($p > 0.2$). Patterns were also examined in relation to trawl fishing intensity (average number of times an area is fished per year), derived from Baird et al. (2015) (these effort data do not include the scallop or oyster fisheries, and so do not reflect the effort data used in the analysis for this region). Substrate destabilisers (Functional group 4) show a clear decline in abundance as fishing intensity increases (suggesting a 50% reduction in abundance in areas fished 2 – 3 times a year), while there was less evidence of an effect of fishing on emergent epifauna (Functional group 6) (Figure 27).

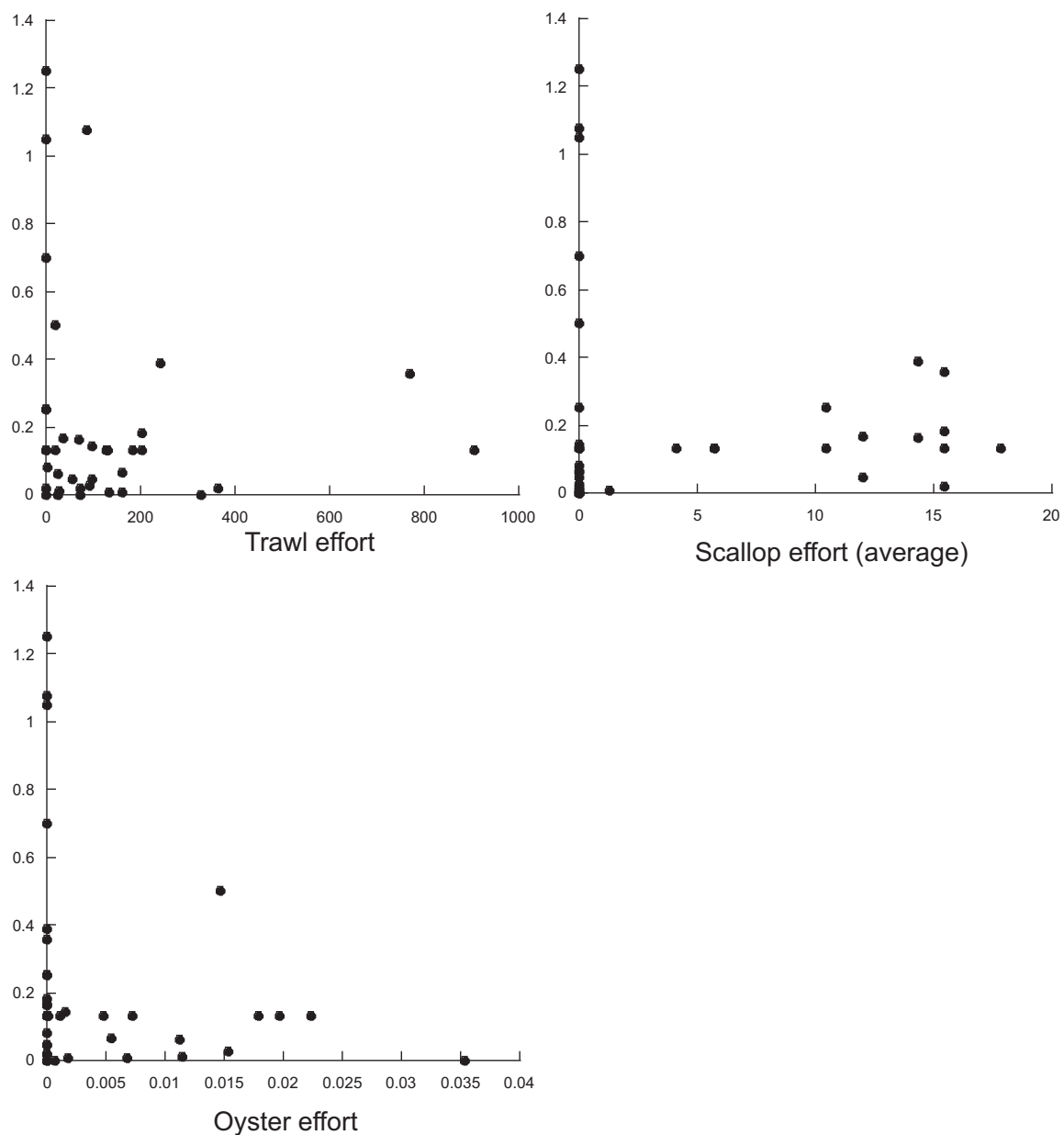


Figure 26: The abundance (number of individuals) of Tasman and Golden Bays epifauna comprising Functional group 6 (emergent epifauna) in relation to trawl, scallop and oyster fishing effort.

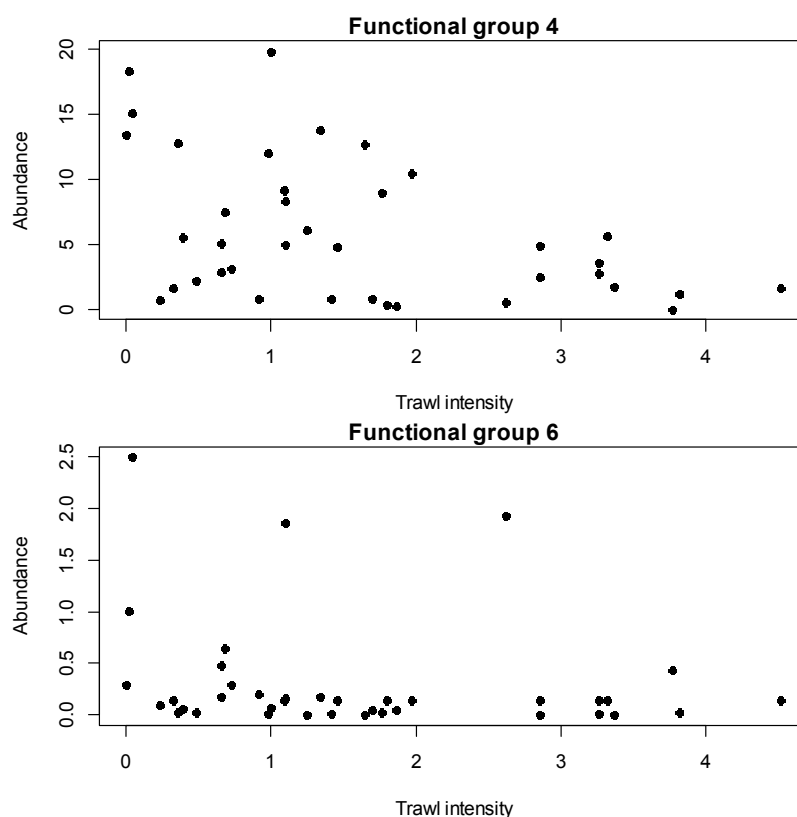


Figure 27: The abundance (numbers of individuals) of Tasman and Golden Bays epifauna (Functional groups 4 ‘Substrate destabilisers’ and 6 ‘Emergent epifauna’) in relation to trawl fishing intensity (average number of times fished per year; source Baird et al. (2015)).

3.1.3 Conclusions

The analyses of the epifaunal and infaunal communities were very similar in terms of variables that were important in explaining the observed variance, and the overall variance explained. Both fishing effort and environmental variables were identified as being important in explaining the patterns in the community data observed. The important environmental variables included maximum current speed, maximum wave height, depth, % mud, and salinity, and some spatial patterns were also identified within the study area. The abundance of erect structure forming epifaunal species was significant in accounting for variability in infaunal community structure, and a number of the univariate metrics examined, which is consistent with previous studies elsewhere (Hewitt et al. 2005b; Mormul et al. 2011; Buhl-Mortensen et al. 2012).

A range of species were negatively correlated with the different measures of fishing effort, including emergent or epifaunal species (*Atrina*, colonial ascidians, echinoderms) and infaunal organisms (bivalves, tube building polychaetes, hermit crabs and gastropods). *Atrina* species are considered to be particularly sensitive to mobile fishing gear (Hall-Spencer et al. 1999), and echinoderms are also likely to be very vulnerable to physical disturbance. The bivalve *Corbula*, tube building polychaetes, large gastropods and hermit crabs have also previously been identified as sensitive to fishing (Currie & Parry 1996; Tuck et al. 1998; Cryer et al. 2002). Predators and scavengers (like gastropods and hermit crabs) might be expected to be more abundant in fished areas, but previous studies have shown hermit crab response to benthic disturbance varies even between species with similar dietary characteristics (Ramsay et al. 1996), and that some gastropods are sensitive to physical disturbance from fishing, increasing vulnerability to predation (Ramsay & Kaiser 1998).

Examining the univariate measures, the species richness (number of taxa) for the epifaunal community had a greater percentage of the variance explained by fishing terms than did the infaunal community.

Epifaunal diversity and species richness were negatively related to trawl effort, but positively related to scallop effort. The infaunal community had more spatial structure which interacts with other variables. Infaunal species richness, diversity and evenness were generally all negatively related with fishing effort, and positively related with the abundance of erect structure forming epifauna. Variance in the Shannon-Weiner diversity measure was also less well explained by fishing variables for the infaunal community. Epifaunal and infaunal biomass and productivity appear to be affected by fishing, although the effects are not consistent. Across the study sites examined, models predict that fishing may have reduced epifaunal biomass by 21% (median), and epifaunal fish prey biomass by 10% (median), but increased infaunal biomass (total and fish prey) by 20% (median) and infaunal productivity by 9–12% (medians for total and fish prey productivity).

Overall, measures of recent scallop and trawl effort (averaged over three years prior to sampling) were consistently important in explaining the variance in both infaunal and epifaunal communities observed. The other fishing effort variables (trawl marks, average scallop effort 4–6 years prior to sampling, and oyster effort) were also important for some analyses. The full range of explanatory variables explained over 50% of the variance observed, with the fishing effort terms explaining between 15 – 22%. These results (in terms of total variance explained, and proportion attributable to the effects of fishing) are comparable with previous investigations into the effects of fishing on benthic communities in New Zealand (Thrush et al. 1998; Cryer et al. 2002; Tuck & Hewitt 2013). Functional trait effects were also detected, with a significant negative effect of fishing identified for long lived, sedentary, habitat-forming species, in relation to both trawl and oyster effort.

3.2 Opportunistic analysis of survey bycatch data

In order to compare the results obtained from the dedicated benthic community sampling described above with more opportunistically collected data, the invertebrate trawl bycatch community from recent West Coast South Island (WCSI) survey stations (MacGibbon & Stevenson 2013) in the Tasman and Golden Bays region (Figure 28) was examined in relation to some of the same environmental and fishing effort data described above (Table 2), in addition to other EEZ scale data layers collated by Baird et al. (2013). Although a survey was conducted in 2009 (kah0904), bycatch data were not recorded from stations within the study area relevant here, and bycatch data are limited to the 2007 and 2011 surveys.

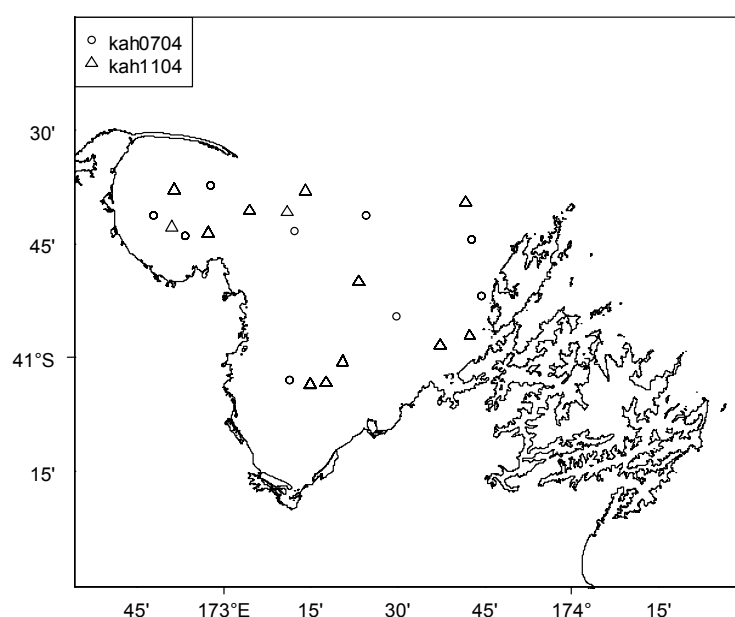


Figure 28: West Coast South Island trawl survey stations for which benthic invertebrate bycatch have been recorded in recent surveys, within the Tasman and Golden Bay area.

The benthic community data comprised 18 benthic invertebrate OTUs recorded from 22 stations in the Tasman and Golden Bays area, from the 2007 and 2011 surveys. An MDS plot of Bray-Curtis similarity of the transformed community data (Figure 29) shows reasonable overlap between surveys.

Table 8: Environmental variables available for use in the statistical analysis, and abbreviations used in later plots and tables for opportunistic analysis of Tasman and Golden Bays trawl survey bycatch.

Data	Metric	Abbreviation
Fishing effort	Average trawl effort previous 3 years	TEffort
	Average scallop effort previous 3 years	SEffort
	Average oyster effort previous 3 years	OEffort
Environmental data	Salinity minimum	mnSal
	Salinity maximum	mxSal
	Average significant wave height	sigwave
	Maximum wave height	mxWaves
	Minimum dissolved organic carbon	DOC
	Mean dissolved organic nitrogen	DON
	Summer primary production	PP_S
	Maximum tidal current	mxCurrents
	Sediment class (sand or mud)	sed_S, sed_M
	Summer SST	SST_S
	Seabed depth	depth
	Survey year (2007 or 2011)	y_2007, y_2011

The Bray Curtis similarity matrix of the square root transformed community data was analysed in relation to the environmental variables with DistLM, using backwards selection based on the AIC criterion (Figure 30). DistLM marginal tests for each variable retained in the final model are provided in Table 9, with maximum wave height explaining over 20% of the variance in the community composition individually, and maximum tidal current, summer sea surface temperature and depth each explaining over 15%.

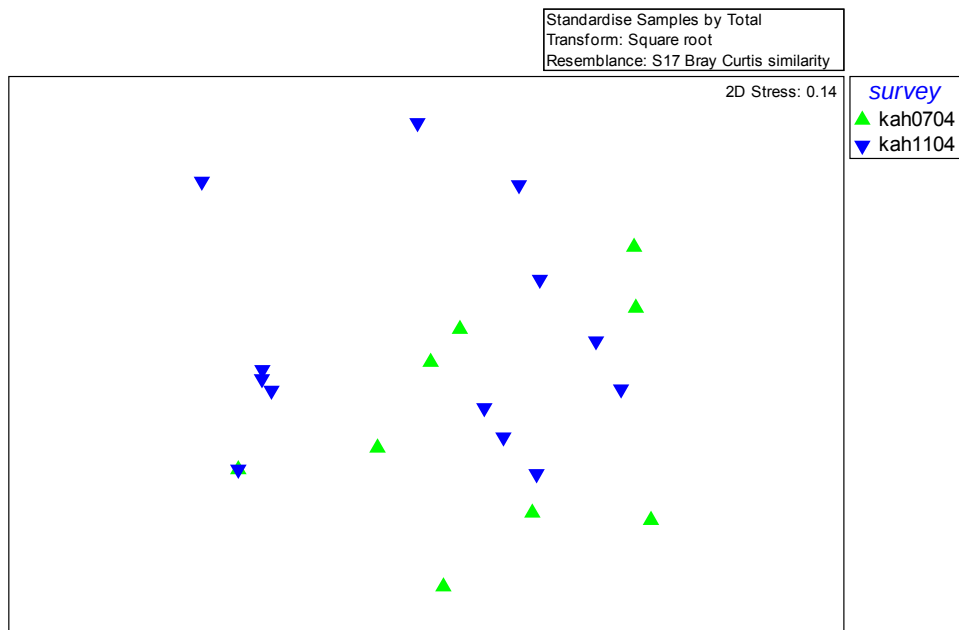


Figure 29: Non-metric multidimensional scaling plot of the Tasman and Golden Bays trawl bycatch community.

Table 9: DistLM marginal tests for analysis of the Tasman and Golden Bays trawl bycatch composition (only terms retained in the final model are presented). Prop represents the proportion of the variance explained by each term when fitted individually. Variable name abbreviations provided in Table 8.

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df
SEffort	7 629.5	3.366	0.015	0.144	20	2
DOC	1 101.7	0.425	0.813	0.021	20	2
mxCurrents	1 042.2	4.900	0.003	0.197	20	2
mxWave	12 854	6.409	0.001	0.243	20	2
SST_S	10 270	4.811	0.003	0.194	20	2
TEffort	4 881.6	2.030	0.086	0.092	20	2
depth	9 097	4.147	0.003	0.172	20	2
sed_	910.6	0.350	0.862	0.017	20	2
y_	5 942.7	2.528	0.052	0.112	20	2

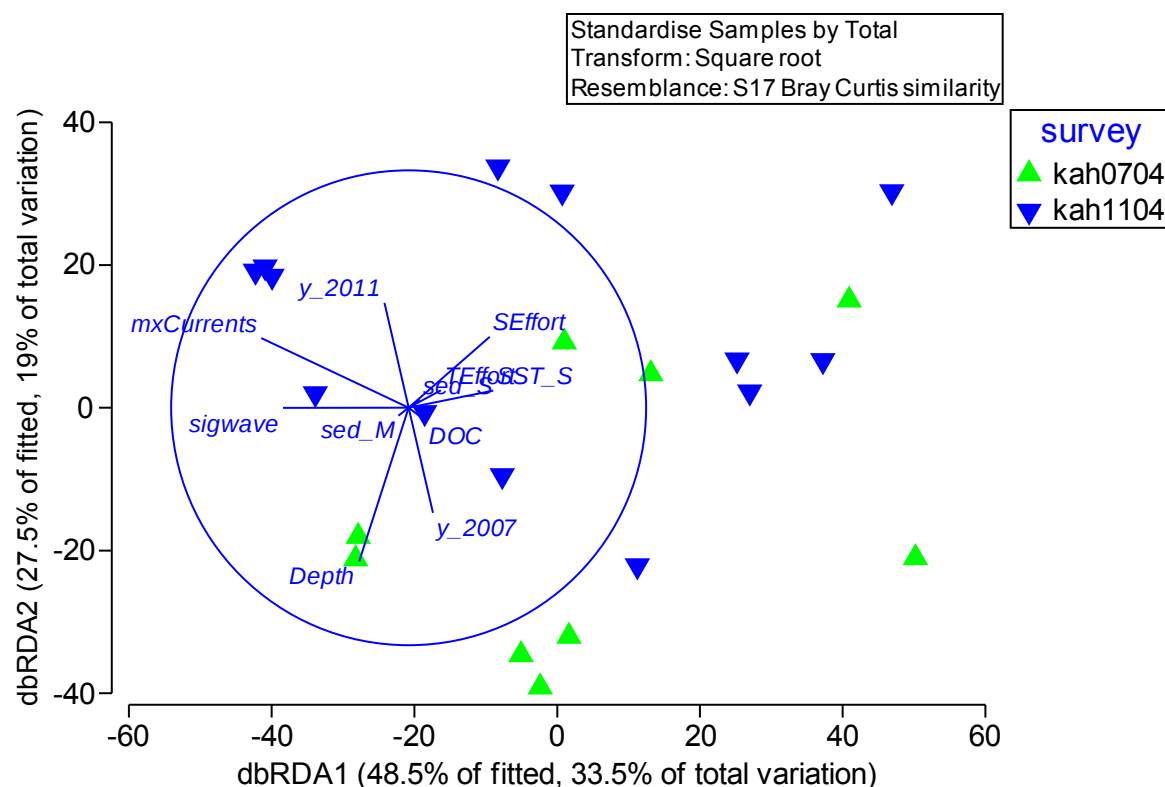


Figure 30: Distance-based redundancy analysis showing the position of sites and environmental drivers selected as best at explaining between site dissimilarities in the Tasman and Golden Bays trawl bycatch communities in two dimensions. Variable name abbreviations provided in Table 8.

The terms selected within the final model (Figure 30) accounted for 68.9% of the total variance in the community composition data. Marginal tests suggest that scallop effort explained 14% of the variability, while trawl effort explained 9%, although these levels include correlation with other terms (Table 9). Excluding single fishing effort terms while forcing inclusion of the remaining selected terms suggests that the scallop effort and trawl effort terms (taking account of the effect of the other explanatory variables) account for 4.7% and 5% of the total variance individually, and 9.7% of the total variance together.

Analysis with CAP (with backwards stepwise model selection using AIC, on the standardised, root transformed community data) retained terms for scallop and trawl effort, DOC, DON, maximum current speed, SST, primary production, sediment type and year, and accounted for 65.5% of the total variance (Figure 31). Examining the effects of the fishing effort variables conditioned on the other retained terms (partialling out their effects) found that the two fishing terms accounted for 10% of the total variance, and showed correlations with some of the species groups (Figure 32). While the sponge *Callispongia* sp. (CPG) shows a positive correlation with trawl effort, both ascidians (ASC) and the white finger bryozoan *Celleporina grandis* (CEG) show negative correlations. *Celleporina grandis* is also negatively correlated to scallop effort, as are the octopus *Pinnoctopus cordiformis* (OCT) and unidentified sponges (ONG). Limiting the benthic community to more sedentary species (excluding octopus) reduced the number of stations available for analysis, but resulted in a very similar final model (not shown), although the term for oyster effort was also retained, accounting for about 1.5% of the total variance.

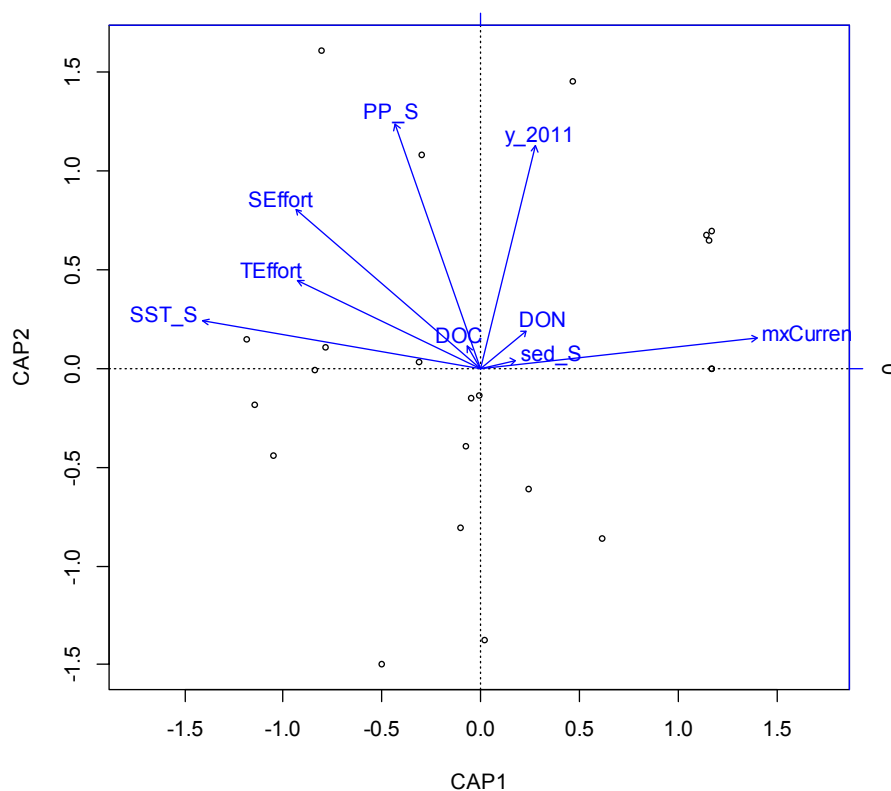


Figure 31: Ordination of stations generated by Constrained Analysis of Principal Coordinates of the Tasman and Golden Bays trawl bycatch data. Arrows represent directions and approximate relative magnitudes of effects of explanatory variables in the minimum adequate model. Variable name abbreviations provided in Table 8.

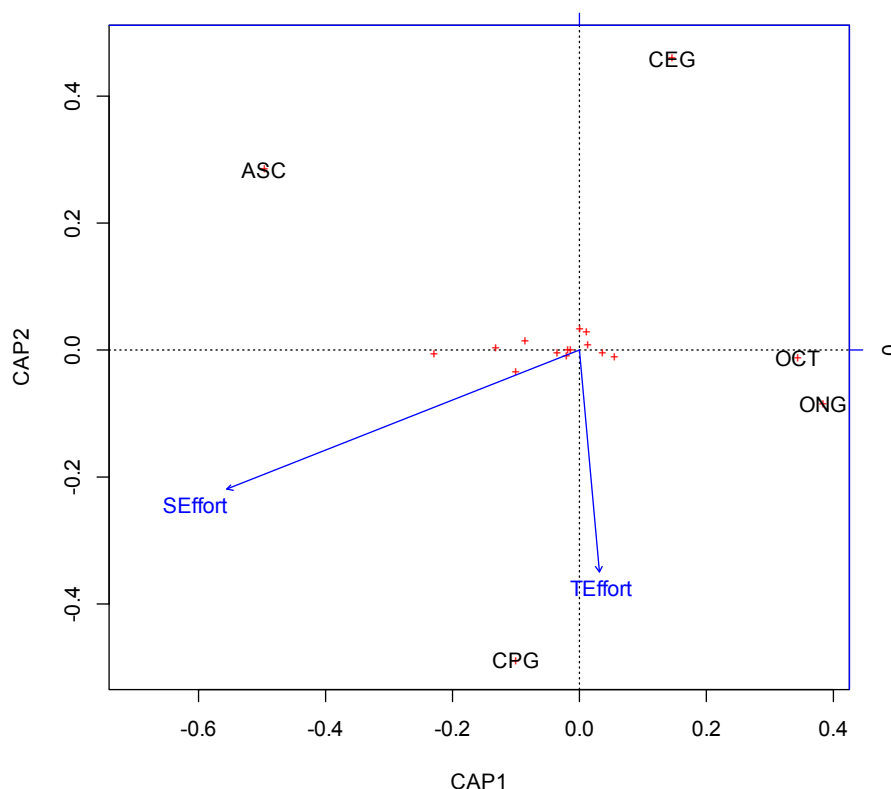


Figure 32: CAP correlation plot for effect of trawl and scallop fishing effort, after partialling out the effects of the other terms retained in the Tasman and Golden Bays trawl bycatch minimum adequate model (Figure 31). ASC – ascidians; CPG – *Callispongia* sp.; CEG - *Celleporina grandis*; OCT - *Pinnoctopus cordiformis*; ONG - unidentified sponges.

3.2.1 Conclusions

The analysis of the trawl survey benthic bycatch community identified significant effects of waves, currents, depth, temperature and sediment parameters, along with terms for both scallop dredge and trawl effort, accounting for almost 70% of the total variance, with fishing effort terms accounting for almost 10%. While the overall set of explanatory variables retained differed from the dedicated benthic community sampling study (Section 3.1; Table 6, Table 7), explanatory terms for tidal currents, waves, depth and sediment were consistently important across all three analyses (dedicated infauna, epifauna and bycatch). The combined fishing terms appeared less important in accounting for variance in community composition in the bycatch data (10%) than for the dedicated community sampling (15 – 19%), but were still retained as significant terms.

Examining individual species correlations with the fishing terms (conditioned on other retained terms) identified that while *Callispongia* sp. was positively correlated with trawl effort, ascidians, the white finger bryozoan *Celleporina grandis*, the octopus *Pinnoctopus cordiformis* and unidentified sponges were negatively correlated with either trawl effort, scallop effort or both. While sponges are generally considered to be negatively affected by fishing disturbance, *Callispongia* sp. is one of a small group that, while easily fragmented by physical disturbance, are able to survive, and colonies may actually be dispersed by such fragmentation (M. Kelly, pers. comm.).

3.3 Tasman and Golden Bays overall conclusions

The combined activity of inshore trawling and scallop and dredge fisheries in the study area generated strong gradients over which to examine the effects of fishing. The results from the dedicated sampling of infauna and epifauna and the opportunistic sampling of trawl bycatch were relatively consistent, both in terms of the environmental (tidal current speed, wave height, depth and sediment) and fishing (scallop and trawl fishing effort) drivers identified to be important in driving the community composition. The strength of fishing effect detected (percentage of variance explained) on the community composition was reasonably consistent between studies, but was greatest for the dedicated study of epifauna (19%), and weakest for the opportunistic study of epifauna (10%) and, where examined, the univariate diversity measures showed the same pattern. Fishing appears to have reduced epifaunal biomass, but increased infaunal biomass and productivity.

Emergent (e.g., horse mussels, bryozoans, some sponges) and fragile epifauna and infauna (e.g., echinoderms, tube building polychaetes) were identified as being negatively affected by fishing disturbance, and a number of these species or groups have been identified as being sensitive to fishing in previous studies. Effects were also examined at the functional group level, and both substrate destabilisers (surface dwelling, mobile, deposit feeding species) and emergent epifauna (long lived, sedentary, habitat forming species) were negatively correlated with fishing effort metrics, suggesting that the abundance within these groups may be reduced by up to 50% in areas fished 2 – 3 times per year (on average).

4. SOUTH CANTERBURY BIGHT

4.1 Dedicated benthic community sampling

4.1.1 Methods

Study area

The study was conducted in the South Canterbury Bight area, between Akaroa and Timaru, and ranged in depth from 20 m along the north western edge of the study area to 130 m further offshore to the south east (Figure 33). It is an important area for inshore commercial fisheries, with trawlers targeting a range of species including barracouta (*Thyrsites atun*), elephantfish (*Callorhinchus milii*), red gurnard (*Chelidonichthys kumu*), red cod, *Pseudophycis bachus*, tarakihi *Nemadactylus macropterus*, dark ghost shark (*Hydrolagus novaezelandiae*) and mixed flatfish species (Bentley et al. 2011). In addition, a range of other commercial species are caught as bycatch.

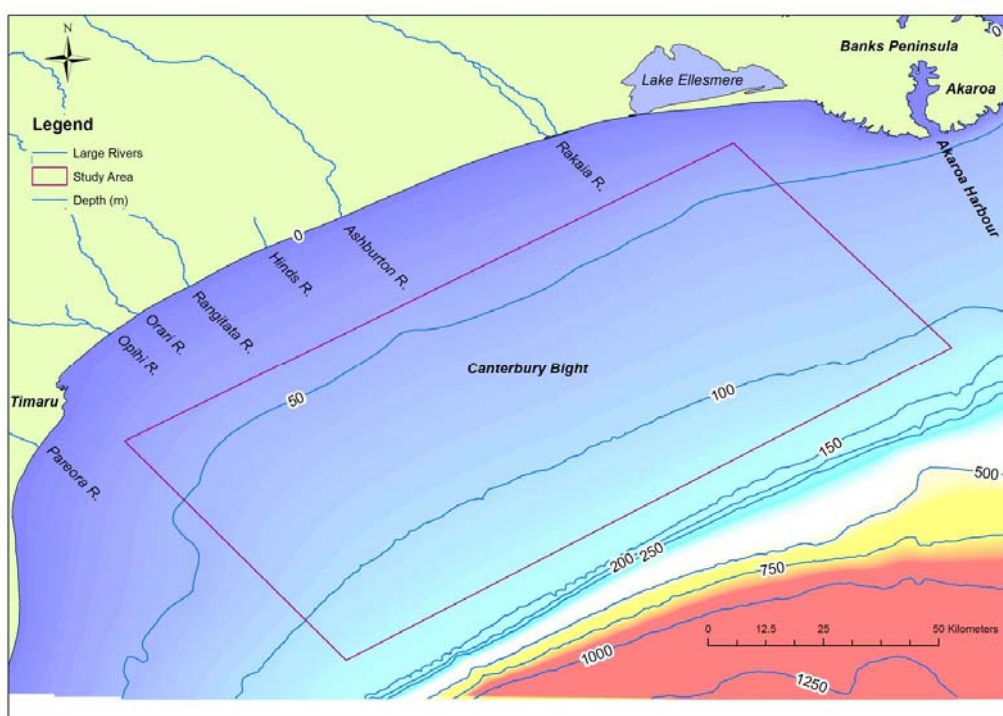


Figure 33: Location map of study area showing 50 m depth contours and major rivers flowing into the South Canterbury Bight. Central red box outline represents the study area.

Environmental and fishing data used

A range of environmental variable GIS layers (maximum M2 and maximum tidal current speed, mean sediment grain size, riverine suspended sediments, bottom water salinity, summer and winter bottom water temperature, mean and 99th percentile of modelled wave height, modelled seabed orbital wave velocity, and distance from earthquake epicentres) were collated for the study area from a number of previous studies and reports, or generated specifically for this study using existing models (Appendix 1). These variables were used in conjunction with seabed depth, to stratify the study area into areas of similar environmental conditions (see below).

Most of the variables do not show pronounced gradients across the study area. Tidal current speeds are slightly elevated around Banks Peninsula, and also along the steeper slope along the south east of the study area. Coarser sediments are present along the coastal fringe, but only extend into the margins of the study area between the Ashburton and Rakaia rivers. The finest sediments in the study area are found towards the southwest. Re-suspended coastal sediments were considered to be uniformly very low across the study area, but the coastal freshwater band does extend into the shallower parts of the study area. Bottom water temperature decreased with depth, and while temperatures are warmer in summer, relatively warmer water extends further offshore in winter. Wave height increases further offshore, but seabed orbital velocity decreases with depth, and is highest in the north west of the study area. The significant earthquakes of 2010 and 2011 had epicentres to the north of the study area, and so distance from the epicentres increases to the south. Further details of these datasets are provided in Appendix 1.

Fishing effort data

Fishing effort data was extracted from a database generated by Baird et al. (2015), which contained individual tow records for all trawl events conducted within 1 m of the seabed and reported on TCEPR or TCER forms between the 2007–08 and 2011–12 fishing years. Trawl events recorded on TCER forms only report start location (and not end location), and the procedures used to estimate finish positions and actual trawl tracks are documented by Baird et al. (2015). While there were limited options

available for dealing with these data, it is acknowledged that the estimation of trawl finish positions introduces additional uncertainty in fishing effort distribution maps compared to those based on recorded start and finish positions. The best estimates of finish positions have been used here, and the uncertainty has not been accounted for in the analysis. Fishing events are plotted individually (as a straight line between start and finish locations), and swept area summed on an annual basis across a 5 km × 5 km grid. Effort patterns were quite consistent between years, and the fishing effort data layer used in subsequent analysis was the sum of swept area by 5 km × 5 km grid cell over the period of fishing years 2007–08 to 2011–12 (Figure 34).

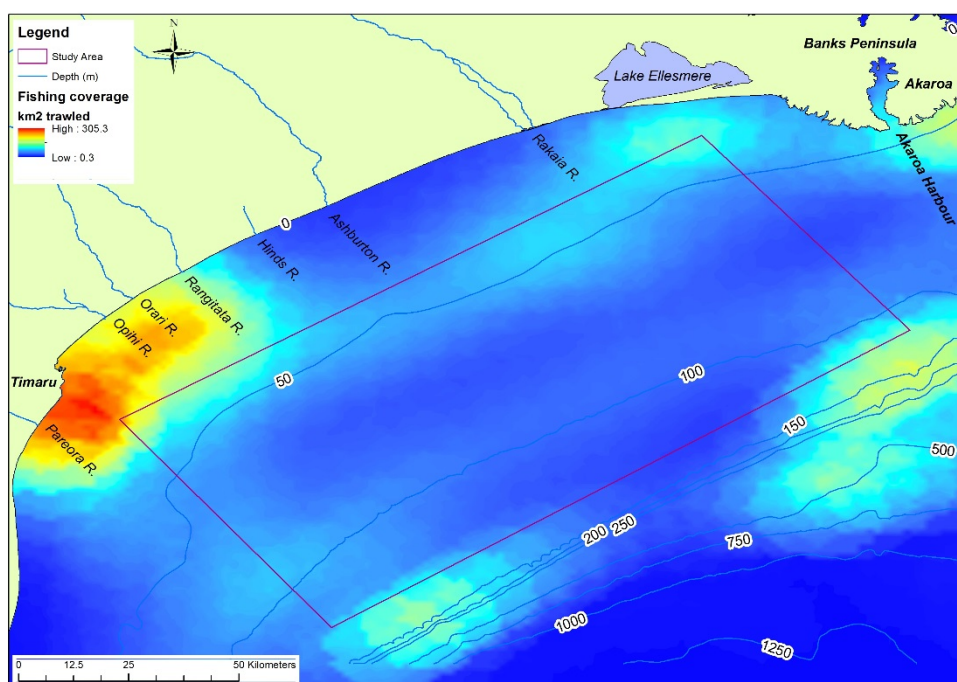


Figure 34: Map of aggregated swept area of trawl fishing (within 1 m of the seabed) per 5 km × 5 km cell, for the South Canterbury Bight study area over the period of fishing years 2007–08 to 2011–12.

Spatial variables

In order to determine whether there was spatial differentiation across the study area likely to affect the analysis, spatial variables representing a polynomial surface were calculated, i.e., kilometres North and East with 0,0 being in the southwest corner). The distance North was highly negatively correlated with Quake (the distance from the epicentre of large earthquakes) (Pearson's $r = -0.97$).

Explanatory variables were categorised (see Table 10) as:

- Expressions of fishing effort;
- Broad-scale environmental information derived from models;
- Smaller-scale environmental information collected during the survey; and
- Spatial variables.

Table 10: Environmental variables available for use in the statistical analysis, and abbreviations used in later plots and tables for the South Canterbury Bight case study.

Data	Metric	Abbreviation
Fishing effort	Average trawl effort previous 5 years	Fish
Broader scale	Bottom water salinity	FW
environmental data	Winter seabed temperature	WinT
derived from models	Summer seabed temperature	SumT
	M2 tidal current speed	M2
	Maximum tidal current speed	mxcurr
	Mean wave height	Wmean
	99 th percentile of wave height	W99
	Seabed orbital velocity	vel
	Median grain size	median
Site specific data	Sediment chlorophyll <i>a</i>	Chla
collected during the	Sediment organic content by loss on ignition	LOI
survey	Sediment % sand	Sand
	Sediment % mud	Mud
	Sediment % shell and gravel	Shell
	Sediment % moisture	Moisture
	Water depth	Depth
Spatial correlates	Northing	North
	Easting	East
	Distance in Northeast direction	NorthEast
	Distance from earthquake epicentres	Quake

Pearson's R correlation analysis was used to identify strong correlations between variables, to help in selecting the most appropriate explanatory variables to include in the analysis.

Site allocation

The gradient analysis approach herein involves testing the hypothesis that fishing effort makes no unique contribution to the overall R^2 found for the regression (note that this is equivalent to the null hypothesis that $\beta_{yF,a}=0$, i.e., the slope related to fishing effort = 0) (Tuck et al. 2011). To achieve a representative and even spread of samples across the study area at the Canterbury Bight, it was decided to use the environmental data (detailed in Appendix 1, along with depth: Figure 33) to partition the area using a non-hierarchical k-means cluster analysis and then to use the fishing effort to identify gradients of fishing effort within each cluster-class, to then assign random sampling sites (see below). Before the cluster analysis was run, the data-range within each variable was standardised (0–1 distribution), and the clusters were joined based on the average distance between all the members after converting the transformed data to a matrix based on the Euclidean distance using R (R version 2.15.2, <http://www.R-project.org/>).

To extract the data for the cluster analysis, a 1 km grid was overlaid onto each environmental variable and values extracted for each intersecting point (Spatial Analyst, extract data to points tool in ArcMap 10). To determine the optimal number of clusters a Sum of Squared Error (SSE) scree plot was generated (Figure 35). The location of the elbow in the resulting plot was not definitive, and following discussion within the MPI Aquatic Environment Working Group, the choice was made to determine six clusters.

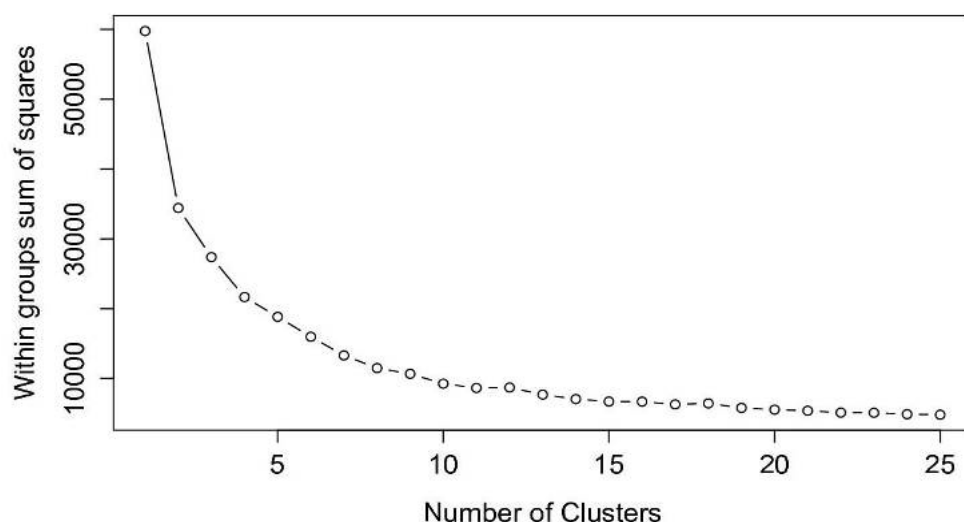


Figure 35: Sum of Square Error (SSE) scree plot for clustering of the South Canterbury Bight environmental data to determine the number of clusters.

The environmental data were again analysed using k-means with the number of clusters set to six, and the 1 km grid was thus partitioned into six polygons (Figure 36). For each cluster-polygon, the trawl effort layer was clipped in ArcMap, creating six clipped raster layers of fishing effort. Each raster layer was then divided into five equal intervals, and a random sampling station was assigned to each of the five fishing effort bands within each of the six cluster-polygons (e.g., Figure 37). This generated 30 random sampling sites for the study area (Figure 38), giving good coverage across environmental clusters and of the gradient of fishing effort present in each of the environmental clusters.

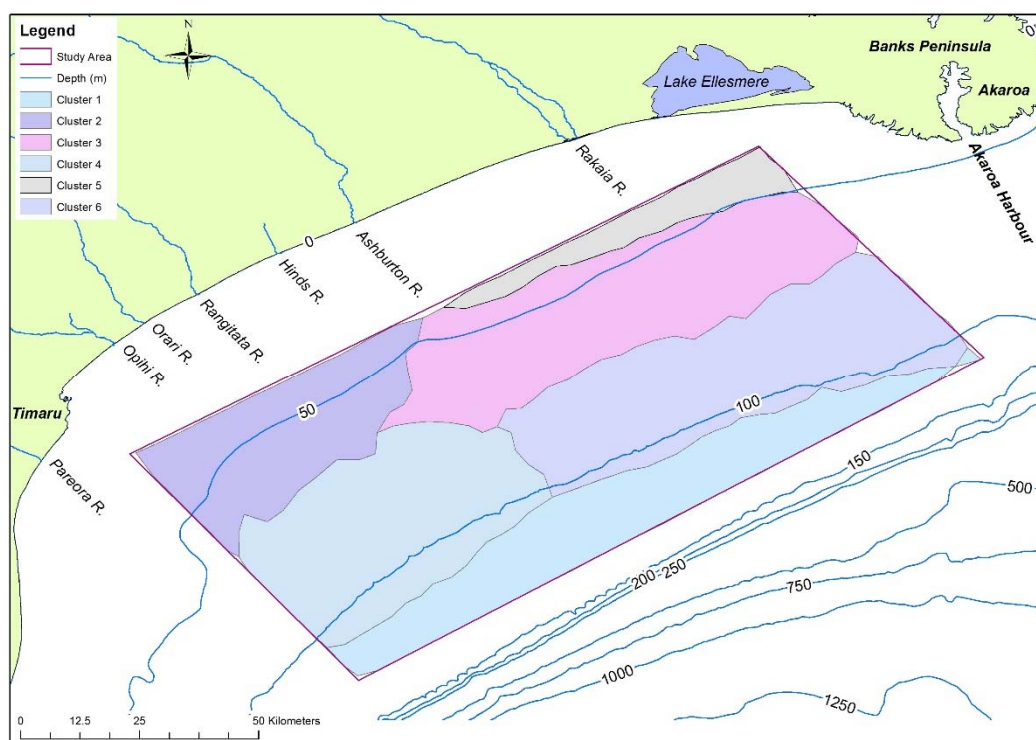


Figure 36: Map of six environmental clusters, determined using k-means clustering of the South Canterbury Bight environmental data.

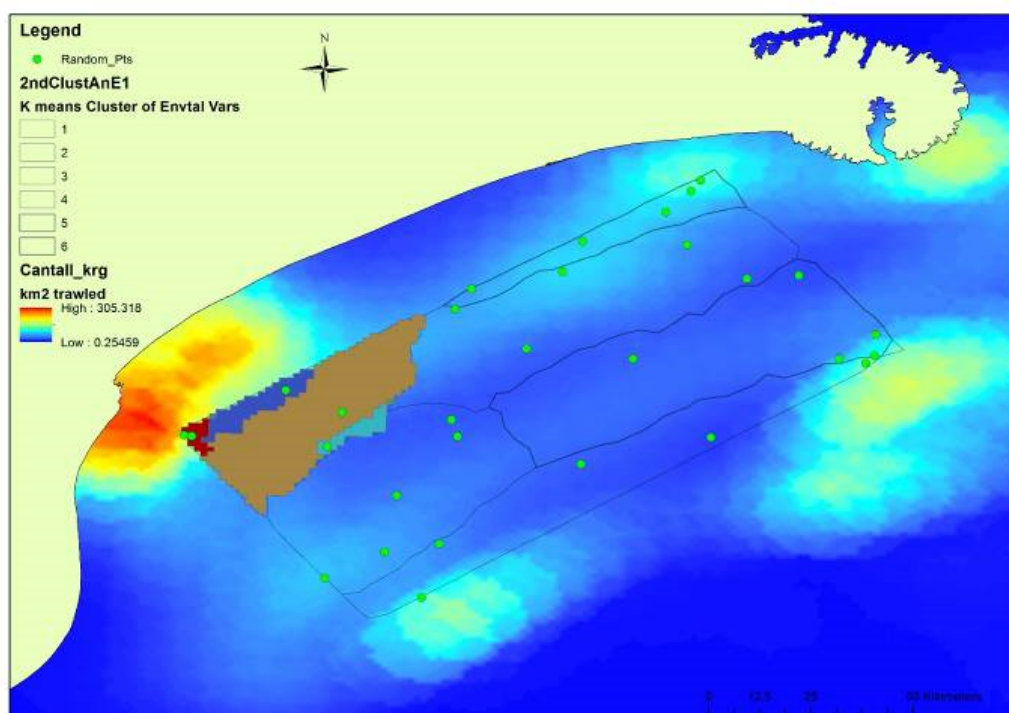


Figure 37: Map of the six South Canterbury Bight environmental clusters, with five levels of fishing effort identified (for one cluster), and random stations allocated.

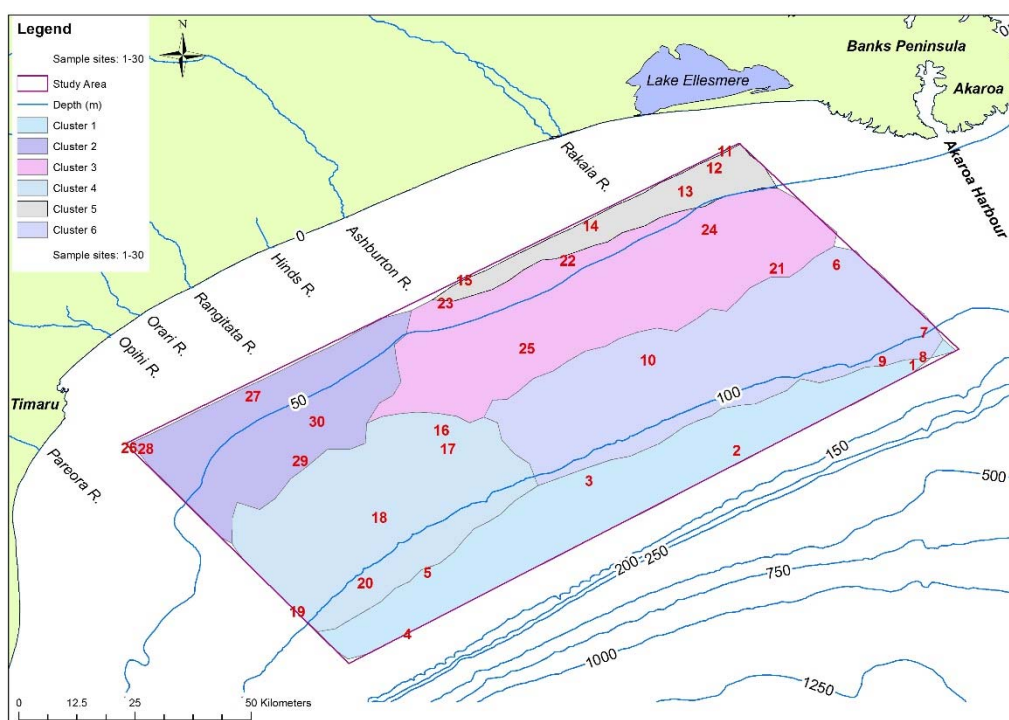


Figure 38: Map showing allocation of sampling stations to the South Canterbury Bight environmental clusters.

Preliminary analysis of effort patterns in the region (based only on TCEPR data) suggested that there was a greater contrast in fishing effort within the study area than is shown in Figure 34 (which includes TCEPR and TECR data). However, despite this reduced contrast in effort, fishing effort gradients

generally ranged from areas being fished once to three to four times over the five year time period, with cluster 2 having areas fished up to eight times (Figure 39).

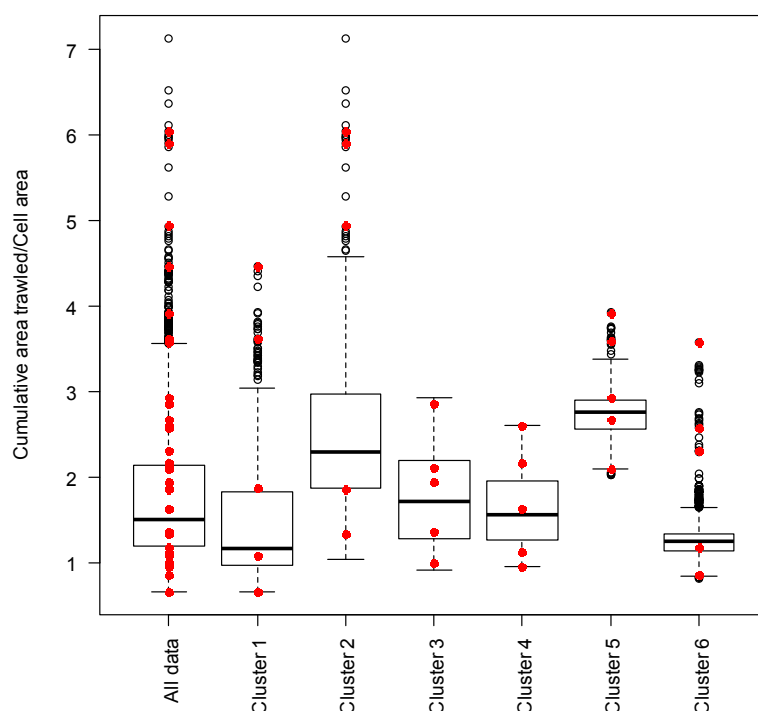


Figure 39: Boxplot showing distribution of fishing effort data extracted by grid cell for the South Canterbury Bight whole study area (All data) and individual clusters. Red symbols overlaid represent the effort levels at each of the sampling locations within each cluster.

Sampling

Sampling was carried out from the RV *Ikatere*, on 7–14 March 2014 following the storms that disrupted the Port of Lyttleton and Akaroa Harbour.

Epifaunal sampling

One 200 m sled tow (Agassiz dredge – 740 mm wide with 28 mm diagonal mesh net; Figure 40) was used to sample epifauna at each of the 30 sites. A GoPro camera equipped with lasers was mounted on the top of the dredge as a potential means of collecting video footage, but a combination of tow speed, poor visibility and low light meant that the video footage was not useful. The complete sled catch was fixed in ethanol for later identification and weighing. Epifaunal community productivity was estimated by applying previously determined Productivity / Biomass ratios (Lundquist & Pinkerton 2008; Pinkerton et al. 2008) to biomass estimates. Both biomass and productivity were examined in terms of total quantity, and total of typical inshore fish prey, on the basis of Stevens et al. (2011).

Infaunal sampling

Within each sampling site, three random grab samples were collected using a Smith-McIntyre grab (0.1 m²) (Figure 41) recording GPS positions of each grab. Infauna were sieved on a 0.5 mm sieve and fixed in ethanol, but first processed to 1 mm in the laboratory. Individual species abundance and biomass were recorded, with biomass used in a similar way to the epifaunal data to estimate infaunal productivity. The 0.5 mm to 1 mm size fractions were retained for future analysis if warranted. Before sieving the samples in the field, the following sub-samples and measurements were recorded:

- Sediment chemistry: A 5 cm diameter, 5 cm depth tube corer was used to sample sediments for analysis for organics content by loss on ignition and Chlorophyll-*a* by freeze dry, acetone extraction on ca.1 g of sediment.
- Sediment grain size: A 5 cm diameter, 5 cm depth tube corer was used to sample sediments for sediment grain-size analysis.



Figure 40: Modified Agassiz dredge used for epi-benthic sled sampling.



Figure 41: Smith McIntyre grab.

Sediments analysis

Sediment samples were wet weighed, and then oven dried to determine moisture content. After drying, they were wet sieved through 2 mm and 1 mm sieves and the filtrate collected on a 63 μm sieve. The 1 mm and 2 mm fractions were then re-dried to constant weight and re-weighed to calculate % sand and shell gravel respectively.

4.1.1 Results

Explanatory variables

Boxplots of the explanatory variables were drawn to determine which were likely to need transformations before analysis (Figure 42). Calcite had a highly skewed distribution, only being detected in coastal regions. Given that the satellite image did not cover the complete study area (A1. 3), it was decided to exclude this variable from the analysis of the faunal communities. The distribution of the grain size shell/gravel was also extremely non-normal and could not be corrected by transformations. The slight non-normality of fishing effort, orbital velocity, M2 speed max and bottom FW max suggested that log-transformations would be useful. However, log transformations did not result in any changes to explanatory power above 1%, and worsened the results for the epifaunal data.

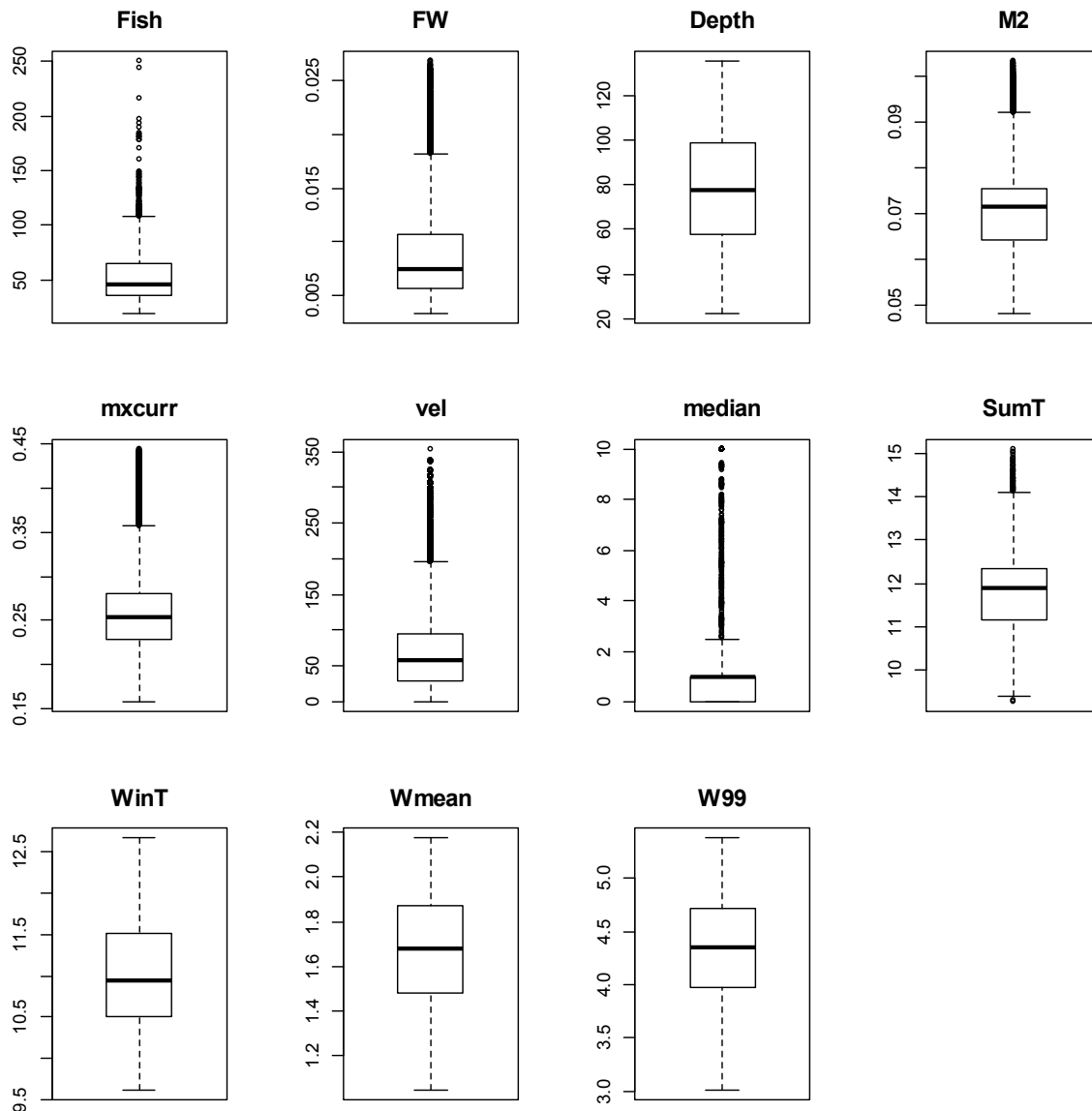


Figure 42: Boxplots of untransformed environmental variables for the South Canterbury Bight study area.

Pearson's R correlation analysis revealed strong correlations between some environmental variables (Table 11).

- The fishing effort data were not correlated with any of the other variables.
- Seabed salinity was negatively correlated with M2 maximum current speed and both wave height statistics. M2 maximum current speed was also positively correlated with the wave height statistics, which were also correlated with each other.
- Mean wave height was negatively correlated with summer seabed temperature and seabed orbital velocity, and positively correlated with depth.
- Depth was also negatively correlated with seabed salinity, both winter and summer seabed temperature and seabed orbital velocity.

- LOI and Moisture content were correlated with % mud.

While the two wave variables (mean and 99th percentile of wave height) were highly correlated (Pearson's R correlations at least 0.97), we decided to initially leave them both in the analysis and let the data drive which was selected. Similarly, although distance to the centre of the quake was highly correlated with distance north, both variables were included in the analysis.

Table 11: Pearson's R correlations over 0.8 observed between environmental variables for the South Canterbury Bight study area. Variable name from Table 10. Variables with no correlations over 0.8 removed from table columns.

	North	M2	median	Wmean	W99	FW	WinT	SumT	vel	Moisture	LOI
East											
North											
M2											
mxcurr											
median											
Wmean		0.92									
W99		0.89		0.98							
FW		-0.83		-0.92	-0.88						
Quake	-0.97										
WinT											
SumT				-0.82		0.87					
vel				-0.86		0.94		0.88			
Fish											
Depth				0.92	0.84	-0.91	-0.81	-0.92	-0.90		
Moisture											
LOI											
Mud										0.82	0.94
Sand											
Shell			0.82								
Chl a											

Principal component analysis (PCA) of the normalised environmental variables, was conducted to investigate how the sites were positioned relative to the variables and whether any of the environmental variables combined to produce strong gradients amongst the sites that could be used as a predictor variable. The first axis represented 47% of the variability, with the second axis representing a further 20% (Figure 43). Five axes were required to account for more than 90% of the variability, suggesting that the environmental variables did not combine to produce strong gradients.

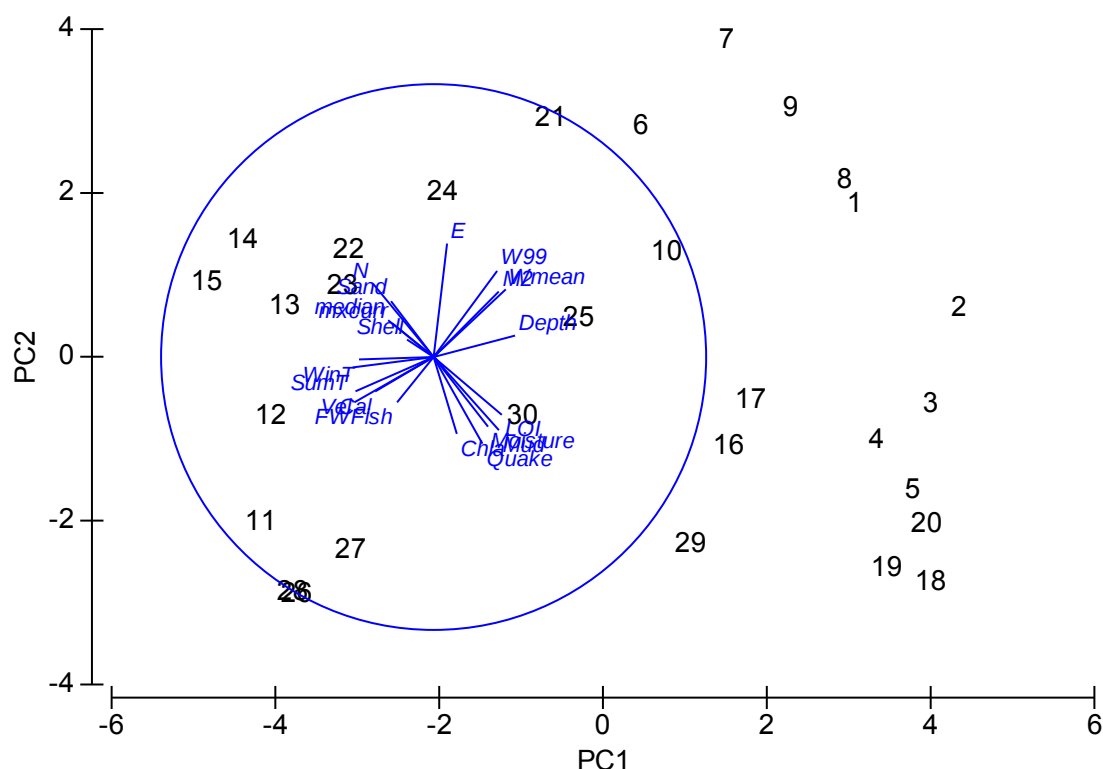


Figure 43: South Canterbury Bight sites, and environmental drivers, along the first two axes of the principal component analysis (accounting for 67.6% of variation). Sites closest together are more environmentally similar. The length of the line representing the environmental variable indicates importance to differences between sites. Variable name abbreviations provided in Table 10.

Effects of fishing on the benthic community

Epifaunal communities

An initial non-metric multidimensional scaling analysis of the epifaunal communities showed a good spread of sites across the ordination space and a satisfactory representation in two dimensions (Figure 44). However, assigning sites to fishing effort categories (five equal intervals) shows that there were very few sites in the higher categories, and there was no clear pattern in the distribution of sites in relation to fishing intensity.

In the DistLM model, backwards selection removed only two environmental variables (seabed orbital velocity and winter SST) leaving the other variables to explain 70% of the variability (Figure 45, Table 12). Fishing effort at a 5 km × 5 km scale explained 5.3% of the total variance, and depth appeared to be the most influential driver of community structure (Figure 45). Addition of variables representing space increased the total amount explained only to 77% with both kilometres North and East retained as important.

Examining the effects of the fishing on individual epifaunal species (using CAP with effort conditioned on other retained variables) suggested that the knobbed whelk *Astrofusus glans*, unidentified hermit crabs and the swimming crab *Nectocarcinus* sp. were negatively correlated with effort, although the correlations were not strong (Figure 46). The squat lobster *Munida gregaria*, Oweniidae polychaetes and a social solitary ascidian were particularly positively correlated with fishing effort, but not strongly.

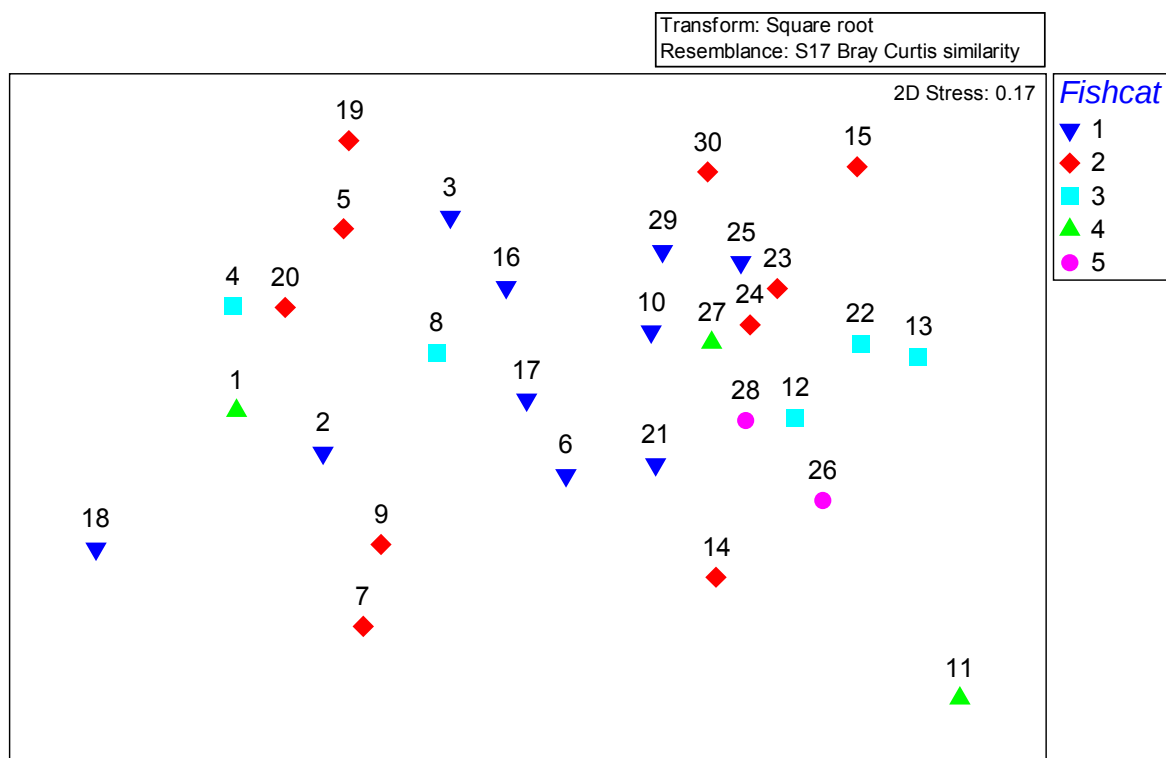


Figure 44: Non-metric multidimensional scaling analysis of the South Canterbury Bight epifaunal communities sampled from video data. Sites that are closest together are most similar. Site labels represent sites presented in Figure 38. Fishcat represents categorical level of fishing effort, with effort increasing from category 1 to category 5.

Table 12: DistLM results: Explanatory power of environmental variables for the South Canterbury Bight epifaunal data based on fishing effort variables only, extra explanatory power for best model based on environmental variables and the best model including spatial variables. Terms marked * are removed from the best model as additional terms are made available, while grey highlighted variables become important as additional terms are made available, suggesting a range of interactions. Variable name abbreviations provided in Table 10. Marginal tests are provided in Appendix 3. Direction of correlation between fishing effort variables and univariate statistics indicated by arrows.

	Fishing effort	Environmental model	Plus spatial variables	Best model
Community	0.05 Fish	0.65 Depth, M2, mxcurr, median, Wmean, W99, FW, Quake, WinT*, SumT, Moisture, LOI, Mud, Sand, Chla	0.07 North, East	0.77
#taxa	0.04 Fish↓	0.61 M2, median, Wmean, W99, FW, Quake, WinT, SumT, vel, LOI, Sand, Chla*	0.16 North, East	0.76
Shannon-Weiner	0.21 Fish↓*	0.53 Depth, M2, mxcurr, median, Wmean, W99, FW, SumT, vel, LOI, Mud, Sand, Chla	0.09 North, East	0.83
Pielou's evenness	0.13 Fish↓*	0.64 Depth, M2*, mxcurr, median, Wmean*, W99, Quake, SumT*, vel, Moisture, LOI, Mud, Chla	0.03 North, East	0.80
Total biomass	0.00	0.70 Depth*, M2, median, Wmean, W99, SumT, Mud, LOI, Sand, Chla*, mxcurr, Quake, WinT, vel	0.10 North	0.80
Fish prey biomass	0.00	0.72 Wmean, W99, Quake, SumT, Mud, Sand, Chla, LOI, Depth, M2, mxcurr, median, WinT, vel, Moisture, LOI	0.13 North	0.85
Productivity	0.00	0.68 Depth*, median, Wmean, W99, SumT, Mud, Sand, Chla*, M2, mxcurr, Quake, WinT, LOI	0.10 North	0.78
Fish prey productivity	0.00	0.74 median, Wmean, FW*, W99, SumT, WinT, LOI, Mud, Sand, Chla, Depth, M2, mxcurr, Quake, vel, Moisture	0.11 North	0.85

Data standardised to 200m tow x 740mm width of Agassiz dredge (148m²)

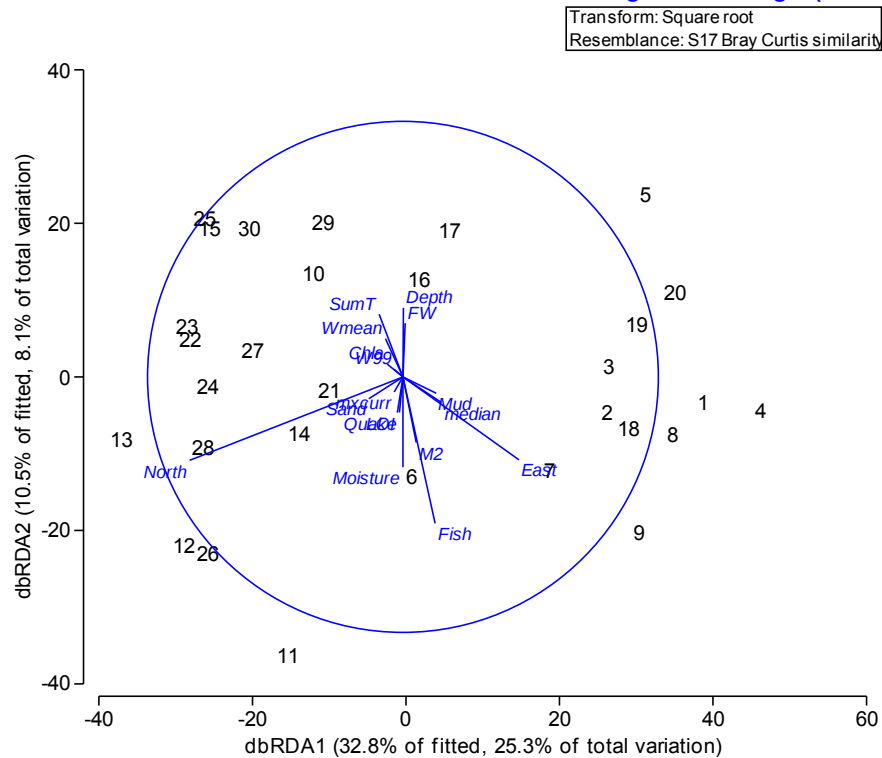


Figure 45: Distance-based redundancy analysis showing the position of sites and environmental drivers selected as best at explaining between site dissimilarities in the South Canterbury Bight epifaunal communities in two dimensions. Variable name abbreviations provided in Table 10.

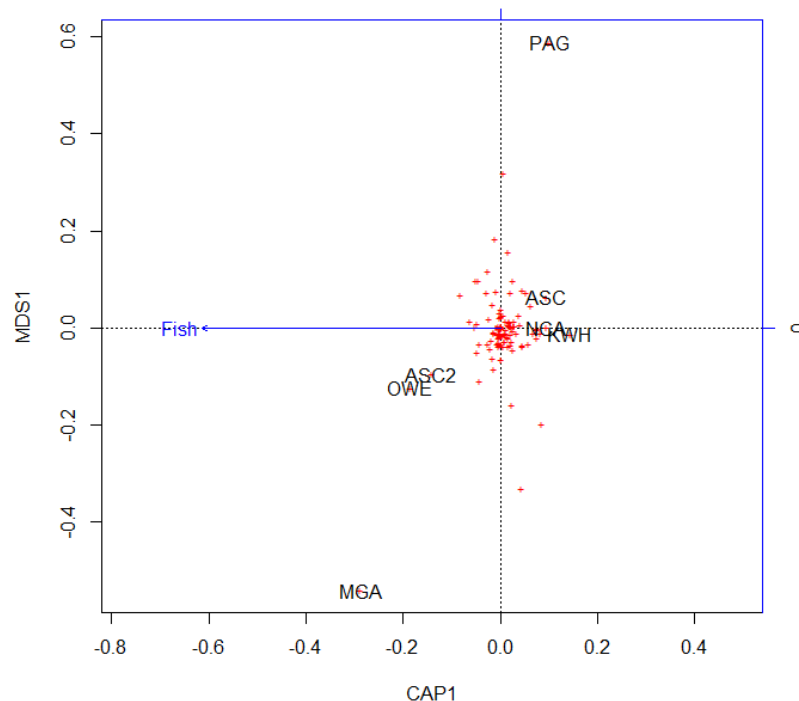


Figure 46: CAP correlation plot for effect of trawl and scallop fishing effort, after partialling out the effects of the other terms retained in the South Canterbury Bight epifaunal community minimum adequate model (Figure 45). ASC – solitary ascidians; ASC2 – social ascidians; PAG – hermit crabs; KWH - *Astrofusus glans*; NCA - *Nectocarcinus* sp; OWE - Oweniidae; MGA – *Munida gregaria*.

Epifaunal diversity, biomass and productivity

The three diversity indices showed an increase in the percentage explained by fishing from Number of taxa (4.5%) through Pielou's evenness (13%) to Shannon Weiner diversity (21%) (Table 12), but fishing was not retained in the final model for Pielou's evenness. All three univariate community measures were negatively related to fishing intensity. When environmental variables were made available to the Number of taxa model, nine were retained, explaining an extra 61%. Some additional environmental variables were retained when spatial variables were made available, with the final model accounting for 76% of the total variance.

For the Shannon-Weiner index, the inclusion of environmental variables (accounting for 53% of the variance) led to the exclusion of the fishing term, although this was retained in the final model once spatial variables were also included, with the final model accounting for 83% of the variance. The inclusion of environmental variables (accounting for 64% of the variance) excluded fishing from the model of Pielou's evenness, and spatial variables were also retained in the final model, which accounted for 80% of the variance.

The measures of epifaunal biomass and productivity showed similar patterns in relation to fishing and environmental drivers (Table 12). The fishing effort was only retained within the models once environmental variables were also offered (generally adding less than 1% to the overall explained variability), and was dropped once spatial variables were offered. The minimum adequate models selected when all variables are offered account for between 78–85% of the variability, but fishing effort was never included as a term in these final DistLM models. The effects of fishing pressure and environmental variables on biomass and productivity were also examined within a generalised linear modelling framework. Using a forward stepwise model selection approach, and retaining terms increasing AIC, fishing effort was not retained in the model examining total epifaunal biomass, but was retained in models examining fish prey biomass, and both total epifaunal and fish prey productivity. On the basis of the minimum adequate models generated (comparing model estimated biomass from the original data set with predictions with no fishing), across the sites studied, fishing has reduced the epifaunal fish prey by 21–88% (median across sites studies of 54% decrease), while total epifaunal productivity is reduced by 10–63% (median 30% decrease) and fish prey productivity is reduced by 22–90% (median 55% decrease).

Infaunal communities

An initial non-metric multidimensional scaling analysis of the infaunal communities showed a good spread of sites across the ordination space and a satisfactory representation in two dimensions (Figure 47). A number of the lower fishing effort sites were tightly clustered in the MDS, while the higher effort sites were more spread.

Backwards selection removed seven of the environmental variables (moisture, summer and winter temperature, M2 speed, maximum current speed, FW and depth) leaving the other variables to explain 64% of the variability (Figure 48, Table 13). Fishing effort at a 5 km scale explained 9% of the total community variance, with sediment parameters and seabed orbital velocity having a greater influence on community composition (Figure 48). Addition of variables representing space increased the total amount explained to 70% with both kilometres North and East retained as important.

Examining the effects of the fishing on individual infaunal species (using CAP with effort conditioned on other retained variables) identified the polychaete *Owenia* sp. and the amphipod *Diastylis* sp. as being positively correlated with fishing effort (Figure 49). No species were very strongly negatively correlated with fishing effort, but the most negatively correlated included the amphipods *Liljeborgia* sp., *Bathymedon* sp., and *Harpiniopsis nadania*, the polychaetes *Prionospio* sp., *Opheliidae* sp. and *Aricidea* sp., the heart urchin *Echinocardium cordatum* and the brittle star *Amphiura* sp..

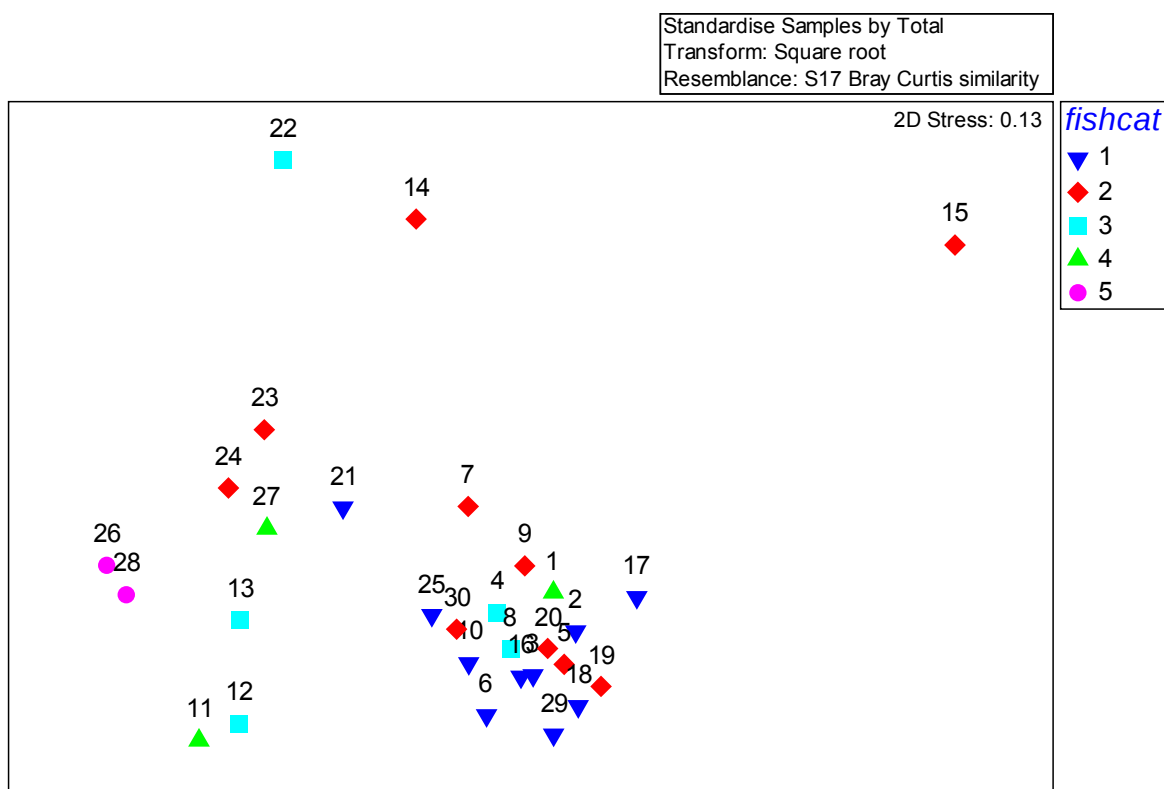


Figure 47: Non-metric multidimensional scaling analysis of the South Canterbury Bight infaunal communities. Sites that are closest together are most similar. Fishcat represents categorical level of fishing effort, with effort increasing from category 1 to category 5.

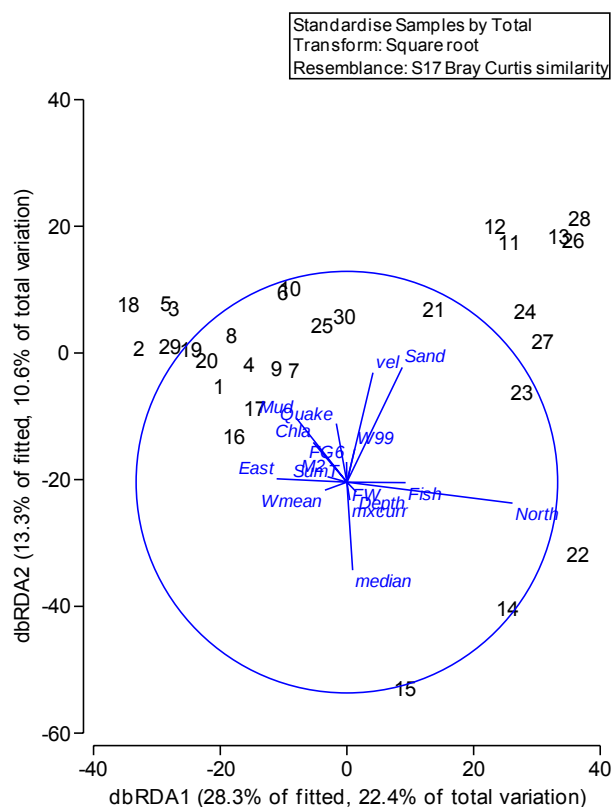


Figure 48: Distance-based redundancy analysis showing the position of sites and environmental drivers selected as best at explaining between site dissimilarities in the South Canterbury Bight infaunal communities in two dimensions. Variable name abbreviations provided in Table 10.

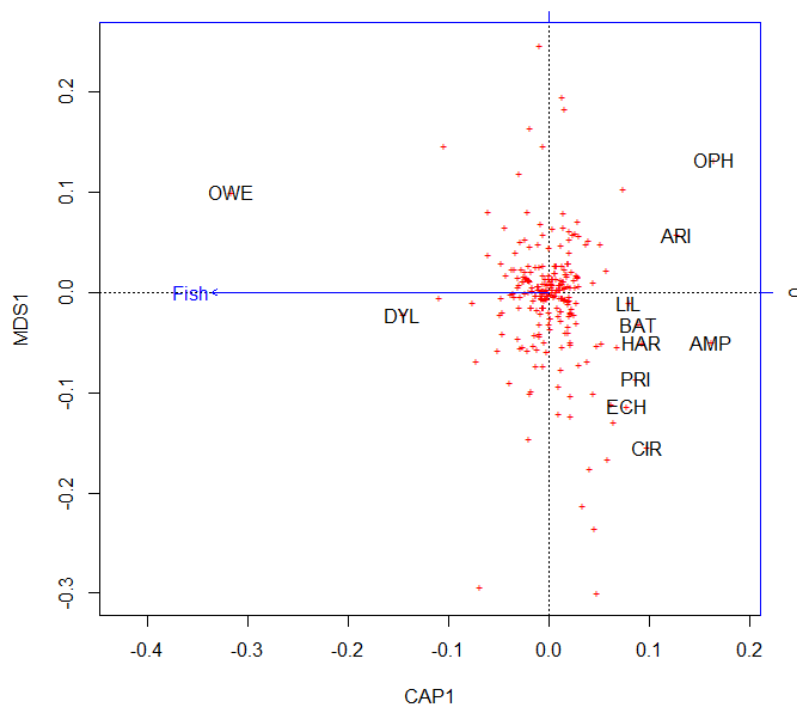


Figure 49: CAP correlation plot for effect of trawl and scallop fishing effort, after partialling out the effects of the other terms retained in the South Canterbury Bight infaunal community minimum adequate model (Figure 48). OWE – *Owenia* sp.; DYL – *Diastylis* sp.; OPH – *Opheliidae* sp.; AMP – *Amphiura* sp.; ARI – *Aricidea* sp.; CIR - *Cirratulidae*; HAR – *Harpiniopsis nadania*; BAT – *Bathymedon* sp.; PRI – *Prionospio* sp.; LIL – *Liljeborgia* sp.; ECH – *Echinocardium cordatum*.

Table 13: Explanatory power of environmental variables for the South Canterbury Bight infaunal data based on fishing effort variables only, extra explanatory power for best model based on the abundance of erect structure forming epifauna (FG6), environmental variables and the best model including spatial variables. Terms marked * are removed from the best model as additional terms are made available, while grey highlighted variables become important as additional terms are made available, suggesting a range of interactions. Variable name abbreviations provided in Table 10. Direction of correlation between fishing effort variables and univariate statistics indicated by arrows.

	Fishing effort	FG6	Environmental model	Plus spatial variables	Best model
Community	0.09 Fish	0 FG6	0.62 mxcurr, median, Wmean, W99, FW, Quake, vel, LOI, Mud, Sand, Chla	0.07 North, East	0.79
#taxa	0	0.05 FG6	0.75 depth, median, Wmean, Quake, SumT, Mud, Sand, WinT, vel, LOI	0.04 North, East	0.84
Shannon-Weiner	0 Fish↓	0 FG6	0.81 depth, median, Wmean, W99, FW, Quake, SumT, LOI, Mud, Sand, Chla	0.05 North, East	0.86
Pielou's evenness	0 Fish↓	0.06 FG6	0.66 depth*, M2*, median, Wmean, W99, FW, Quake, SumT, Moisture*, LOI, Mud, Sand	0.08 North	0.80
Total biomass	0.06 Fish↑	0.04 FG6*	0.61 depth, M2, mxcurr, median, Wmean, W99*, FW, Quake, SumT, Mud*, Sand, vel, LOI	0.08 North	0.79
Fish prey biomass	0.04 Fish↑	0.07 FG6*	0.62 Depth, M2, median, Wmean*, W99, FW, SumT, vel*, Mud, Sand, mxcurr, Quake, Moisture, LOI	0.17 East, North	0.80
Productivity	0.10 Fish↑	0.04 FG6	0.51 Depth, M2, median, Wmean*, W99*, FW, Quake, WinT*, SumT, Mud, Sand, mxcurr, vel, Moisture, LOI, Chla	0.24 East, North	0.85
Fish prey productivity	0.07 Fish↑	0	0.44 Depth, M2, median, FW, WinT*, SumT, LOI, Sand, mxcurr, Quake, Moisture, Mud, Chla	0.29 East, North	0.80

Infaunal diversity, biomass and productivity

Number of species, the Shannon-Weiner index and Pielou's evenness were best explained by a number of environmental variables but fishing effort was not one of them (Table 13). However, the abundance of erect epifaunal structure forming species (FG6) was retained as an explanatory variable for number

of species and Pielou's evenness (explaining 5–6% of the variability). Sediment parameters provided most explanatory power for the number of species, with other environmental variables being comparable to FG6, while for Pielou's evenness, FG6 provided the most explanatory power (from marginal test).

The different measures of infaunal biomass and productivity showed similar patterns in relation to environmental drivers (Table 13), and also showed positive correlations with fishing effort and FG6. The minimum adequate models selected when all variables are offered account for between 79–84% of the variability, with fishing effort included as a term. The effects of fishing pressure and environmental variables on biomass and productivity were also examined within a generalised linear modelling framework. Using a forward stepwise model selection approach, and retaining terms increasing AIC, fishing effort was not retained in any of the models.

Functional traits

In order to better determine whether macrofaunal communities associated with higher levels of fishing effort were structured as might be predicted from the typical effects of fishing documented in the literature, despite the low amount of variability explained by fishing effort, we examined changes in functional groups across fishing effort gradients (Lundquist et al. 2013).

The abundance of Canterbury epifaunal taxa comprising Functional group 6 varied with fishing effort (Figure 50-A). A typical factor ceiling response is seen whereby at low fishing effort the abundance of Functional group 6 can vary from 0 to 12 in response to a number of factors, but as fishing effort increases, the amount of variability decreases and the maximum abundance attainable decreases. This type of response is difficult to analyse by ANOVA and regressions which focus on mean responses, as can be seen by considering the mean and standard error of the Functional group 6 abundances in each of the four fishing effort classes (Figure 50-C). Standard techniques to analyse for factor ceilings include: (1) calculating the upper percentiles (or maximums) of the distributions within classes and running weighted regressions on these values (Blackburn et al. 1992; Cummings et al. 2003); and (2) quantile regressions, where regression can be focussed on a specific percentile (Cade et al. 1999). The 90th percentile found in each fishing effort class and the significant polynomial regression of them against fishing effort is given in Figure 50-B. Quantile regression (on the 90th percentile) detected a significant negative effect of fishing effort squared on the abundance of Functional group 6 ($p = 0.0216$).

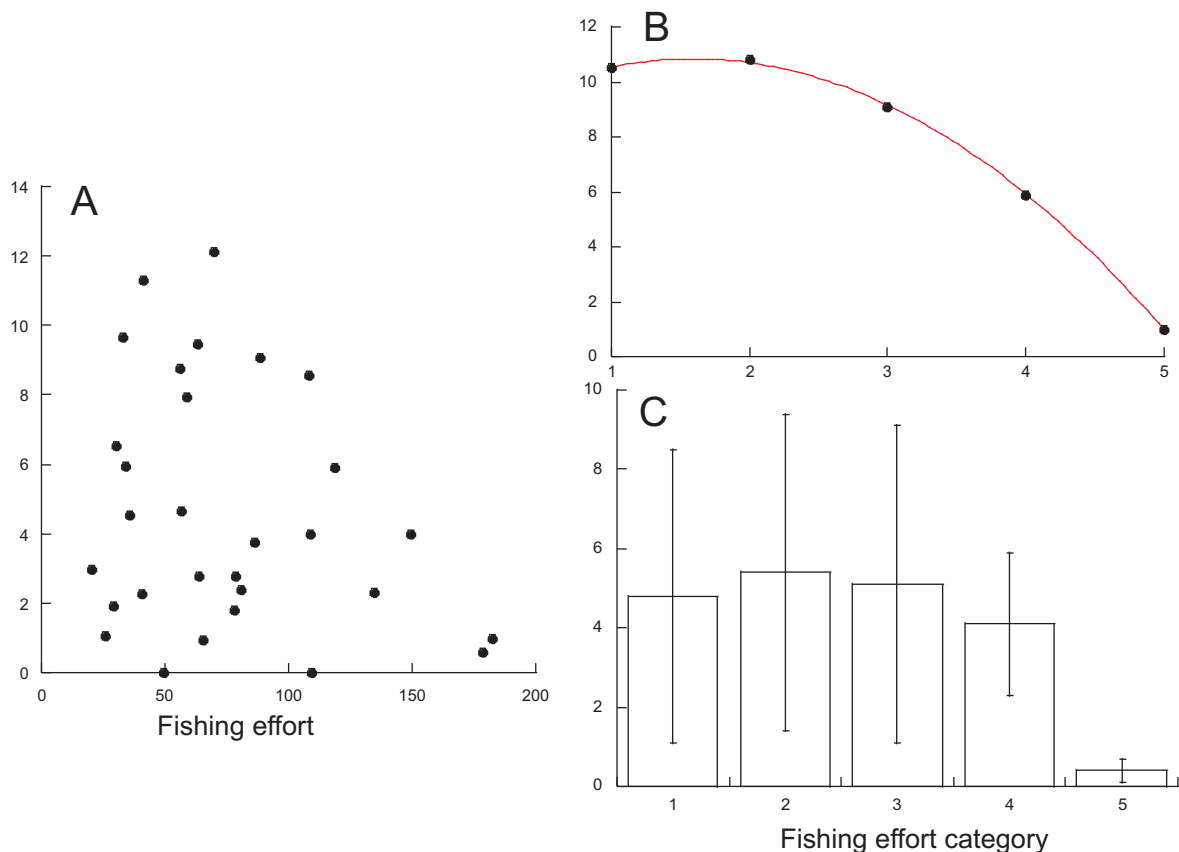


Figure 50: A- The abundance of Canterbury epifaunal comprising Functional group 6 varies with fishing effort (aggregate area swept over 5 years within a 5 km × 5 km cell), B- 90th percentiles of the abundance of Functional group 6 in each fishing effort class, C – mean abundance and standard errors for the 5 fishing effort classes.

4.1.1 Conclusions

The analyses of the epifaunal and infaunal communities were reasonably similar in terms of variables that were important in explaining the observed variance, and the overall variance explained. Both fishing effort and environmental variables were identified as being important in explaining the patterns in the community data observed, although fishing effort accounted for only a relatively small component of the overall variance (5 – 9%). The important environmental variables for both infaunal and epifaunal community analysis, included sediment grain size and organic carbon parameters, wave height parameters, chlorophyll-*a*, and distance from earthquake epicentres (although it must be remembered that this parameter was correlated with distance north). Epifaunal structure was less consistently retained in final models for infauna than in the Tasman and Golden Bays study. Some spatial patterns were also identified within the study area.

The knobbed whelk, hermit crabs, swimming crabs, some ascidians, heart urchins, brittle stars and a number of polychaetes and amphipods were negatively correlated with fishing effort. Oweniidae polychaetes were found to be positively correlated with fishing intensity in both the epifaunal and infaunal data sets. Reiss et al. (2009) found Oweniidae abundance to increase with fishing intensity, although biomass (not examined here) decreased. While whelks are a QMS species, actual landings in the region are thought to be minimal (Ministry for Primary Industries 2014), and so commercial harvest is unlikely to be responsible for this relationship. However, whelks have previously been found to be sensitive to physical disturbance from fishing, which was considered to increase vulnerability to predation (Ramsay & Kaiser 1998), and this may also be the case here. *Echinocardium* have previously been identified as being particularly sensitive to the effects of fishing (Eleftheriou & Robertson 1992; Jennings & Kaiser 1998), and have also been shown to provide valuable ecosystem services through

their bioturbation activities (Lohrer et al. 2004). Kaiser & Ramsay (1997) found that fishing disturbance was associated with a shift in flatfish feeding, from “grazing” on *Amphiura* arms to consumption of oral discs, the animals having been exposed by the fishing disturbance, thereby increasing predation pressure.

Examining the univariate measures, fishing effort consistently accounted for a greater proportion of the overall variance for the epifauna than the infauna, but other than for the Shannon-Weiner index for epifauna (21%), the proportion of variance accounted for by fishing was relatively low (no more than 11%), and fishing was not retained in the models for infaunal diversity measures, although epifaunal structure was retained for models of infaunal species richness (number of taxa) or Pielou’s evenness. Where fishing intensity was retained as a term in the models explaining the univariate measures, its effect was consistently negative in nature. Epifaunal and infaunal biomass and productivity appear to be affected by fishing, although the effects were not consistent between modelling approach or fauna. Fishing effort was not retained in the final DistLM models for epifauna, but across the study sites examined, the generalised linear models predicted that fishing had reduced fish prey by 54% (median), and total epifaunal and fish prey productivity by 30% and 55% respectively (medians). Fishing effort was consistently retained in final DistLM models for infaunal biomass and productivity, but was not retained in any of the generalised linear models.

Although the strength of the fishing effects (up to 21%, but more often 5 – 11% and not significant for some measures) appears less than in studies of other areas of New Zealand (Thrush et al. 1998; Cryer et al. 2002; Tuck & Hewitt 2013), this may reflect the relatively narrow range of fishing effort across the study area. While fishing effort gradients were identified and sampled within each habitat cluster, only two stations were located in particularly high effort areas. In addition to the weak (but mostly significant) fishing effects detected in relation to species based community and univariate measures, functional trait effects were also detected, with the predicted factor ceiling response identified for long-lived, sedentary, habitat-forming species, and a significant negative effect of fishing identified on this functional trait group.

4.2 Opportunistic analysis of survey bycatch data

In order to compare the results obtained from the dedicated benthic community sampling described above with more opportunistically collected data, the invertebrate trawl bycatch community from recent ECSI survey stations (Beentjes et al. 2013) in the South Canterbury Bight region (Figure 51) was examined in relation to the same environmental data used in the habitat stratification (above), along with fishing effort data (2007/08 – 2011/12).

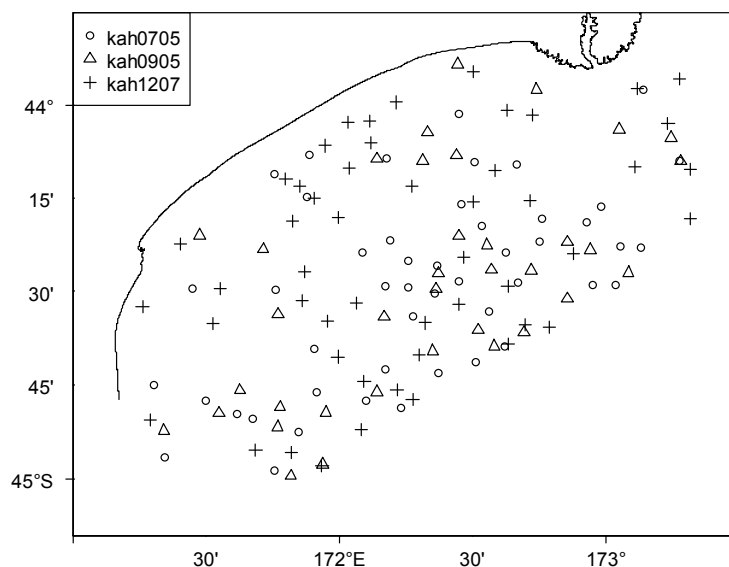


Figure 51: East Coast South Island trawl survey stations for which benthic invertebrate bycatch have been recorded in recent surveys, within the South Canterbury Bight area.

The benthic community data comprised 59 OTUs recorded from 134 stations in the South Canterbury Bight area, from the 2007, 2009 and 2012 surveys.

The Bray Curtis similarity matrix of the square root transformed, standardised community data was analysed in relation to the environmental variables (listed in Table 10, along with survey year) with DistLM, using backwards selection based on the AIC criterion. DistLM marginal tests for each variable are provided in Table 14, and none of the terms explained more than about 10% of the variance in the community composition individually, and most explained very little.

The backward model selection procedure retained terms for sediment grain size, maximum tidal current, tidal M2 current, summer surface temperature, mean wave height, depth, calcite, freshwater input, and survey, explaining 31.7% of the variance in the bycatch community data (Figure 52). Fishing effort was not retained in the model, indicating that it does not explain a significant proportion of the variance in the community composition.

Table 14: DistLM marginal tests for analysis of the South Canterbury Bight trawl bycatch composition. Prop represents the proportion of the variance explained by each term when fitted individually. Variable name abbreviations provided in Table 10.

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df
median	13272	3.500	0.001	0.026	132	2
mxcurr	20466	5.476	0.001	0.040	132	2
SumT	45873	12.940	0.001	0.089	132	2
Wmean	42701	11.964	0.001	0.083	132	2
M2	8532	2.229	0.022	0.017	132	2
depth	42093	11.779	0.001	0.082	132	2
FW	38482	10.686	0.001	0.075	132	2
W99	18108	4.822	0.001	0.035	132	2
year	32596	4.437	0.001	0.063	131	3

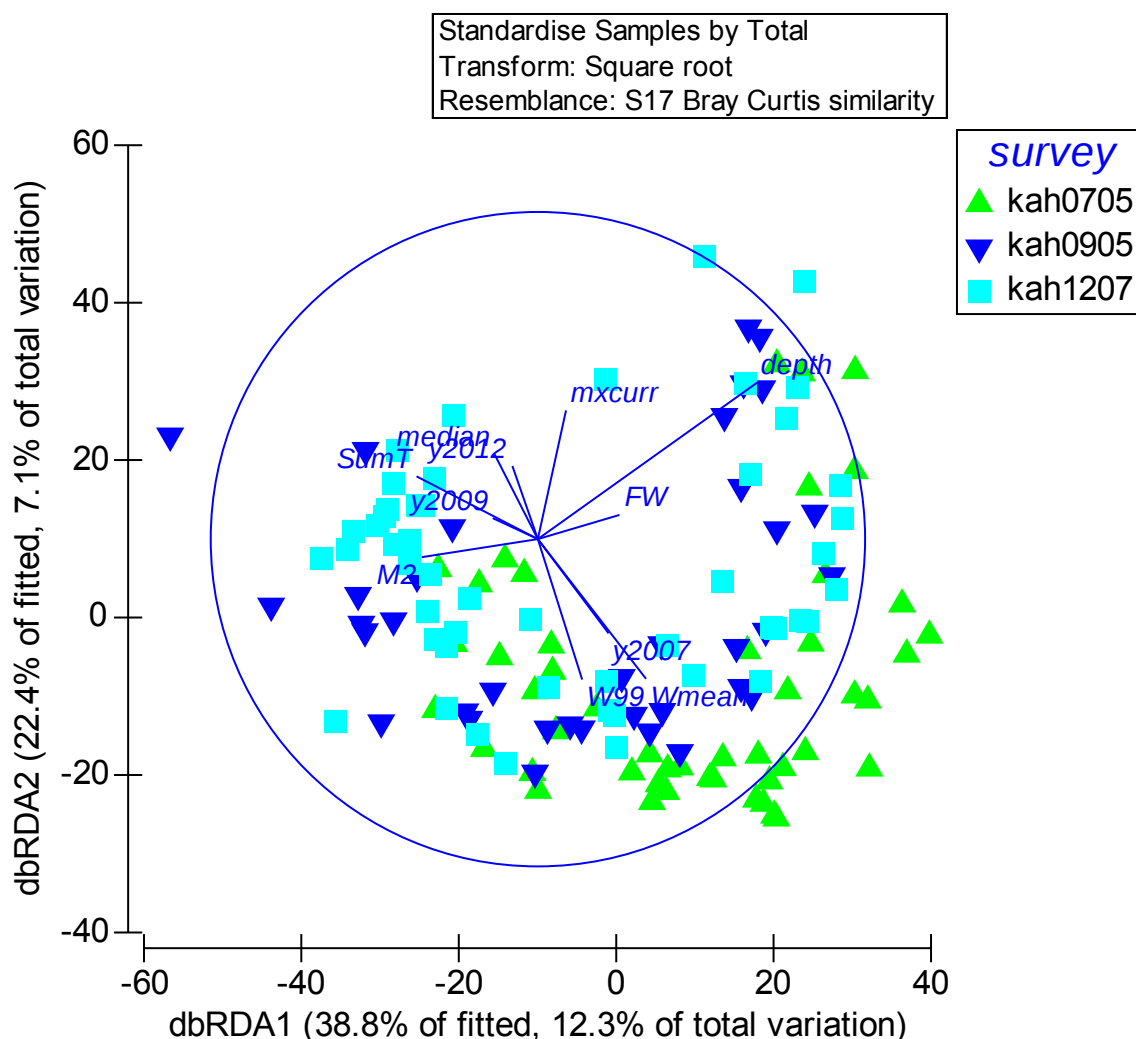


Figure 52: Distance-based redundancy analysis showing the position of sites and environmental drivers selected as best at explaining between site dissimilarities in the South Canterbury Bight trawl survey bycatch communities. Variable name abbreviations provided in Table 10.

The analysis was repeated using CAP, with the same terms retained (not shown), with 25.2% of the total variance accounted for. The results were not sensitive to the similarity matrix or data transformation. The correlation of species abundance with fishing effort was examined, having partialled out the effects of those terms included in the minimum adequate model. As might be expected, given that fishing effort was not retained in the minimum adequate model, none of the species were strongly correlated with fishing effort (not shown).

The DistLM and CAP analyses were also repeated, examining only the bycatch data from the 2012 survey, on the basis that the surveys in 2007 and 2009 may have been influenced by fishing effort patterns prior to the 2007/08 fishing year, for which we do not have fine resolution (tow by tow) data. In both analyses, overall explanatory power was increased slightly from the full data set, but neither retained fishing effort as a significant term.

4.2.1 Conclusions

The spatial extent of the trawl survey stations included in this analysis (Figure 51) was slightly larger than the study area considered in the dedicated benthic community study (Figure 38; Section 4),

providing a slightly greater contrast in fishing effort gradient. However, while the same general suite of environmental variables were retained in the analysis of the trawl survey benthic bycatch community (waves, currents, depth, temperature, sediment parameters, salinity) as the dedicated benthic sampling study, fishing effort did not explain a significant proportion of the trawl bycatch community variance and was not retained in the full model. This same pattern was observed when only examining the most recent survey data, suggesting our ability to detect an effect of fishing was not being confounded by the effects of relatively recent fishing effort (but still prior to fine scale reporting) on the earlier surveys.

4.3 South Canterbury Bight overall conclusions

The fishing effort data used in the analysis for this region was based on a recent analysis of data reported since the 2007/08 fishing year, largely by inshore trawlers. The overall strength of the fishing effort gradient was somewhat weaker than that estimated for the Tasman and Golden Bays study, and the strength of effects detected was generally less. Consistent with the Tasman and Golden Bays study, fishing appears to have reduced some measures of epifaunal biomass and productivity. All three South Canterbury Bight studies (dedicated sampling of epifauna and infauna, and opportunistic sampling of epifauna) identified a similar group of environmental variables (sediment parameters, depth, wave parameters) as being important drivers for the benthic communities, but while the dedicated studies detected weak fishing effects (accounting for 5 – 9% of the total community variance, with a stronger effect detected for infauna), no significant fishing effect was detected in the trawl bycatch data. In contrast to the community composition data, when examining the univariate community measures, fishing effort had a greater effect on the epifauna than the infauna.

The species identified as being most strongly negatively correlated with fishing effort (e.g., whelks, heart urchins, brittle stars) have been identified as being sensitive to fishing pressure in previous studies elsewhere, either through sensitivity to physical impact, or to disturbance and subsequent predation.

While the strength of fishing effects were moderate or weak, compared to levels reported previously (over stronger effort gradients) in New Zealand, effects on a broad functional group (composed of long-lived, sedentary, habitat-forming species) were apparent, with maximum abundance reducing by about half in areas fished (on average) more than once per year.

5. CHATHAM RISE AND CHALLENGER PLATEAU

5.1 Opportunistic benthic community sampling

During 2006 and 2007, three Ocean Survey 20/20 voyages were conducted, to map and compare the distribution of seabed habitats and their associated biological diversity across the Chatham Rise and the Challenger Plateau (Bowden 2011). The principal objectives were: to determine the distribution of bottom habitats and biodiversity on the seabed in depths from 200 to 1200 m on the Chatham Rise and the Challenger Plateau; to assess the utility of the Marine Environment Classification (Snelder et al. 2004) as a proxy for habitat types and biodiversity distribution and further develop habitat mapping techniques; and to assess the influence of bottom trawling as a broad-scale driver of sea-bed biodiversity. Analysis of the samples collected from these voyages was funded under the MPI projects ZBD200701 and BEN200701, with various analyses of the benthic communities documented in a series of reports (Hewitt et al. 2010; Bowden 2011; Hewitt et al. 2011; Floerl et al. 2012). None of the analyses conducted to date have specifically examined the benthic communities in relation to gradients of fishing pressure, although Lundquist et al. (2013) did so to some extent using functional traits.

Bowden (2011) provides full details of the various sampling activities conducted during the voyages. Analysis described here examined the data from DTIS still images, the seamount sled and beam trawl.

Data collected by the three sampling approaches were initially examined separately, but owing to the low level of beam trawl sampling, it was decided to combine these data with the seamount sled data, allowing for sampling method as an explanatory factor in the analysis. DTIS sampling was completed at all sites, but the seamount sled and beam trawl were employed at different subsets of the sites. Benthic community composition was examined in relation to environmental variables and fishing pressure. Some environmental variables were extracted from existing GIS layers (and so were available for all sites), while others, particularly related to sediments, were only available where physical sediment sampling was successful (Nodder et al. 2011). Fishing effort data (bottom trawl and midwater trawl within 1 m of the seabed) were extracted from GIS layers generated within MPI project DAE2010/04A (Black et al. 2013). This dataset only includes fishing effort reported on TCEPR within the New Zealand EEZ. There is unlikely to be significant trawl effort relevant to the Chatham Rise and Challenger Plateau areas that has not been reported on the TCEPR system. Some of the Challenger Plateau stations are outside the New Zealand EEZ, however, and therefore no effort data are available for these. Stations with missing sediment or fishing effort data have been excluded from the analyses. Patterns in fishing effort were initially examined over years, but patterns were highly correlated, and so benthic community composition was examined in relation to a single fishing effort term (average annual swept area over the 2001/02 – 2006/07 fishing years). Although one of the objectives of the voyages was to assess the influence of bottom trawling as a broad-scale driver of sea-bed biodiversity, in balancing this with the other objectives it was not logistically possible to sample across the full extent of the fishing effort gradient on the Chatham Rise and Challenger Plateau areas, and the effort range over which sampling was conducted was quite narrow (averaged over the five years prior to the surveys, sampled cells were fished from 0 – 4 times per year).

5.1.1 DTIS sampling

The benthic community data from the DTIS analysis comprised 233 OTUs recorded from 100 stations (73 on the Chatham Rise, 27 on the Challenger Plateau)(Figure 53). These data were limited to OTUs of benthic invertebrates, and excluded fish. An initial non-metric multidimensional scaling analysis of the DTIS epifaunal communities showed a distinction between the Chatham Rise and Challenger Plateau areas, although the plot had a high stress level, suggesting that the stations were not well represented in two dimensions (plot not shown).

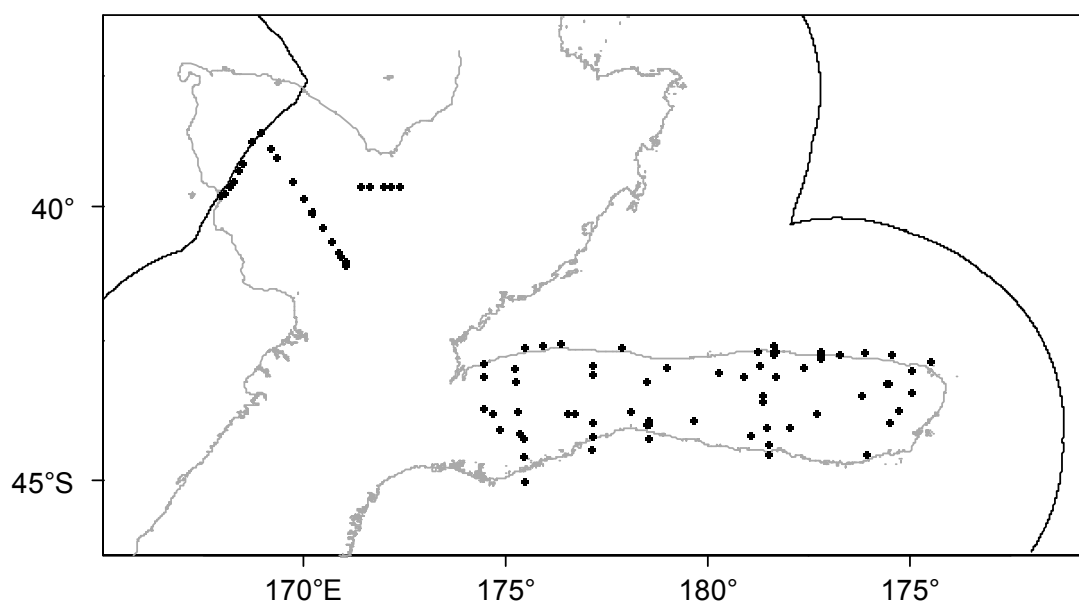


Figure 53: Chatham Rise and Challenger Plateau Ocean Survey 20/20 station locations included in DTIS analysis. 1000 m depth contour and limit of EEZ also shown.

The Bray Curtis similarity matrix of standardised square root transformed community data was analysed in relation to the environmental variables with DistLM, using backwards selection based on the AIC criterion. Explanatory variables included in the analysis were taken from data layers collated by Compton et al. (2013) and Nodder et al. (2011), where further details of their derivation can be found, and are listed in Table 15. Fishing intensity data were from Black et al. (2013).

Table 15: Environmental variables available for use in the statistical analysis, and abbreviations used in later plots and tables for Chatham Rise and Challenger Plateau case study.

Data	Metric	Abbreviation
Fishing effort	Average fishing intensity (5 years prior to survey)	effort0207
Broader scale environmental data derived from models	Sea surface temperature anomaly	sstanom
	Sea surface temperature gradient	sstgrad
	Winter sea surface temperature	sstwin
	Seabed temperature	bedtemp
	Sea surface productivity	vgpm
	Depth averaged maximum tidal current	tidalcur
	Sea surface salinity	sal
	Seabed temperature residual	tempres
	Dissolved organic matter	disorgm
	Sea surface temperature anomaly amplitude	sstanam
Site specific data collected during the survey	Depth	bathy
	Seabed slope (derived from multibeam)	slope
	Seabed roughness (derived from multibeam)	rough
	Calcium carbonate in the sediment	CaCO ₃
	Phaeopigments in the sediment	Phaeo
	Sediment % mud	mud
	Sediment community oxygen consumption	SCOC
	Sediment total organic matter (loss on ignition)	TOM
	Sediment moisture content	water
Sampling method	Sampling gear (only included in beam trawl / sled)	Gear
Spatial correlates	Latitude	lat
	Longitude	lon
	Region (Chatham Rise or Challenger Plateau)	Chat or Chal

The backward model selection procedure retained terms for location, sea surface temperature (SST) anomaly, winter SST, salinity, sea surface productivity, depth averaged maximum tidal current, dissolved organic matter, suspended particulate matter, slope, depth, Calcium carbonate in the sediment, phaeopigments, region and fishing effort, explaining 36.6% of the variance in the bycatch community data (Figure 54). DistLM marginal tests for each variable are provided in Table 16, and none of the terms explained more than 10% of the variance individually. Fishing effort was only able to explain 2.5% of the variability (1.4% uncorrelated with other terms), although none of the variables explained a high percentage individually, and fishing effort explained just under half the variability explained by the strongest environmental variable (bathymetry).

Table 16: DistLM marginal tests for analysis of the Chatham Rise and Challenger Plateau DTIS community composition. Prop represents the proportion of the variance explained by each term when fitted individually. Variable name abbreviations provided in Table 15.

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df
lon	13 082	3.889	0.001	0.038	98	2
lat	16 770	5.042	0.001	0.049	98	2
sstanom	8728.8	2.561	0.001	0.025	98	2
sstwin	17 709	5.339	0.001	0.052	98	2
sal	17 115	5.151	0.001	0.050	98	2
vgpm	11 781	3.488	0.001	0.034	98	2
tidalcur	9988.6	2.942	0.001	0.029	98	2
disorgm	13 218	3.931	0.001	0.039	98	2
suspart	13 152	3.911	0.001	0.038	98	2
slope	12 686	3.767	0.001	0.037	98	2
bathy	20 561	6.254	0.001	0.060	98	2
CaCO3	10 651	3.143	0.001	0.031	98	2
Phaeo	10 028	2.954	0.001	0.029	98	2
effort0207	8736.5	2.563	0.001	0.025	98	2
Area	15 169	4.538	0.001	0.044	98	2
long	13 082	3.889	0.001	0.038	98	2

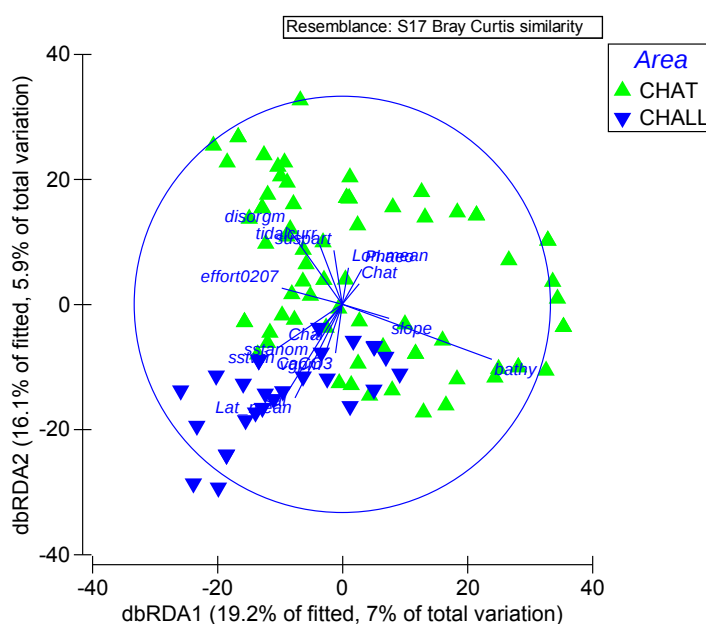


Figure 54: Distance-based redundancy analysis showing the position of sites and environmental drivers selected as best at explaining between site dissimilarities in the Chatham Rise and Challenger Plateau DTIS benthic community data. Variable name abbreviations provided in Table 15.

The analysis was repeated using CAP with terms for latitude, longitude, winter sea surface temperature, sea surface temperature anomaly, salinity, sea surface productivity, tidal current speed, dissolved organic matter, suspended particulate matter, seabed slope, depth, % Calcium carbonate, phaeopigments, area and fishing effort retained with backward model selection using AIC (Figure 55), accounting for 35.3% of the total variance.

The correlation of species abundance with fishing effort was examined, having partialled out the effects of the other terms retained in the minimum adequate model (Figure 56). Most species showed little evidence of correlation with fishing effort, but Quill worms (Onuphidae) and the echinoderm

Gracilechinus multidentatus appeared to show positive and negative correlation with the fishing effort variable, respectively. Previous studies have identified increases in scavenger numbers (Ramsay et al. 1996; Ramsay et al. 1998), and decreases in abundance of fragile species (Kaiser et al. 2006) in relation related to fishing disturbance. Other studies on the Chatham Rise have also suggested quill worms respond positively to fishing disturbance (David Bowden pers. comm.). Functional group abundance (for group 4 substrate destabilisers, and group 6 emergent epifauna) (from Lundquist et al. 2013) have also been examined in relation to fishing intensity (Figure 57). Both groups showed weak effects over the relatively limited range of fishing intensity observed.

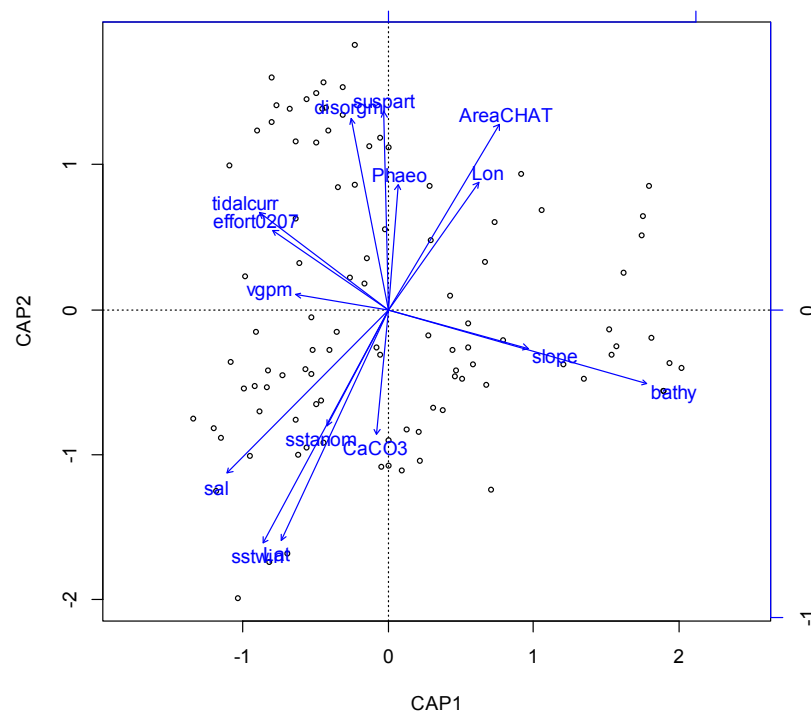


Figure 55: CAP correlation plot for minimum adequate model (selected using AIC) examining Chatham Rise and Challenger Plateau DTIS benthic community in relation to environmental variables. Variable name abbreviations provided in Table 15.

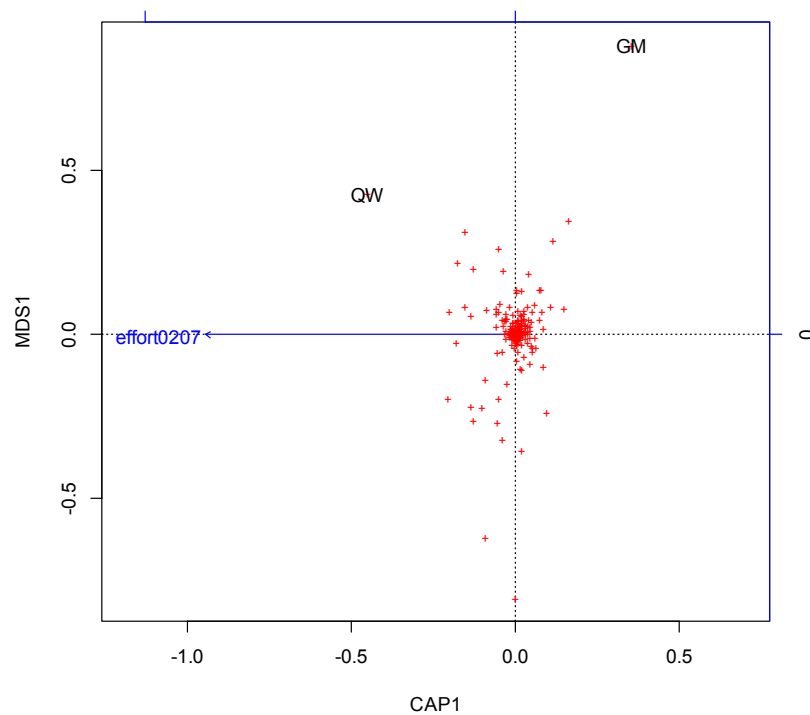


Figure 56: CAP correlation plot for effect of trawl fishing effort on Chatham Rise and Challenger Plateau DTIS benthic community, having partialled out the effects of the other terms retained in the minimum adequate model (Figure 55). The two species appearing most correlated to fishing effort (QW – Quill worm and GM – the echinoderm *Gracilechinus multidentatus*) are marked.

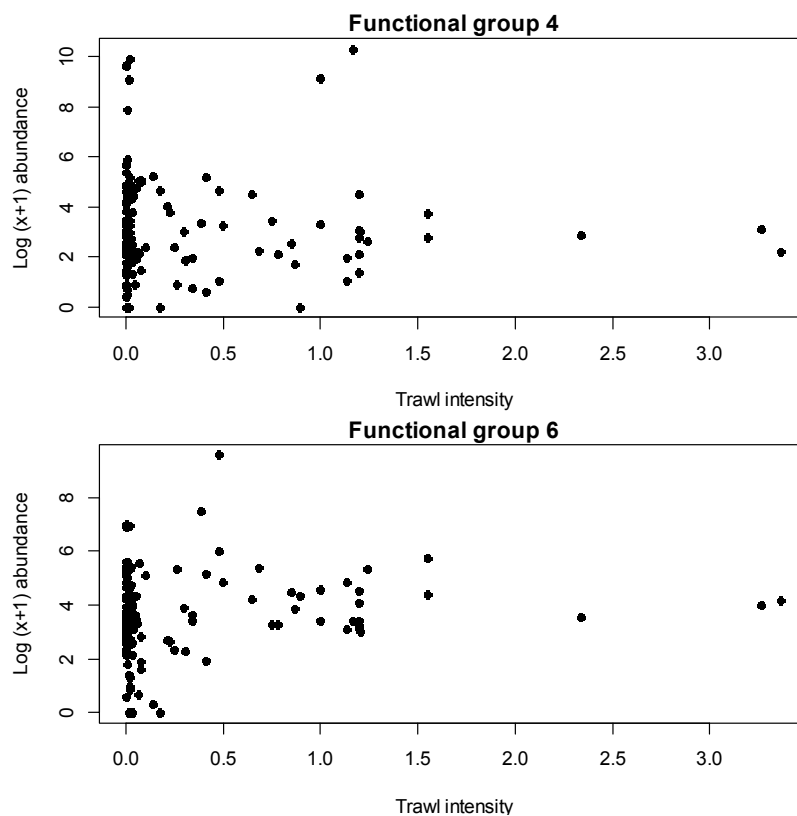


Figure 57: The abundance of Chatham Rise / Challenger Plateau DTIS epifauna (functional group 4 and 6) in relation to trawl fishing intensity (average number of times fished per year).

5.1.2 Seamount sledge and beam trawl

The benthic community data from the seamount sledge analysis comprised 873 OTUs recorded from 101 samples (70 on the Chatham Rise, 31 on the Challenger Plateau) (Figure 58) from 84 individual sites. Seventy-six samples were collected with the seamount sled, and 25 samples with the beam trawl. Epifaunal community productivity was estimated by applying previously determined Productivity / Biomass ratios (Lundquist & Pinkerton 2008; Pinkerton et al. 2008) to biomass estimates. Both biomass and productivity were examined in terms of total quantity, and total of typical benthic feeding fish prey, on the basis of Stevens et al. (2011).

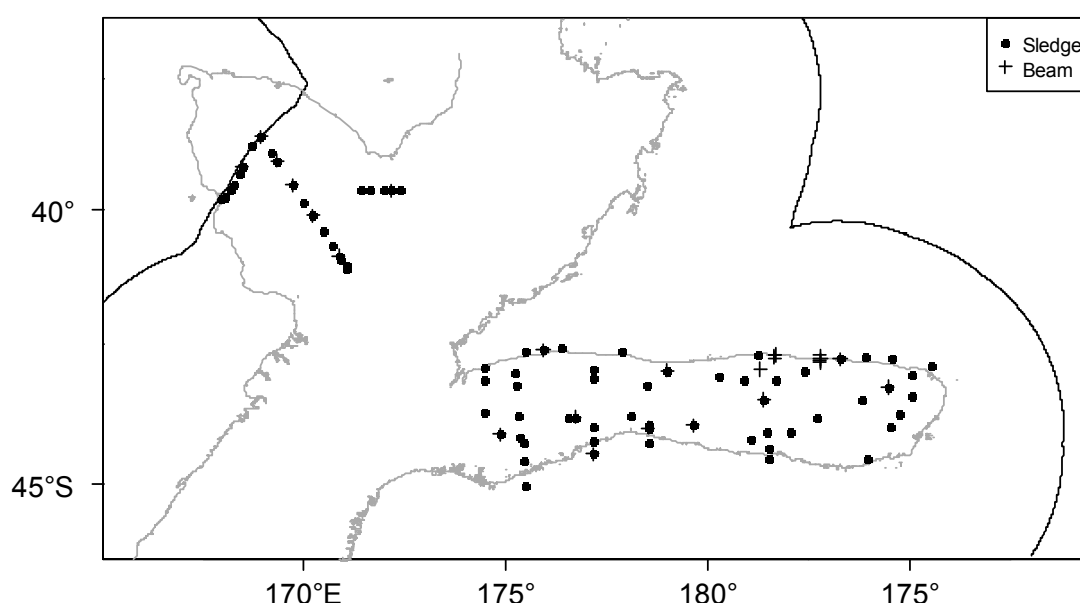


Figure 58: Chatham Rise and Challenger Plateau Ocean Survey 20/20 station locations included in seamount sledge analysis. 1000 m depth contour and limit of EEZ also shown.

The Bray Curtis similarity matrix of the square root transformed, standardised community data was analysed in relation to the environmental variables with DistLM, using backwards selection based on the AIC criterion. Explanatory variables included in the analysis are listed in Table 15. DistLM marginal tests for each variable are provided in Table 17, and none of the terms explained more than about 5% of the variance in the community composition individually.

The backward model selection procedure retained terms for location, sea surface temperature anomaly, seabed temperature, seabed temperature residual, sea surface productivity, dissolved organic matter, suspended particulate matter, depth, percentage mud, sampling gear and area, explaining 28.7% of the variance in the sledge and beam trawl community data (Figure 59). Fishing effort explained 2% of the variance when fitted individually (marginal test), but was not retained as an explanatory variable. Similar analyses were conducted for univariate diversity measures (number of taxa, Shannon-Weiner diversity and Pielou's evenness), and biomass and productivity estimates. For all except Pielou's evenness, fishing effort explained 2–3% of the variance when fitted individually (marginal test), but was not retained as an explanatory variable once environmental variables were available to the models. Fishing effort explained less than 0.5% of the variance in Pielou's evenness, and was not retained in the model, even when this was the only term offered. The effects of fishing pressure and environmental variables on biomass and productivity were also examined within a generalised linear modelling framework. Using a forward stepwise model selection approach, and retaining terms increasing AIC, fishing effort was not retained in any of the models.

Table 17: DistLM marginal tests for terms retained in the analysis of the Chatham Rise and Challenger Plateau epibenthic community composition. Prop represents the proportion of the variance explained by each term when fitted individually. Variable name abbreviations provided in Table 15.

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df
lon	16 887	4.048	0.001	0.039	99	2
lat	20 342	4.917	0.001	0.047	99	2
tempres	17 215	4.130	0.001	0.040	99	2
sstanom	8423.2	1.979	0.001	0.020	99	2
bedtemp	18 331	4.410	0.001	0.043	99	2
vgpm	12 372	2.934	0.001	0.029	99	2
disorgm	16 964	4.067	0.001	0.039	99	2
suspart	18 041	4.337	0.001	0.042	99	2
bathy	16 483	3.947	0.001	0.038	99	2
mud	12 522	2.970	0.001	0.029	99	2
gear	12 496	2.964	0.001	0.029	99	2
area	20 303	4.908	0.001	0.047	99	2

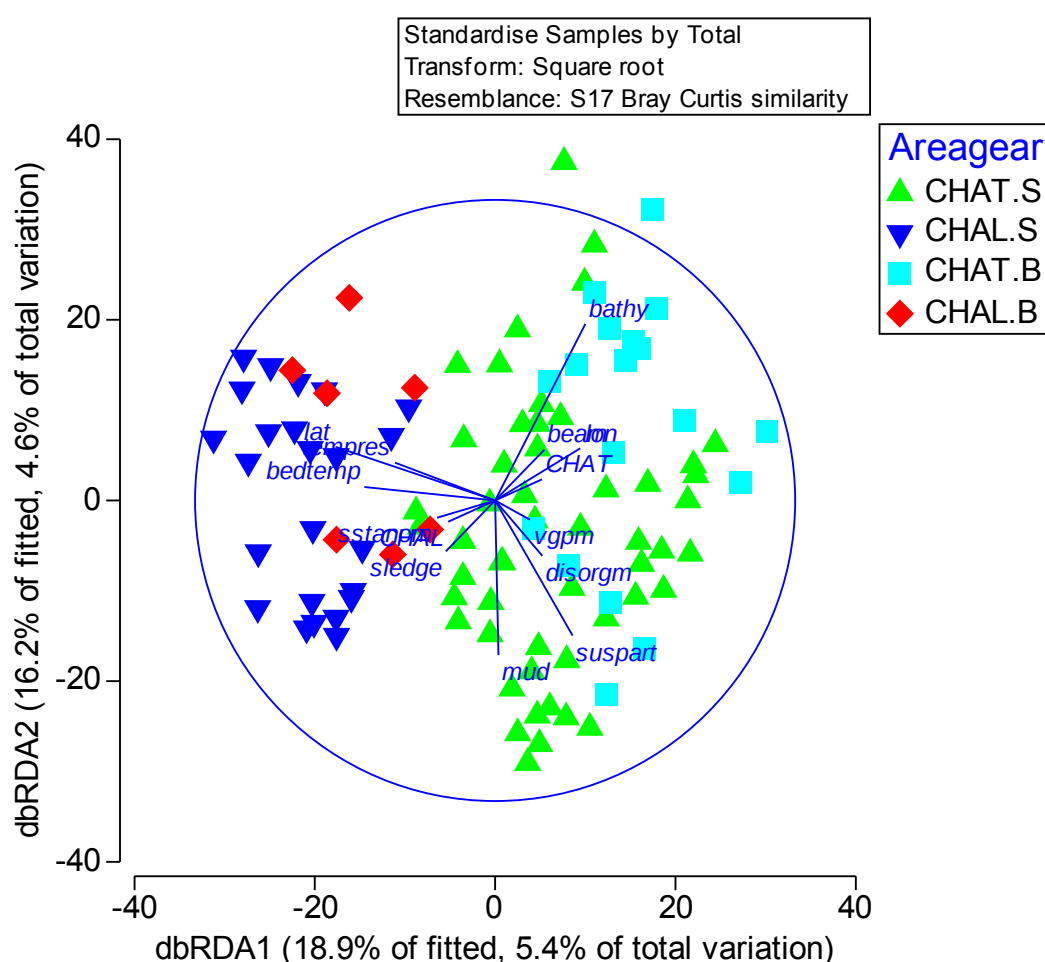


Figure 59: Distance-based redundancy analysis showing the position of sites and environmental drivers selected as best at explaining between site dissimilarities in the Chatham Rise and Challenger Plateau epibenthic communities. Sites labelled by region (CHAT – Chatham Rise; CHAL – Challenger Plateau) and sampling method (S – seamount sledge; B – beam trawl). Variable name abbreviations provided in Table 15.

The analysis was repeated using CAP (not shown), with terms retained for latitude and longitude, sea surface temperature anomaly, seabed temperature and seabed temperature residual, sea surface productivity, dissolved organic matter, suspended particulate matter, depth, percentage mud and area. As with the DistLM analysis, fishing effort was not retained in the final model.

5.2 Opportunistic analysis of survey bycatch data

Annual trawl surveys have been conducted from the *RV Tangaroa* on the Chatham Rise since the early 1990s, targeting hoki, hake and ling. All catch is recorded on a tow by tow basis, and as taxonomic guides have become more available, the resolution to which benthic bycatch species have been identified has improved. The most significant improvement in taxonomic identification was associated with the development of the Ministry of Fisheries guide to deepsea invertebrates (Tracey et al. 2011a) which was first published in the early 2000s, and surveys after 2006 are considered to have applied these guides consistently (N. Bagley, *pers. comm.*). Since fishing effort data layers were only available to the end of the 2010/11 fishing year (Black et al. 2013), only data for surveys from 2007 to 2012 (inclusive) were examined. Environmental variables were taken from 1 km² grid layers prepared for previous EEZ scale analysis (Baird et al. 2013). Combining the large number of stations available from these surveys provided a broader range of fishing effort to examine benthic communities (averaged over the five years prior to the surveys, sampled cells were fished from 0 – 6 times per year).

The benthic community data from the trawl survey analysis comprised 204 OTUs recorded from 580 stations (between 104 and 133 stations per year) (Figure 60), collected between 2008 and 2012. The numbers of OTUs recorded per station varied considerably (between 1 and 27). The recording and identification of benthic bycatch is considered to have been undertaken consistently since 2006. This variability is therefore assumed to relate to variability in the benthic community and the ability of the sampling gear (demersal otter trawls) to collect a sample. It is acknowledged that benthic organism catchability is likely to be poor, given the trawl and footrope used on this survey gear (60 mm net mesh, 350 mm bobbins on the footrope)(Hurst & Bagley 1994).

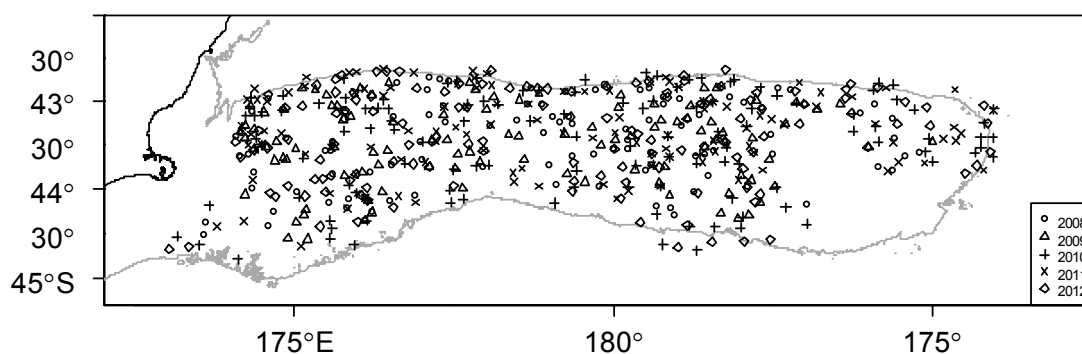


Figure 60: Chatham Rise research trawl survey station locations from 2008–2012 included in trawl survey bycatch analysis. 1000 m depth contour also shown.

Preliminary examination of the benthic community data identified a number of stations where very few species were recorded, and these stations showed as clear outliers in MDS plots. Excluding stations that recorded fewer than five OTUs (225 stations excluded, 39% of the total) improved the spread across the ordination space, but increased the plot stress (Figure 61). Exclusion of these stations did not change the distribution of effort values over which the analysis was conducted.

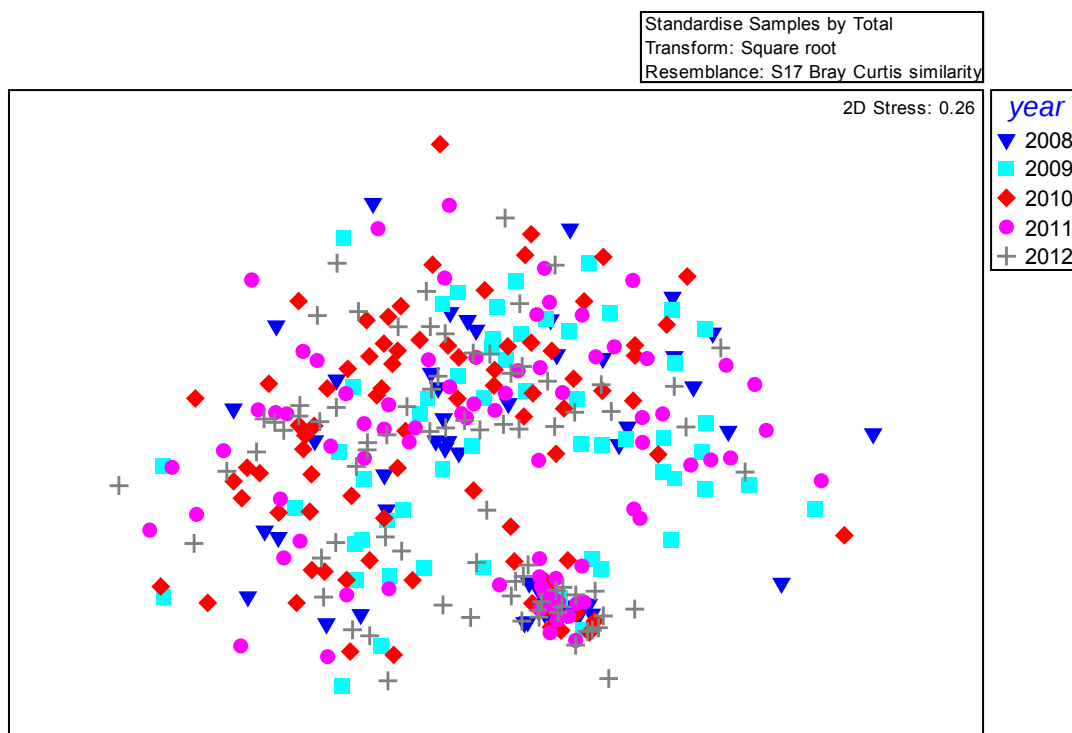


Figure 61: Non-metric multidimensional scaling analysis of the benthic communities (excluding stations where fewer than five species were recorded) sampled from the Chatham Rise trawl bycatch data. Sites that are closest together are most similar.

The Bray Curtis similarity matrix of the square root transformed, standardised community data was analysed in relation to the environmental variables with DistLM, using backwards selection based on the AIC criterion. Explanatory variables included in the analysis were taken from data layers collated by Leathwick et al. (2012) and Baird et al. (2013) are listed in Table 18. Fishing intensity data were from Black et al. (2013).

Backwards model selection using DistLM only removed terms for seabed roughness, primary productivity (VGPM), orbital velocity at the seabed and suspended particulate matter, leaving the other variables to explain 23.5 % of community variance (Table 19, Figure 62). Fishing effort at a 5 km scale explained 0.7% of the variance. Reducing the overall variability in the dataset (by excluding stations with only one, or fewer than five OTUs recorded) resulted in a greater proportion of the overall variability being explained by the retained environmental variables (outputs not presented). Retained variables were very similar (for the model excluding stations with fewer than five OTUs, terms for dynamic topography, dissolved organic matter sea surface temperature gradient and sediment type excluded from final model, instead of suspended particulate matter and primary production). While the fishing effort term was retained for all models, the proportion of total variance accounted for by the fishing effort term was consistently low (less than 1%).

Table 18: Environmental variables available for use in the statistical analysis, and abbreviations used in later plots and tables for Chatham Rise trawl survey bycatch analysis.

Data	Metric	Abbreviation
Fishing effort	Average fishing intensity (5 years prior to sampling)	effort
Broader scale environmental data derived from models and maps	Seabed roughness, ratio of real surface area to flat surface area, estimated from bathymetry	rough
	Seabed slope, estimated from bathymetry	slope
	Sea surface productivity	vgpm
	Sea surface temperature residual	SSTres
	Winter sea surface temperature	SSTwin
	Bottom water temperature residual	Btres
	Sea surface temperature anomaly amplitude	sstanom
	Sea surface temperature gradient	SSTgrad
	Mean sea surface solar radiation	radmn
	Sea floor salinity	sal
	Suspended particulate matter	sspm
	Particulate organic carbon	POC
	Dynamic oceanography, mean sea surface above geoid.	dynoc
	Seabed orbital velocity	orbvel
	Depth averaged maximum tidal current	current
	Mixed layer depth	mld
	Bottom phosphate	phos
	Bottom nitrate	nitrate
	Depth	depth
	Apparent oxygen utilisation	aou
	Dissolved oxygen at depth	do
	Dissolved organic matter	disorgm
	Sediment type	sed
Sampling details	Survey year	year
Spatial correlates	Latitude	lat
	Longitude	lon

Table 19: DistLM marginal tests for analysis of the Chatham Rise trawl bycatch composition (only terms retained in the final model are presented). Prop represents the proportion of the variance explained by each term when fitted individually. Variable name abbreviations provided in Table 18.

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df
lon	27 505	6.521	0.001	0.011	578	2
lat	62 232	14.968	0.001	0.025	578	2
effort	18 404	4.347	0.001	0.007	578	2
SSTres	83 323	20.218	0.001	0.034	578	2
slope	42 203	10.066	0.001	0.017	578	2
POC	78 923	19.114	0.001	0.032	578	2
dynoc	43 012	10.263	0.001	0.017	578	2
sal	80 707	19.561	0.001	0.033	578	2
Btres	128 740	31.844	0.001	0.052	578	2
phos	119 160	29.356	0.001	0.048	578	2
nitrate	112 180	27.554	0.001	0.046	578	2
depth	125 060	30.887	0.001	0.051	578	2
aou	128 780	31.856	0.001	0.052	578	2
tide	85 136	20.673	0.001	0.035	578	2
sstanom	75 455	18.248	0.001	0.031	578	2
radmn	25 851	6.125	0.001	0.010	578	2
do	45 258	10.809	0.001	0.018	578	2
mld	47 387	11.327	0.001	0.019	578	2
SSTwin	69 485	16.762	0.001	0.028	578	2
year	39 100	2.317	0.001	0.016	575	5
sed	63 095	3.775	0.001	0.026	575	5

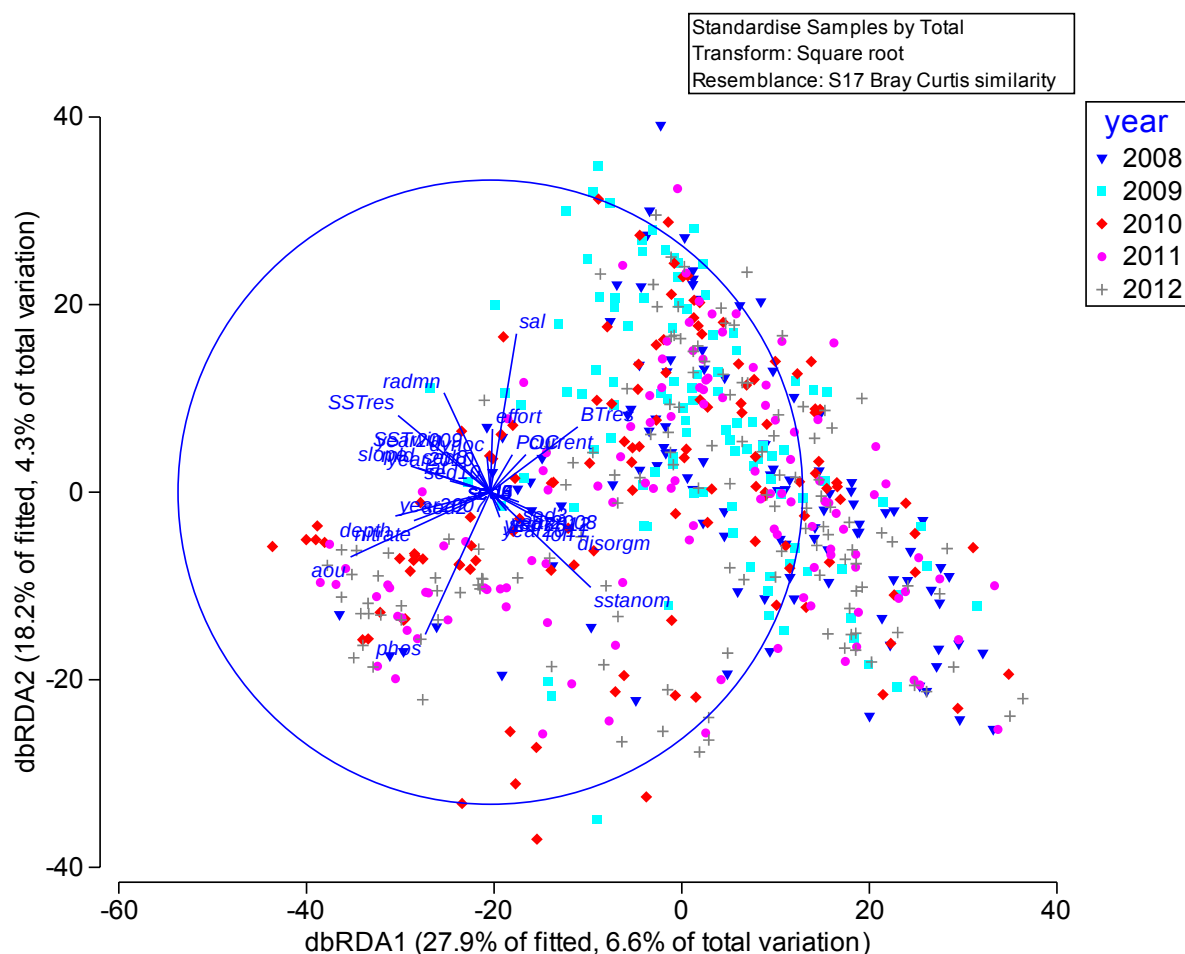


Figure 62: Distance-based redundancy analysis showing the position of stations and environmental drivers selected as best at explaining between stations dissimilarities in the Chatham Rise trawl bycatch benthic communities in two dimensions. Variable name abbreviations provided in Table 18.

The analysis was repeated using CAP (with backwards stepwise model selection using AIC, on the standardised, square-root transformed community data) (Figure 63), with terms retained for latitude and longitude, year, seabed slope, winter sea surface temperature and anomaly, bottom water salinity and temperature residual, dissolved organic matter, mixed layer depth, bottom phosphate and nitrate, apparent oxygen utilisation, dynamic oceanography, mean tidal current and fishing effort, explaining 17.5 % of the variance in the benthic bycatch community data. Excluding effort from this model reduced the explained variance to 17.0% of the total. Examining the effects of fishing effort on individual species (partialling out the effects of the other terms) did not suggest that any species were strongly correlated with fishing effort (as expected, given the low proportion of the overall variance accounted for by the term).

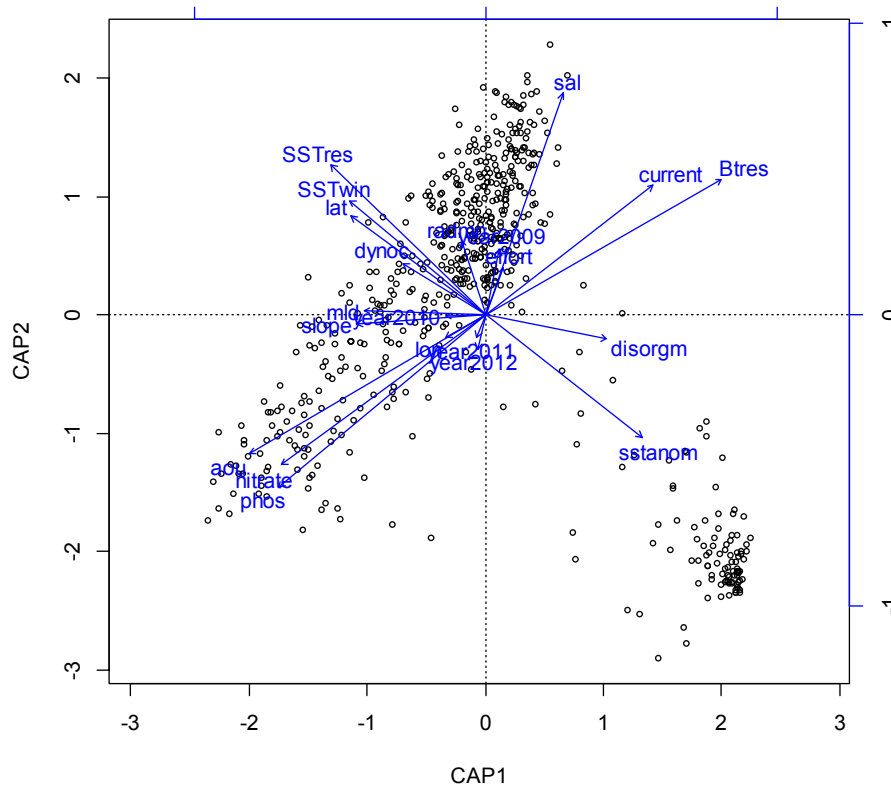


Figure 63: CAP correlation plot for minimum adequate model (selected using AIC) examining the Chatham Rise trawl survey benthic bycatch community in relation to environmental variables. Variable name abbreviations provided in Table 18.

5.3 Chatham Rise and Challenger Plateau overall conclusions

While the Chatham Rise and Challenger Plateau study included dedicated sampling of the benthic communities, the sampling design for the Ocean Survey 20/20 voyages was largely based around addressing objectives related to the examination of seabed habitats and biological diversity, and the analysis in relation to gradients of fishing pressure is somewhat opportunistic. Site selection for the Ocean Survey 20/20 study was largely based on the New Zealand Marine Environment Classification (Snelder et al. 2004), and the fishing effort gradients available across the study sites examined are relatively weak. The analysis of the trawl survey bycatch data was conducted over a larger number of stations and a slightly larger fishing effort gradient, but found similar results. The overall community variability explained by environmental variables was moderate to low (23 – 35%), and the variance explained by fishing was very low (1 – 2%, not retained as a significant term for the beam trawl and dredge data).

While overall effects of fishing appeared to be weak, there was some evidence of both positive and negative correlations of individual species abundances with fishing, which fit with expectations in relation to life histories. However, the examination of abundances at the functional group level showed only weak evidence of a fishing effect.

6. SCAMPI FISHERIES

Scampi (*Metanephrops challengeri*) is a mud burrowing decapod, and as such is only available to trawl sampling when emerged from a burrow. While trawl sampling is used to provide fishery independent abundance indices, concerns over variability in trawl catchability in relation to patterns in burrow emergence have led to the development of photographic surveys to provide counts of burrow numbers as indices of population abundance (Tuck et al. 2015b). Therefore, for the main scampi grounds, both trawl and photographic surveys provide opportunistic approaches to sample benthic communities. Photographs are taken by a still camera maintained at an altitude of 3 – 4 m above the seabed, providing high resolution images of burrows or epifauna, with the footprint of each image estimated from the distance between two parallel scaling lasers.

Analysis of opportunistically collected data on epifaunal community composition (from trawl and/or photographic surveys) across a number of scampi fisheries (muddy habitats, 250 – 500 m depth) are described here, including previously published research and additional work conducted within this project.

6.1 Bay of Plenty (SCI 1) and Hawke Bay / Wairarapa (SCI 2)

6.1.1 Trawl survey bycatch

Shallower areas of the Bay of Plenty (SCI 1) have been trawled for tarakihi since the 1930s, but the 1980s saw an expansion of effort into deeper waters (down to about 600 m), targeting gemfish, hoki and scampi, with scampi trawling being the predominant activity in the area. Cryer et al. (2002) examined patterns in the invertebrate catch composition from a scampi targeting trawl survey (200 – 600 m) in relation to historical patterns in fishing effort (for the four main target species), latitude, longitude and depth. Across the study sites, the estimated gradient of total fishing effort (all demersal tows combined) ranged from zero to over 1000 tows estimated to have passed within 200 m of a sampled station (between 1989 and the study, late 1996 and early 1998).

Summarising the Cryer et al. (2002) study, trawl survey tows were conducted in a standardised manner, with all benthic invertebrate catch from the 66 stations identified to the lowest possible taxonomic level. Taxonomic identification was conducted by a benthic invertebrate specialist, and resulted in the benthic community being assigned to 163 OTUs, considerably more than would typically be recorded in an opportunistic trawl survey benthic bycatch data set. Overall benthic community composition, and univariate community measures (diversity and species richness) were examined in relation to fishing pressure gradients and environmental drivers, using ordination and generalised linear modelling approaches. Within the study area, depth and trawling for scampi were considered the dominant influences on benthic community structure (with trawling for gemfish, hoki and tarakihi having lesser effects), with overall fishing effort accounting for 11 – 40% of the variability in benthic community structure. Species richness and species diversity were both also negatively correlated with scampi trawl effort. A number of species were found to be significantly correlated with trawling for scampi. These included gastropod, sea urchin, starfish, brittle star, prawn, slipper lobster, crab, hermit crab and holothurian species that were negatively correlated with scampi trawling, and prawn, starfish, squat lobster, sponge and anemone species that were positively correlated. Scampi abundance was also positively correlated with scampi trawling, as might be expected, given that this is the target species of the fishery.

Similar bycatch data were recorded from a less intensive programme of seasonal sampling (33 stations over 6 seasonal sampling events, 1999–2000) in SCI 2 (Wairarapa), but as each of these sampling events was in the same general region (as the objective was to examine seasonal patterns), there is relatively little gradient in fishing effort over which to examine the data (aggregate effort ranging up to being fished just over twice per year). This and the Cryer et al. (2002) study were unique amongst the

opportunistic bycatch data, in that taxonomic identification was undertaken by a recognised expert and was typically to species level, rather than survey staff using identification guides, with more species identified to groups.

Preliminary examination of the Cryer SCI 2 data (CAP on Bray-Curtis dissimilarity matrix of standardised, square-root transformed community composition; Figure 64) suggested that the effects of season and depth had a far greater contribution to explaining the variability in the benthic community than fishing effort (estimated from Black et al. 2013), and only depth was retained by a stepwise model selection process using AIC as the model selection criterion, accounting for 29.4 % of the total community variance.

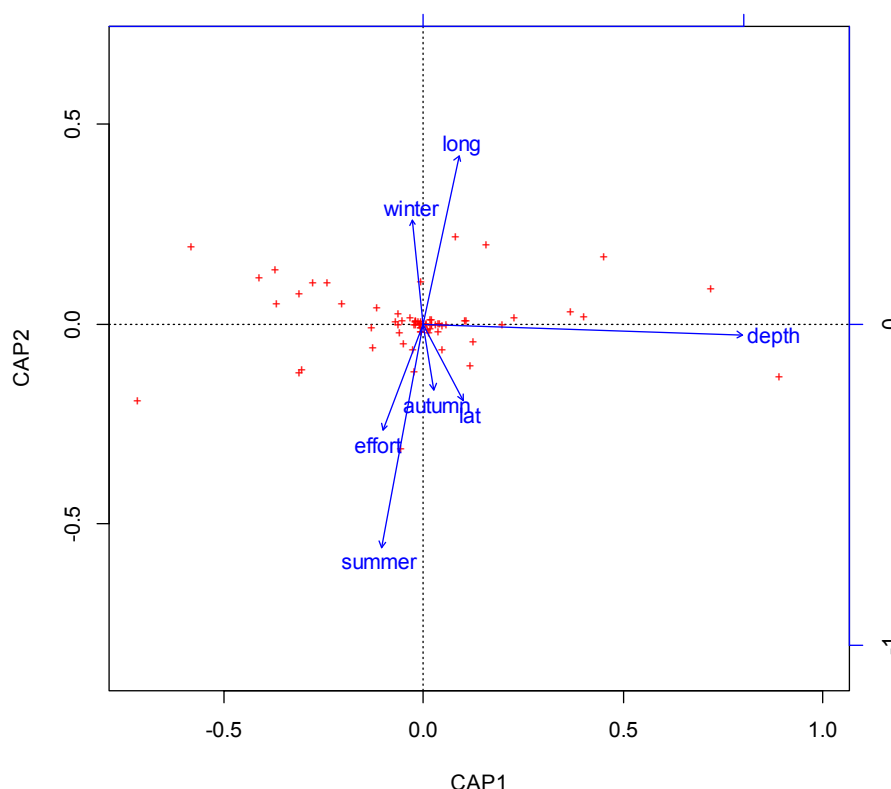


Figure 64: Ordination of stations generated by Constrained Analysis of Principal Coordinates of SCI 2 trawl bycatch data. Arrows represent directions and approximate relative magnitudes of effects of explanatory variables in the preliminary (full) model. Term abbreviations, depth - depth; lat – latitude; long – longitude; effort – average fishing intensity over previous 5 years; spring, summer, autumn, winter – season of sampling.

To rule out taxonomic expertise as a driver of results, a similar analysis was conducted on more recent scampi trawl survey benthic community bycatch data, using the data recorded routinely through the survey (as opposed to identification by a recognised expert). Trawl survey data were collated across the SCI 1 and SCI 2 areas over the two most recent surveys for each area to provide a benthic community dataset of 61 OTUs recorded over 55 stations. The initial DistLM minimum adequate model (including terms for latitude, depth, year and fishing effort in the previous year) accounted for 40.9% of the total community variance (Figure 65), and no further effort terms improved this. The fishing effort term accounted for 3.6% of the total community variance. Partialling out the effects of the other retained variables (Figure 66), scampi abundance was positively correlated with fishing effort, while the sea pens Pennatulacea were negatively correlated. This appears to be driven by the SCI 2 data, as when examined individually, fishing effort was not retained in the minimum adequate model for SCI 1 (within

which latitude, year and depth accounted for 39.8% of the total variation), but fishing effort accounted for 15.6% of total variability for SCI 2.

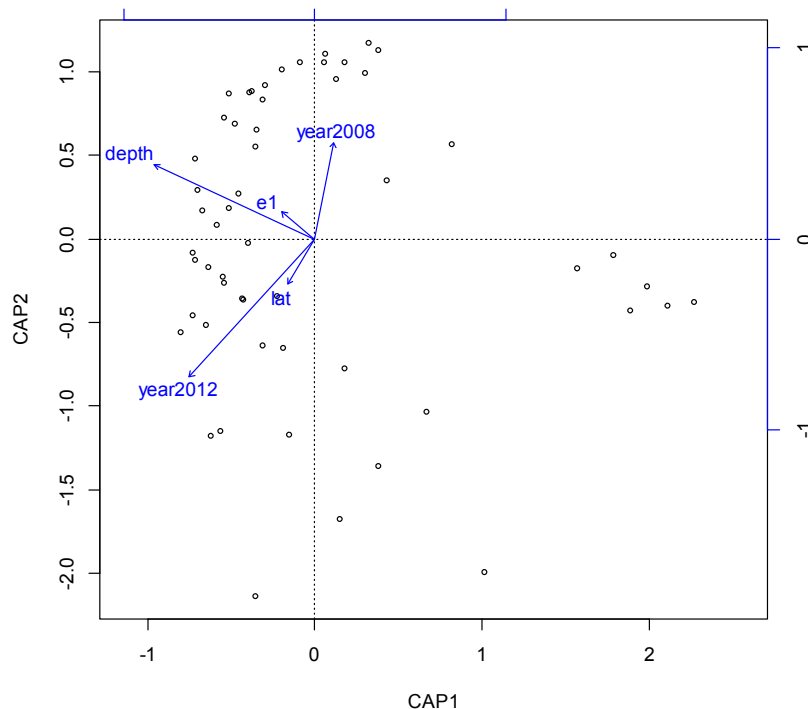


Figure 65: Ordination of stations generated by redundancy analysis of SCI 1 and SCI 2 trawl survey benthic epifauna data. Arrows represent directions and approximate relative magnitudes of effects of explanatory variables in the final model. Term abbreviations, depth - depth; lat – latitude; e1 –fishing intensity in the year prior to sampling; year – sampling year.

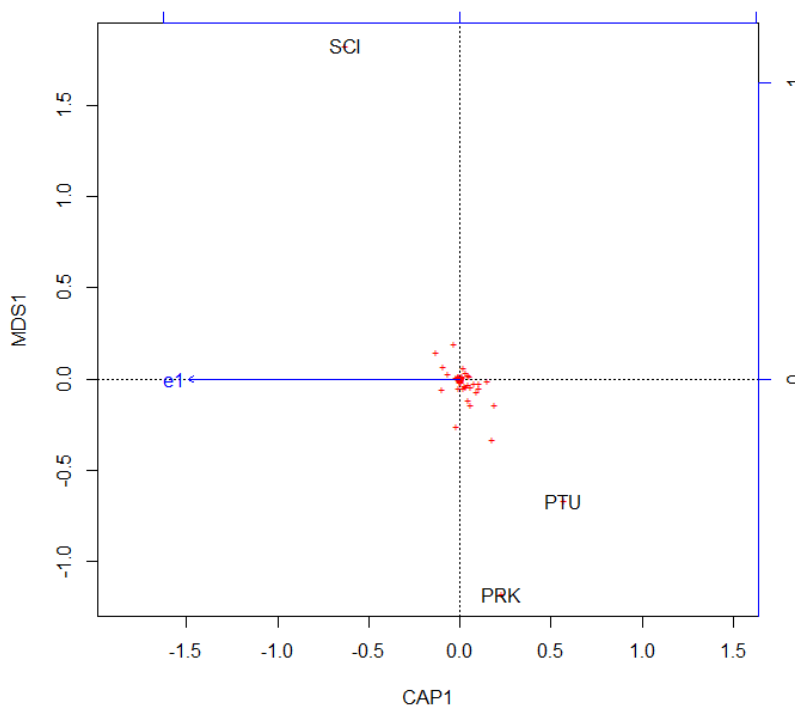


Figure 66: Ordination of species generated by redundancy analysis of SCI 1 and SCI 2 trawl survey benthic epifauna data (Figure 65), examining the effect of fishing effort having partialled out the effects of other variables. Species abbreviations: SCI – scampi, PTU – Pennatulae, PRK – Prawn killer.

6.1.2 Photographic survey data

Images from the 2012 surveys of SCI 1 and SCI 2 (Tuck et al. 2012) were examined, with all visible epifauna identified to the highest level images allowed. Data from images were combined to provide a station community sample (density of each species) for each of the 100 photographic stations examined across the two surveys. Thirty eight OTUs were identified from the images.

Species densities were standardised to station totals, and square root transformed prior to analysis of the variability in community composition. Fishing effort data for each station location were extracted from GIS shapefiles (Black et al. 2013) and included as individual years. Since 1989, aggregate effort across the study sites ranges between 0 and 80 times fished, and in the year prior to the survey, ranged from 0 to 9 times fished (although only one station in SCI 2 reached this level, with all other stations ranging between 0 and 3 times fished). Effort for the most recent year was included in the initial DistLM model (on the basis that fishing effort in recent years is more likely to have influenced current benthic community patterns than more historic effort patterns, although this does of course depend on the relative change in effort gradient over time) along with latitude, longitude and depth variables, from which a minimum adequate model was derived by a stepwise model selection process using AIC as the model selection criterion. This minimum adequate model retained terms for depth, latitude and fishing effort (Figure 67), confirming that recent fishing effort patterns (2011) contributed significantly to explaining the benthic community composition observed in the 2012 photographic survey, although the overall explanatory power of the model was low (11.5%), with fishing effort accounting for 3.3% of the variance (2.9% uncorrelated with other variables). The addition of a term for the previous fishing effort pattern (2010) did not reduce the model AIC, suggesting that older effort patterns did not have a detectable effect on the observed community. Having partialled out the other terms to examine the effect of fishing effort on the species ordination, none of the individual species appear to be particularly sensitive (Figure 68), although *Metanephrops* abundance was positively correlated with fishing effort, while Lithothid crabs and Pennatulaceae were negatively correlated. The exclusion of the target species from the benthic community data had minimal effect of the analysis.

Examining the two areas individually, the same terms were retained when analysing SCI 1, with the minimum adequate model accounting for 16.6% of the total variance, and fishing in the most recent year accounting for 3.9% of the total variance (3.8% uncorrelated with other variables). For SCI 2, more recent fishing effort was not retained in the model, but more historical effort patterns accounted for 7.2% of total variance (with the minimum adequate model including depth, longitude and effort accounting for 21.9% of total variance). When examined individually, fishing effort was negatively correlated with Pennatulaceae (sea pen) abundance (having partialled out the effects of the other retained terms).

An analysis of lower resolution taxonomic data on photographic survey derived benthic community composition from the 2008 SCI 1 survey was also conducted in conjunction with SCI 6A data, and this is described below (Section 6.2.2).

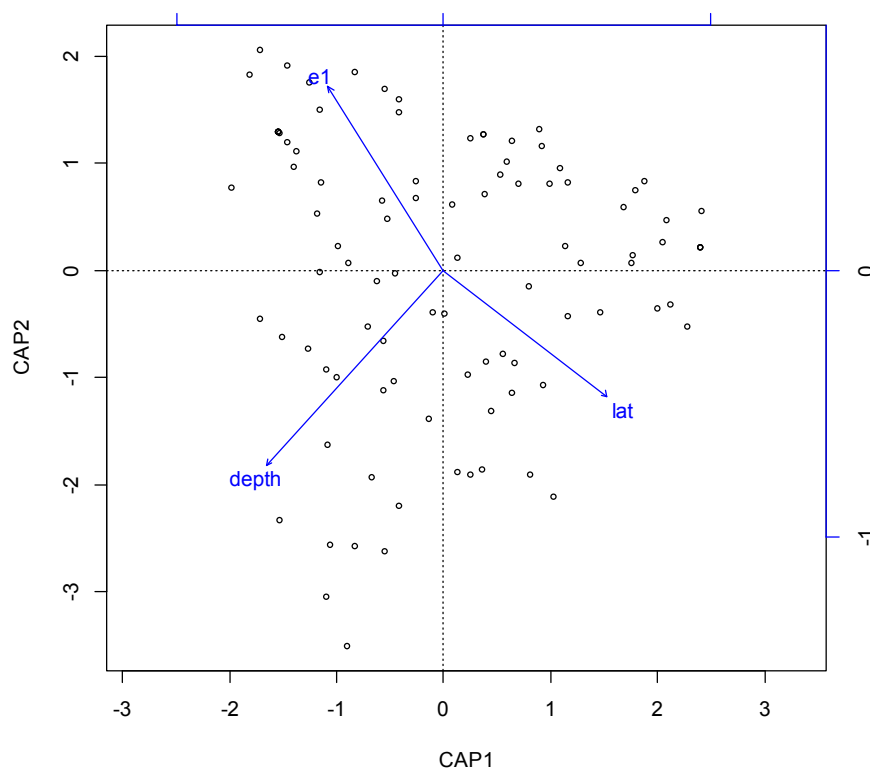


Figure 67: Ordination of stations generated by redundancy analysis of the 2012 SCI 1 and SCI 2 photo survey benthic epifauna data. Arrows represent directions and approximate relative magnitudes of effects of explanatory variables in the final model. Term abbreviations, depth - depth; lat – latitude; e1 –fishing intensity in the year prior to sampling.

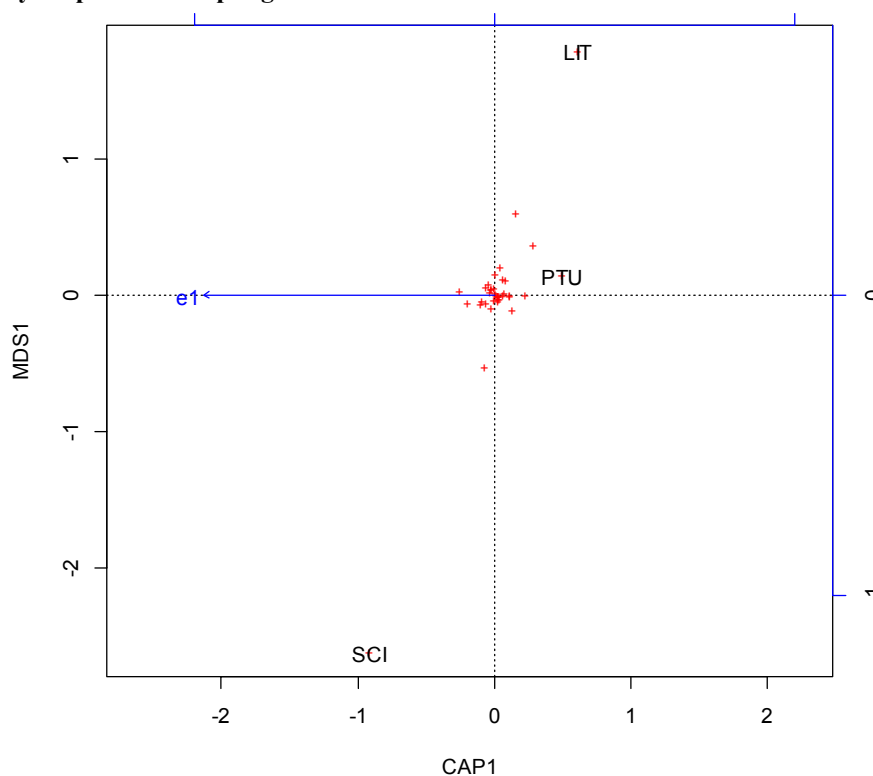


Figure 68: Ordination of species generated by redundancy analysis of SCI 1 and SCI 2 photo survey benthic epifauna data (Figure 67), examining the effect of fishing effort having partialled out the effects of other variables. Species abbreviations: SCI – scampi, PTU – Pennatulae, LIT – Lithothid crabs.

6.2 Auckland Islands (SCI 6A)

Following some exploratory fishing in 1990–91 (with 2 tonnes of scampi landed), the Auckland Islands scampi fishery expanded rapidly to land 325 tonnes in 1991–92, and has maintained an annual average landing of around 300 tonnes for much of its history, declining to an annual average of just under 200 tonnes in recent years (Tuck 2015). The scampi target fishery is focussed in the 350 – 550 m depth range, and while other fisheries operate in the region (e.g., squid SQU 6T), the scampi target fishery is the dominant demersal trawling on the scampi grounds.

6.2.1 Trawl survey bycatch

Scampi surveys were conducted annually in SCI 6A from 2007 to 2009 and again in 2013. Trawl sampling was conducted in a standardised manner, and as with the recent SCI 1 and SCI 2 analysis above, trawl survey benthic community bycatch was identified and recorded following normal survey practices. Combining all the trawl survey data from SCI 6A (four surveys) provided a benthic community dataset of 58 OTUs recorded over 69 stations. The community data were analysed in relation to fishing effort and environmental variables using CAP on a Bray-Curtis distance matrix. No fishing effort terms were retained in a minimum adequate model, with terms for latitude, longitude and year accounting for 36.3% of the total community variance (Figure 69).

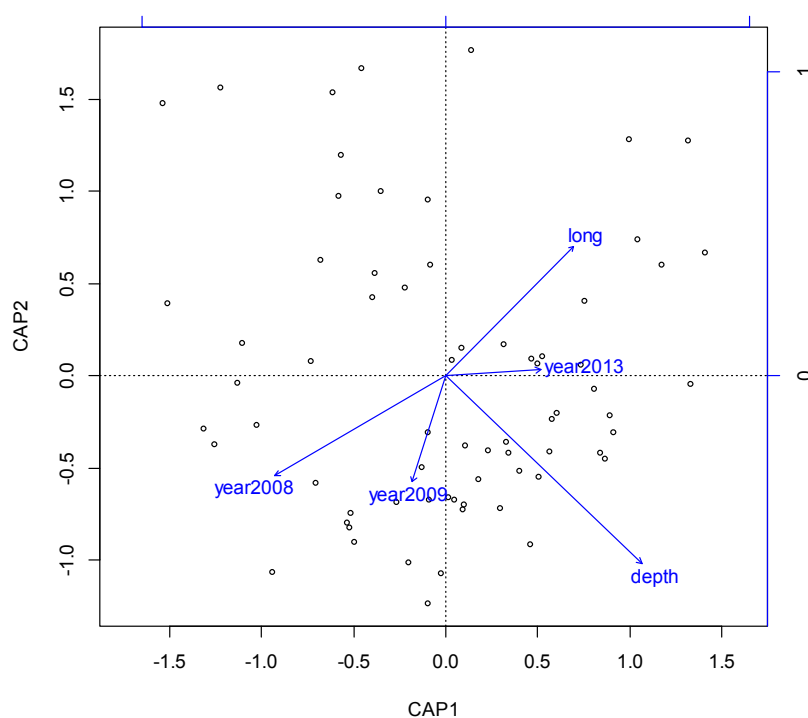


Figure 69: Ordination of stations generated by redundancy analysis of SCI 6A trawl survey benthic epifauna data. Arrows represent directions and approximate relative magnitudes of effects of explanatory variables in the final model. Term abbreviations, depth - depth; long – longitude; year effect – relative to 2007.

6.2.2 Photographic survey data

Images from the 2008 Bay of Plenty (SCI 1) and Auckland Islands (SCI 6A) surveys were also examined in the same way as that described above (Section 6.1.2), although for this analysis, all visible epifauna were identified to ten broad OTUs (sea urchins, holothurians, starfish, crinoids, cnidarians, sponges, arthropods, scampi, molluscs and bryozoans). Data from images were combined to provide a

station community sample (density of each species) for each of the 82 (39 from SCI 1, 43 from SCI 6A) photographic stations. Across the study sites in both regions, aggregate fishing effort ranged between 0 and 3 times fished per year.

Species densities were standardised to station totals, and square root transformed prior to analysis of the variability in community composition using CAP on a Bray-Curtis distance matrix. Fishing effort data for each station location were extracted from GIS shapefiles (Black et al. 2013). As the various effort metrics showed very strong positive correlations (mostly over 0.95), the total effort (aggregate fished area) over the five years preceding the survey was used. Backwards model selection using AIC retained terms for fishing effort, depth and fishery, but only accounted for 11.1% of the total variation, with 5.5% accounted for by the fishing effort term (4% uncorrelated with other variables) (Figure 70). Partialling out the effects of depth and fishery (Figure 71), scampi abundance was positively correlated with fishing effort, while holothurian abundance was negatively correlated.

Examining the two fisheries individually, a model including fishing effort, depth and longitude accounted for 25.3% of total community variation in SCI 1, with 11% accounted for by fishing effort (6% uncorrelated with other variables), while for SCI 6A, latitude, longitude and depth accounted for 19.6% of total community variation, and fishing effort was not retained.

Individual OTU densities were also examined in relation to fishing pressure (not accounting for other effects), with a number of the groups (sea urchins, holothurians, cnidarians, molluscs, bryozoans) displaying a typical factor ceiling (high variability at low fishing effort, low abundance and low variability at high fishing effort) response (Figure 72).

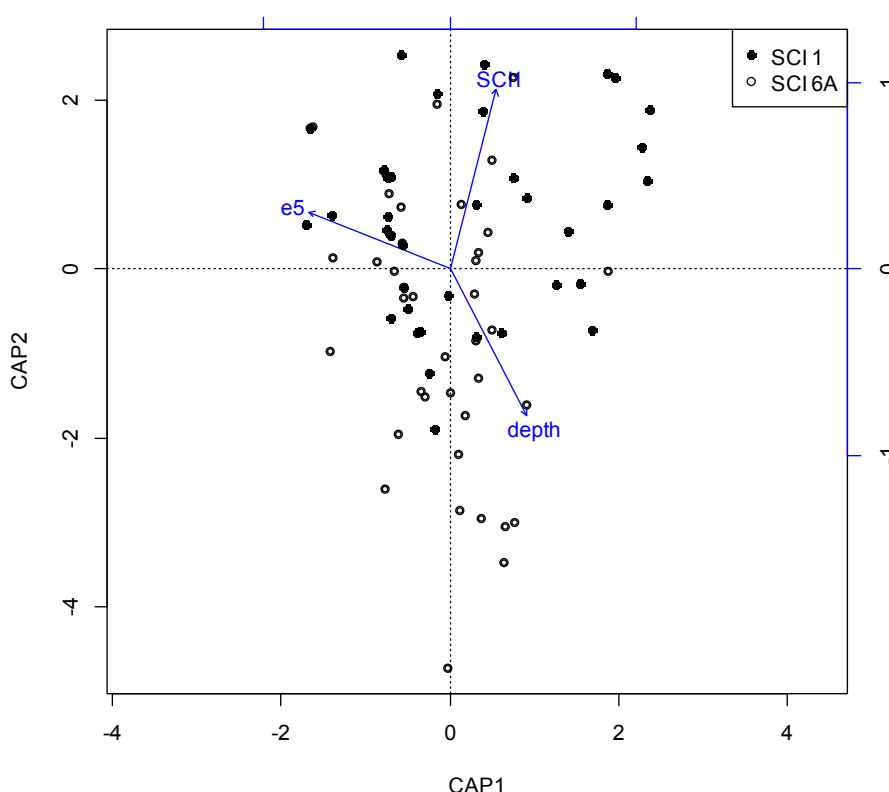


Figure 70: Ordination of stations generated by redundancy analysis of SCI 1 and SCI 6A photo survey benthic epifauna data. Arrows represent directions and approximate relative magnitudes of effects of explanatory variables in the final model. Term abbreviations, depth - depth; lat - latitude; e5 -fishing intensity averaged over the 5 years years prior to sampling, SCI1 – area effect (SCI 1 compared to SCI 6A).

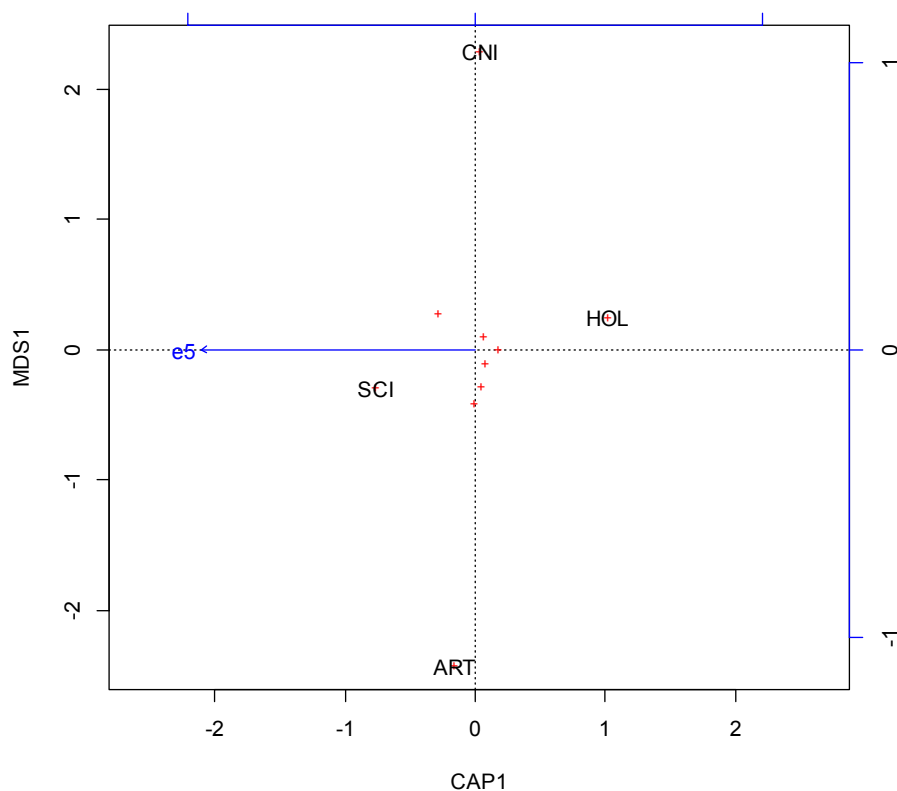


Figure 71: Ordination of species generated by redundancy analysis of SCI 1 and SCI 6A photo survey benthic epifauna data (Figure 70), examining the effect of fishing effort having partialled out the effects of other variables. Species abbreviations: SCI – scampi, HOL – Holothurians, CNI – Cnidarians, ART - Arthropods.

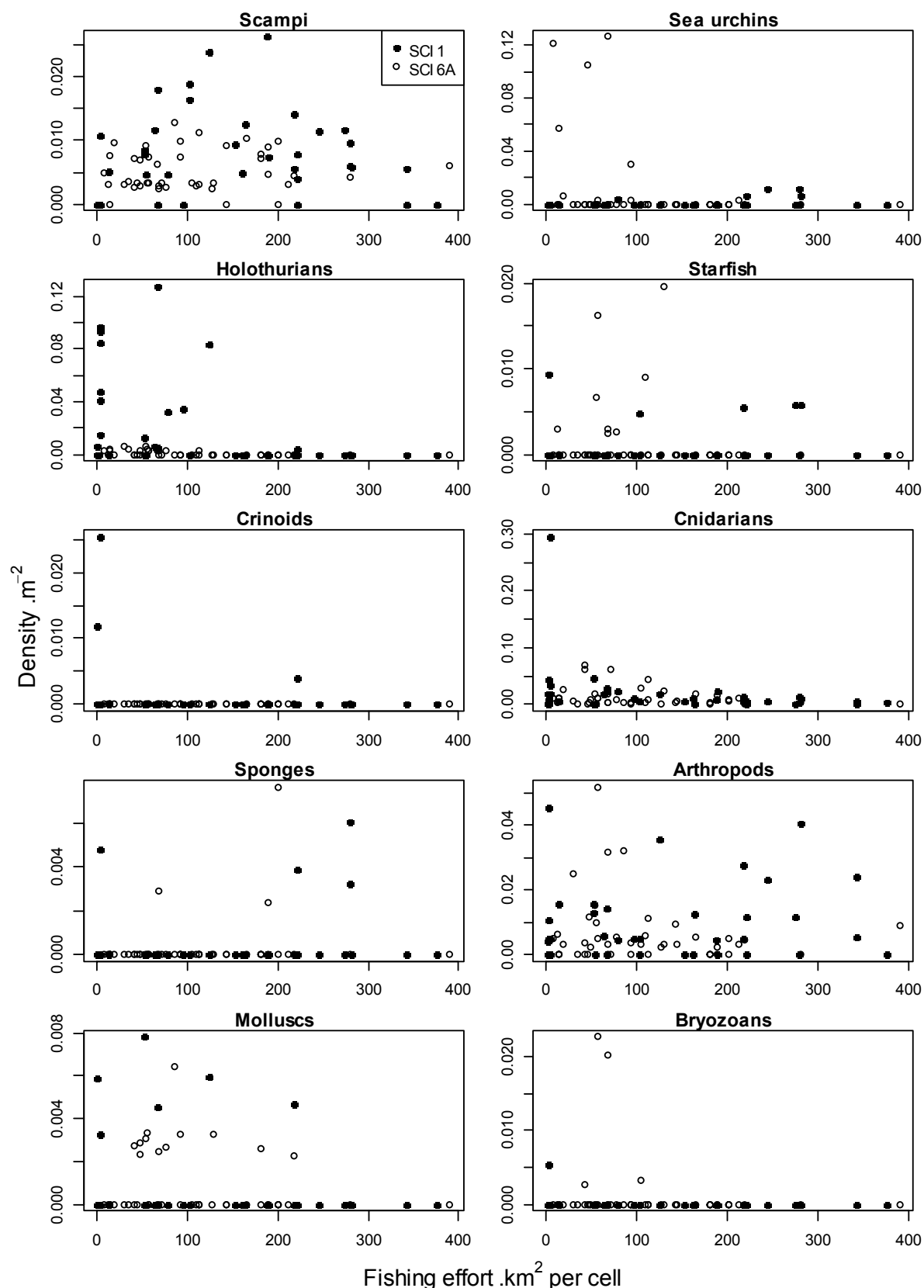


Figure 72: Density of the ten OTUs recorded from the 2008 SCI 1 and SCI 6A photographic surveys against fishing effort (aggregate effort over 5 years prior to surveys, per 25 km² cell).

6.3 Scampi fishery overall conclusions

The nature of scampi surveys (including both a trawl and photographic component) allows for the comparison of the two sampling approaches in examining the effects of fishing. However, the opportunistic nature of the data collection (random stations, stratified by depth) means that station locations are not necessarily ideal for examining gradients of fishing pressure. In fact, other than the Cryer et al. (2002) study (which included additional stations targeting high effort locations), fishing effort gradients sampled were moderate or weak. The magnitude (percentage of variance explained) of fishing effects detected declined from 11 – 40% across a strong gradient to 0 – 6% across weak gradients. There was some suggestion that trawl bycatch data was less useful in detecting fishing effects over weaker gradients, although the overall number of studies was low.

Across the various studies, a range of species were identified as being negatively correlated with fishing pressure, including whelks, sea urchins, brittle stars, sea pens and holothurians. These patterns were apparent, even in studies where no significant effect was detected on community composition.

7. OTHER PREVIOUS NEW ZEALAND STUDIES

7.1 Spirits Bay

Spirits Bay, at the northern-most tip of the North Island of New Zealand, considered to be a dynamic habitat exposed to considerable wave disturbance and strong tides, is an area of cultural significance to Maori, and also supports important commercial fisheries. Voluntary (applying only to the scallop fishery) and then regulated (applying to all mobile bottom fishing) closures were introduced in 1997 and 1999, respectively, in response to concerns over the effects of fishing on the highly unusual, sponge, bryozoan and hydroid dominated epifaunal community observed in the area.

Following a broad-scale survey of the area between North Cape and Cape Reinga in 1999 (Cryer et al. 2000), surveys focussing on a more limited area were conducted in 2006 (Tuck et al. 2010) and 2010 (Tuck & Hewitt 2013). These surveys have collected infaunal community samples through grab sampling, and data on epifaunal communities through seabed photography, sampling between 30 and 40 stations in each survey. The survey in 2006 also included acoustic mapping components, and the 2010 survey was stratified on the basis of this. The most recent analysis of both the 2006 and 2010 survey datasets is fully documented in Tuck & Hewitt (2013).

Fishing effort data for the study area were compiled from MPI data and information provided by the scallop fishing industry, and the benthic communities were examined in relation to environmental variables and fishing terms, using multivariate approaches. Individual stations ranged from not having been fished for 20 years to recent high levels of fishing effort. The analysis of both epifaunal and infaunal community data consistently identified year, habitat and depth effects, but the fishing terms (fishing intensity and years since fishing) were also found to explain a significant component of the overall variance. The models for the epifaunal communities explained more of the variance than those for the infaunal data. The combined fishing terms typically explained 15 – 30% of the variance (median 20%), which is comparable to previous studies conducted in New Zealand. Similar effects were also identified on univariate measures of the community, although for the epifaunal data, species richness appeared far less sensitive to fishing than the other measures considered (number of individuals, Pielou's evenness and Shannon-Weiner diversity index).

The main epifaunal species observed in the area were classified in terms of their sensitivity to and recoverability from different types of disturbance, on the basis of morphology and life history characteristics. The community data were examined with Canonical Correspondence Analysis (CCA) to partial out other significant effects, allowing species responses to individual fishing terms to be

identified. For both data sets, different species were found to be sensitive to the different fishing effort terms, which is predicted to reflect the different types of disturbance associated with the gears, but overall, the sponges *Halichondrida* sp. 5 and *Aaptos* sp., and the hydroids *Iophon minor* and *Nemertesia elongata* were the most sensitive. Comparison with previous epifaunal work in this region on sensitivity to fishing disturbance (Tuck et al. 2010) demonstrated that species identified as most sensitive to fishing in the present analyses had previously been categorised (on the basis of size, morphology, and other life history characteristics) as either sensitive to dredging disturbance, or moderately sensitive to dredging but growing to a medium or large individual size. Most of these species were also considered to have a poor probability of recovery following disturbance. With the exception of the hydroid *Hydrendron mirabile* (which still showed a negative relationship with fishing effort), epifaunal species classed as most vulnerable (combining sensitivity to disturbance, ability to recover and likelihood of disturbance) to the effects of fishing were rarely observed at sites that had been fished in the preceding three years.

7.2 Hauraki Gulf region

The Hauraki Gulf is an important inshore fishery area, with valuable trawl, Danish seine and long line fisheries focussing on snapper (*Pagrus auratus*), but also catching a mix of other demersal species, and the area also supports an important scallop dredge fishery. Three studies have examined the effects of fishing on the benthic communities in the area.

Thrush et al. (1995) conducted a small scale, short term (up to three month) experiment that focussed on the effects of scallop dredging at the scale of the individual dredge track, and was conducted across two shallow (24 m) sites with different fishing histories (one site regularly commercially fished, the other not commercially fished – informed by anecdotal knowledge), using a BACI design, and analysed with ANOVA and ANOSIM (Clarke & Green 1988). Community composition differed between the sites, but both were dominated by small and short-lived species. The density of common infaunal species, total abundance and species richness at each site decreased as a result of the dredging, with some species still significantly different after three months. Significant differences in community assemblage structure between the dredge and control plots were also recorded over the experiment. The effects were more pronounced at the site that had not previously been commercially fished. The bivalve *Nucula nitidula* and tube building polychaetes were consistently sensitive to the effects of fishing, showing significant reductions in abundance at both sites following dredging.

Thrush et al. (1998) examined benthic communities from 18 locations within the area using a combination of video (for epifauna) and grab, suction dredge and core (for infauna) approaches. The study was conducted prior to the routine use of fine scale catch and effort reporting in fisheries statistics for inshore fisheries, and so the benthic communities were examined across gradients of fishing pressure (and environmental variables) on the basis of rankings of potential habitat disturbance by commercial demersal trawling and dredging, estimated from fisheries legislation (unrestricted access for vessels under 20 m, trawling prohibited, trawling and Danish seine prohibited, commercial fishing prohibited, Marine Park) and anecdotal information from fishery managers and scallop fishers.

A combination of ordination and generalised linear modelling approaches were employed to analyse the data. The fishing pressure gradient accounted for 15 – 20% of the variability in benthic community structure, and also had a significant effect on species richness and benthic community diversity. Increases in fishing pressure significantly reduced the density of large (and long lived) epifauna and echinoderms, and significantly increased the density of small opportunist species, with the effect on deposit feeders varying with the sampling approach. Results were consistent with *a priori* predictions developed on the basis of short term experimental studies documented in the literature. No effect on scavengers was observed, and while the attraction of scavengers to disturbed areas to feed on damaged fauna has commonly been observed in manipulative studies (e.g., Kaiser & Spencer 1994; Ramsay et al. 1996), such effects are likely to be very transient in nature, and unlikely to be observed in broad scale studies.

Morrison et al. (2016) used video transects to examine the distribution and abundance of benthic epifauna and fish species in five areas inside and up to 2.5 km outside the Hauraki Gulf Cable Protection Zone (CPZ), which is considered to have been an effective closed area to fishing and anchoring since 1999. Within the analysis, fishing was considered as a two level factor (inside or outside the CPZ), and individual species abundances and community composition were examined in relation to environmental drivers and CPZ status using generalised linear modelling and distance based linear modelling (Anderson 2001; McArdle & Anderson 2001). CPZ status (inside or outside) had a significant effect on common species abundances and univariate community diversity measures, in the main drivers of community composition and species abundance appeared to be location and depth, with CPZ status only explaining 1.4% of total variance. There was no discernible effect of the CPZ on fish assemblages.

8. SUMMARY OF NEW ZEALAND STUDIES, AND THE IMPLICATIONS OF THE EFFECTS OF FISHING

Study approaches

It is widely acknowledged that demersal fishing has an impact on benthic communities. While effects vary between gears and habitats (Collie et al. 2000; Kaiser et al. 2006), epifaunal species are likely to be more vulnerable than infaunal species, simply because of the fact that they live on or near the seabed surface, rather than under it. While approaches that integrate the sampling of the full infaunal and epifaunal community are clearly going to provide more comprehensive community data than an approach that only focuses on (a part of) one component, more limited (or opportunistic) sampling may still be adequate for some purposes, and is likely to be considerably cheaper to collect, particularly when data are provided opportunistically from other studies. Within this research, we have compared dedicated benthic sampling studies with integrated infaunal and epifaunal sampling approaches with opportunistic epifaunal data sets collected by trawl and photographic surveys, with the results summarised, along with previous New Zealand studies using gradient approaches in Table 20. To aid interpretation and comparison between studies, a somewhat arbitrary scale of fishing effort gradient has been developed, related to the range of average aggregate fished area per cell observed across the study sites, as a measure of the number of times an area might be fished per year. While for the more recent or deeper water studies this metric can be readily estimated from the data provided by Black et al. (2013), for the other studies, this had to be inferred, largely from fisher knowledge. All studies had unfished or very lightly fished areas, and so gradients were defined on the basis of the maximum levels of effort, with categories for weak (fewer than five times per year), moderate (between five and ten times per year) and strong (over ten times per year). While the dedicated studies were specifically designed to examine benthic communities across fishing effort gradients (and so sampled across the maximum gradient available within an area) the effort gradients from the opportunistic studies generally reflected the distribution of fishing effort within a region, and gradients were weaker.

The strengths of effects detected varied between location (and even between study within location), which may be related to sampling approach or the fishing effort gradient over which the communities were examined (or both), and the overall magnitudes of effects detected appeared to be positively correlated with the strength of the fishing effort gradients. There was no clear evidence that the magnitude of effects varied between habitats (for a given strength of effort gradient), but there were probably insufficient studies to examine this appropriately, and the types of meta-analysis previously applied to 50 – 100 studies (e.g., Collie et al. 2000; Kaiser et al. 2006) would be more likely to be able to detect such differences. Where more than one sampling approach has been considered within a dedicated study (Spirits Bay, Chatham Rise / Challenger Plateau, Tasman and Golden Bays, and South Canterbury Bight), the different types of data were reasonably consistent with the magnitudes of the effects detected (Table 20). While this table has been compiled from ordination type analysis approaches of the community composition data, univariate measures (species richness, measures of diversity) showed similar patterns. The opportunistic analysis of trawl survey bycatch data (for Chatham Rise / Challenger Plateau, Nelson Bays, and South Canterbury Bight) generally identified weaker (or

no) effects compared to the dedicated sampling (although for the Chatham Rise / Challenger Plateau an equally weak effect was detected over a stronger fishing gradient).

The strongest effects detected by opportunistic studies were those observed by Cryer et al. (2002), examining high taxonomic resolution trawl bycatch data over a strong fishing effort gradient (measured over a finer spatial scale than other studies discussed) in the Bay of Plenty, but analysis of the same quality of data over a weaker effort gradient off the Wairarapa coast failed to detect a significant effect. The analysis of (lower taxonomic resolution) trawl bycatch data over weak fishing effort gradients failed to detect effects in the Bay of Plenty (two separate analyses) and the Auckland Islands, although weak effects were identified from epifaunal community data from photographs from the same Bay of Plenty surveys, with the results not appearing to be sensitive to the resolution of taxonomic identification. In an analysis from the Hawke Bay / Wairarapa area over a moderate effort gradient, both benthic trawl bycatch and epifaunal community data from photographs were able to detect effects of fishing (with the trawl data detecting stronger effects).

Overall, either no effect, or weak and inconsistent (between methods) effects were detected over weak fishing gradients, but as sites with a greater intensity of fishing were included, the strength of detected effects increased, and more approaches consistently detected effects. Opportunistic studies consistently detected effects of fishing over strong gradients, but were less consistent over weaker gradients. The relative strengths of the effects detected by photo/video sampling of the epifauna and trawl bycatch varied between studies, and there was no consistent pattern. While opportunistic studies have the advantage of providing data on (a subset of) the benthic community composition at minimal cost, the fact that sampling (typically random stations within strata based on target species abundance) has not been designed with an analysis over gradients of fishing pressure in mind often means that fishing gradients available may not be particularly strong, or representative. This was the case in all of our opportunistic studies, and by design the Cryer et al. (2002) study combined both random survey stations and targeted high fishing effort locations to provide a strong effort gradient.

Table 20: Summary of approaches and strength (% of total community composition variance explained by fishing variables alone) of effects of fishing detected from New Zealand studies.

Area / study	Environment		Fishing effort						Data and effects detected				Source
	Depth range (m)	Sediment types*	Statistical area	Tow by tow (1991 – study)	Tow by tow (2007 – study)	Fisher knowledge	Fishing gradient**	Maximum effort level***	Infaunal grab/core	Epifauna dredge	Epifaunal video/photo	Trawl survey bycatch	
Spirits Bay	30 – 100	S, G	•	•		•	S	note 1	15–29		17–30		Tuck & Hewitt (2013)
Hauraki Gulf	10 – 40	S, M	•			•	S	note 2	15–20				Thrush et al. (1998)
Bay of Plenty	200 – 600	M		•			S	4				11–40†	Cryer et al. (2002)
Bay of Plenty (2012 data)	300 – 450	M		•			W	3			4	X	Present study
Bay of Plenty (2008 data)	300 – 450	M		•			W	3			6‡		Present study
Wairarapa	220 – 450	M		•			W	2				X†	Data collected by Cryer
Hawke Bay / Wairarapa	200 – 500	M		•			M	9			7	16	Present study
Nelson Bays	10 – 45	S, M	•		•	•	M	5	21		18	10	Present study
Chatham Rise / Challenger Plateau	100 – 1800	M		•			W	4		X	2		Present study
Chatham Rise / Challenger Plateau	100 – 1800	M		•			M	6				1	Present study
Sth Canterbury Bight	40 – 120	S, M	•		•		M	6	9	5		X	Present study
Auckland Islands	350 – 520	M		•			W	3			X‡	X	Present study

* Key to sediment types: M – mud; S – sand; G – gravel

** Fishing effort gradient (estimated range of times fished per year); S – strong (0 – >10), M – moderate (0 – 5<10); W – weak (0 – <5)

*** Maximum effort level reported as average times fished per year, except:

Note 1 – effort gradient for Spirits Bay study based on quantitative analysis of trawl effort data and combination of fisher knowledge on fished areas and CELR effort reports

Note 2 – analysis based on ranked effort, based on fisheries legislation and fisher knowledge

Bay of Plenty (Cryer et al. 2002) – estimated over 1000 tows within 200 m of most heavily fished site (1989–1996), although lower average effort estimated by Baird et al. (2015) at 5 km × 5 km cell scale.

† - Cryer et al. (2002) and the Wairarapa study involved specialist taxonomic identification

‡ - These studies identified epifauna to broad OTUs

• - types of fishing effort data available for study

Strength of fishing effect detected (total variance accounted for): X – no effect detected

Detected effects of fishing

Fishing was found to have a significant effect on the composition, species richness, and measures of diversity of benthic communities, with at least 10%, and up to 40% of total variation in community composition accounted for by fishing over strong gradients. As discussed above, the magnitude of effects detected was correlated with the strength of the gradient over which the study was conducted, and for some studies over weak gradients, no or very weak effects were detected. Failure to detect effects in some studies may be because of the weak gradients over which the benthic community data were examined, or it may be because early fishing had already removed the most sensitive species.

While the changes in community composition detected in relation to fishing are a useful metric, in order to determine the likely implications of these changes, it is useful to identify effects on more interpretable indices, and where these are found, which individual species abundances are changing in relation to fishing pressure. When effects were detected, fishing was associated with reductions in the number of taxa, diversity and evenness of both epifaunal and infaunal communities, with the strength of effects correlated with the fishing gradient, and more consistently detected for epifauna than infauna. On the basis of the same community data, fishing appears to have reduced (by up to 50%) the estimated epifaunal biomass and productivity (whole community, and species considered main benthic components of fish prey) (identified in both the Tasman and Golden Bays and South Canterbury Bight studies), and increased (by up to 20%) the estimated infaunal biomass and productivity (identified in the Tasman and Golden Bays).

The species that were most consistently identified as being negatively correlated with fishing pressure were those that either stand erect out of the seabed (horse mussels *Atrina zelandica*, sponges, bryozoans, hydroids, sea pens, ascidians, tube building polychaetes), or live on the sediment surface, and might be particularly sensitive to physical disturbance through either direct physical impact (e.g., *Echinocardium* and other sea urchins, holothurians), smothering (e.g., small bivalves like *Corbula*), or increased vulnerability to predation following disturbance (e.g., brittle star, hermit crabs, large gastropods). In some studies, relatively modest levels of fishing effort (i.e., fishing an area between once and twice per year) appear to reduce the density of the combined group of sediment destabilisers or long lived sedentary habitat forming species (e.g., Figure 26, Figure 27, Figure 50) and individual species group densities of holothurians, crinoids, cnidarians and bryozoans (Figure 72) by up to 50%, while from other studies, comparable effort gradients appeared to have a weaker effect (e.g., Figure 57).

While the species that stand erect out of the seabed provide physical habitat structure for other organisms, the sediment surface organisms cause bioturbation, and can also produce small scale topographic modifications to the seabed. All of these features are considered to enhance habitat complexity, and may play an important role in provision of settlement substrate and refugia from predation. As previously identified, bioturbation influences benthic primary production, organic matter degradation and nutrient recycling (Lohrer et al. 2004; Huettel et al. 2014).

Implications of the effects of fishing

The implications of changes in community structure can be difficult to interpret, but the identified reductions in species richness, diversity and evenness imply both epifaunal and infaunal communities being impacted. The predicted changes in benthic biomass and production (up to 50% decrease for epifauna, up to 20% increase for infauna) may have implications for fish predating on these communities. Hiddink et al. (2006) estimated that trawling had reduced benthic biomass and production in the North Sea by 56% and 21% respectively, but did not discriminate between epifauna and infauna. Reductions in fish condition detected across gradients of fishing pressure have been interpreted as the effects of reduced prey availability (Hiddink et al. 2011), and similar effects could be present in our study areas.

Lundquist et al. (2013) analysed two of the larger data sets from the current study (Tasman and Golden Bays, and Chatham Rise / Challenger Plateau dedicated sampling studies), allocating species into trait-based functional groups, and then examining abundances of these groups in relation to fishing pressure, and in comparison with predictions from a seafloor community dynamics model, developed from a

spatially explicit patch dynamic model (Thrush et al. 2005; Lundquist et al. 2010). Model predictions were validated by field data, suggesting broad generalisations about the effects of fishing on functional diversity, and other aspects of biodiversity and ecosystem function. The functional groups representing emergent epifauna (providing much of the above-sediment habitat structure) and substrate destabilisers (providing much of the physical reworking of surface sediments) showed the strongest negative impacts in the studies documented here, and have also been previously identified as sensitive to fishing disturbance (e.g., Eleftheriou & Robertson 1992; Jennings & Kaiser 1998; Hall-Spencer et al. 1999).

The factor ceiling response type analysis of these functional groups in relation to fishing intensity provides useful indications of the potential magnitudes of change in functional group abundances, and therefore the implications for the processes or functions those groups serve. While the results of the analyses vary between studies, there is reasonably consistent evidence that the abundance of individuals within key functional groups is reduced by fishing. Abundances may be reduced by as much as 50% in areas fished at intensity levels greater than once per year (Figure 27, Figure 50).

Results obtained from this study suggest implications to ecosystem functioning by at least four processes: habitat complexity; bioturbation (surface sediment reworking and destabilisation); suspension feeding; and, because increased density of habitat structure protects the seafloor from disturbance by waves and currents, sediment stabilisation.

- Habitat complexity often results in increased biodiversity (Hewitt et al. 2005b; Mormul et al. 2011; Buhl-Mortensen et al. 2012), and it is likely that much of the decreases in species richness observed in this study result from decreases in habitat complexity. The abundance of erect habitat forming epifauna (Functional group 6) shows a consistent negative factor ceiling response with fishing pressure, and is retained as a significant explanatory variable in a number of the models investigating infaunal community structure and univariate metrics (accounting for 1 – 7% of overall variability when retained). Habitat structure also provides refugia from predation (Talman et al. 2004) and supports higher densities of fish in some habitats (Compton et al. 2012; Miller et al. 2012; Tracey et al. 2012; Parsons et al. 2013). While the data to quantify these services are not available, Talman et al. (2004) observed 44% higher predation on juvenile scallops in fished habitats with little structure compared to in unfished areas with complex habitat structure, and the work of Parsons et al. (2015) suggest that structurally complex habitats may provide small fish with access to high water flow (and planktonic food availability) sites.
- For bioturbation, manipulative studies have shown that the abundance of *Echinocardium* increases nitrogen fluxes from the seafloor to the extent that a decrease of 80% abundance of *Echinocardium* was accompanied by a decrease of 70% in ammoniacal nitrogen flux from the seafloor to the water (Lohrer et al. 2004). Density of *Echinocardium* in shallow waters has also been shown to be positively related to microphytobenthos biomass, probably through changes to nutrient fluxes, with a 50% decrease in *Echinocardium* density resulting in a 25% decrease in chlorophyll a (an indicator of microphytobenthos biomass (Lohrer et al. 2015)). Many other types of fauna (including shrimps, crabs, brittle stars, gastropods, tube worms and bivalves) have been demonstrated to affect nutrient fluxes and primary productivity (e.g., Chennu et al. (2015) found a 150–250% increase in microphytobenthos biomass with the addition of lugworms to sediment).
- Suspension feeding is an important connector between the seafloor sediment and the water column, increasing removal of both sediment and phytoplankton from the water column. There is little information available to link decreases in suspension feeder biomass to decreases in water clarity, however, clearance rates of *Atrina* 10 – 100 mg sediment per hour have been recorded (Hewitt & Pilditch 2004). Indirect effects of *Atrina* on nitrogen fluxes have also been observed, with the presence of *Atrina* predicted to increase these by 80% (Gibbs et al. 2005; Hewitt et al. 2006).
- The potential for habitat structure to protect and stabilise the seafloor is well known, from bed armouring by shellfish through to the potential for dense patches of structure to change benthic

boundary layers and create “skimming” flow. This has been documented for *Atrina* (Nikora et al. 2002) but as it is highly dependent on flow, species and patch characteristics, no attempt has been made here to quantify the degree to which percentage loss of biomass would impact on sediment stability. Importantly, the links between habitat structure, suspension feeding, and sediment stabilisation can form a situation where reduction in the biomass of suspension-feeding, habitat-structuring species could result in increased suspended sediment concentrations, leading to further mortality of these species (Coco et al. 2006). In systems where such species were dominant and where other important species rely on the habitat structure for protection from predation or are affected by low water clarity or high suspended sediments (e.g., scallops), compounding effects could quickly occur.

In order to estimate the level of impact on benthic communities at the scale of fisheries and the EEZ, it is necessary to quantify the levels of fishing effort at appropriate spatial resolution. This has been done for trawl fisheries (bottom trawl or midwater trawl within 1 m of the seabed) for both coastal waters down to 250 m (Baird et al. 2015) and all areas where vessels have reported on the TCEPR forms (Black et al. 2013), at a 5 km × 5 km scale. A single analysis has not yet examined trawl fisheries across the whole EEZ, and so we have generated a composite map (Figure 73) on the basis of the average annual fishing intensity (calculated at aggregate swept area per grid cell divided by cell area) from the most recent 5 years in each data set. This time period was not the same for the two data sets (2007/8 – 2011/12 for the coastal data, 2006/7 – 2010/11 for the TCEPR data), but given the reasonably consistent patterns of effort observed, the map provides a good representation of the overall recent bottom trawl fishing intensity across the EEZ. Since the Baird et al. (2015) (coastal data set) includes all trawling down to 250 m, while the Black et al. (2013) data set only includes vessels reporting on TCEPR (which should be all deep-water vessels, but only a proportion of the inshore fleet), the composite effort map was generated by using the Baird et al. (2015) data to replace anything for the same spatial location in the Black et al. (2013) data. At present, comparable fishing effort maps for any vessels reporting on TCER forms but fishing deeper than 250 m are not available, and are therefore not included. Dredge and Danish seine fisheries are not required to report fine scale (tow by tow) catch and effort data, and are also excluded from Figure 73.

Ideally fishing intensity patterns would be examined in relation to seabed habitat data, but in the absence of an agreed appropriate habitat map, we considered it useful to examine the patterns by depth, on the basis of the mean depth within each 5 km × 5 km cell (Figure 74). The distributions of fishing intensity by depth band are highly positively skewed (except at the greatest depths), and while median fishing intensity levels are quite low, some cells in each of the four shallower depth bands have been fished an average of over 5 times a year. Median fishing intensity declines with depth (Figure 74), while the percentage of the area unfished (estimated over 5 km × 5 km cells, averaged over the 5 most recent year’s data) increases from about 9% in the 0 – 100 m depth band to over 90% for 1000–1600 m, and almost 100% deeper than this (Table 21). Within the 0 – 100 m depth band, 9.2% of the area is fished more than once a year, and this percentage drops to 1.9%, 3.5% and 1.6% for the 100 – 200 m, 200 – 500 m and 500 – 1000 m depth bands, respectively. Examination of the relationships between the abundance within different functional groupings and fishing intensity observed within the current studies would suggest that these relative areas (e.g., 9.2% of the 0 – 100 m depth band) may have substrate destabilisers and emergent epifauna depleted to up to 50% of undisturbed levels.

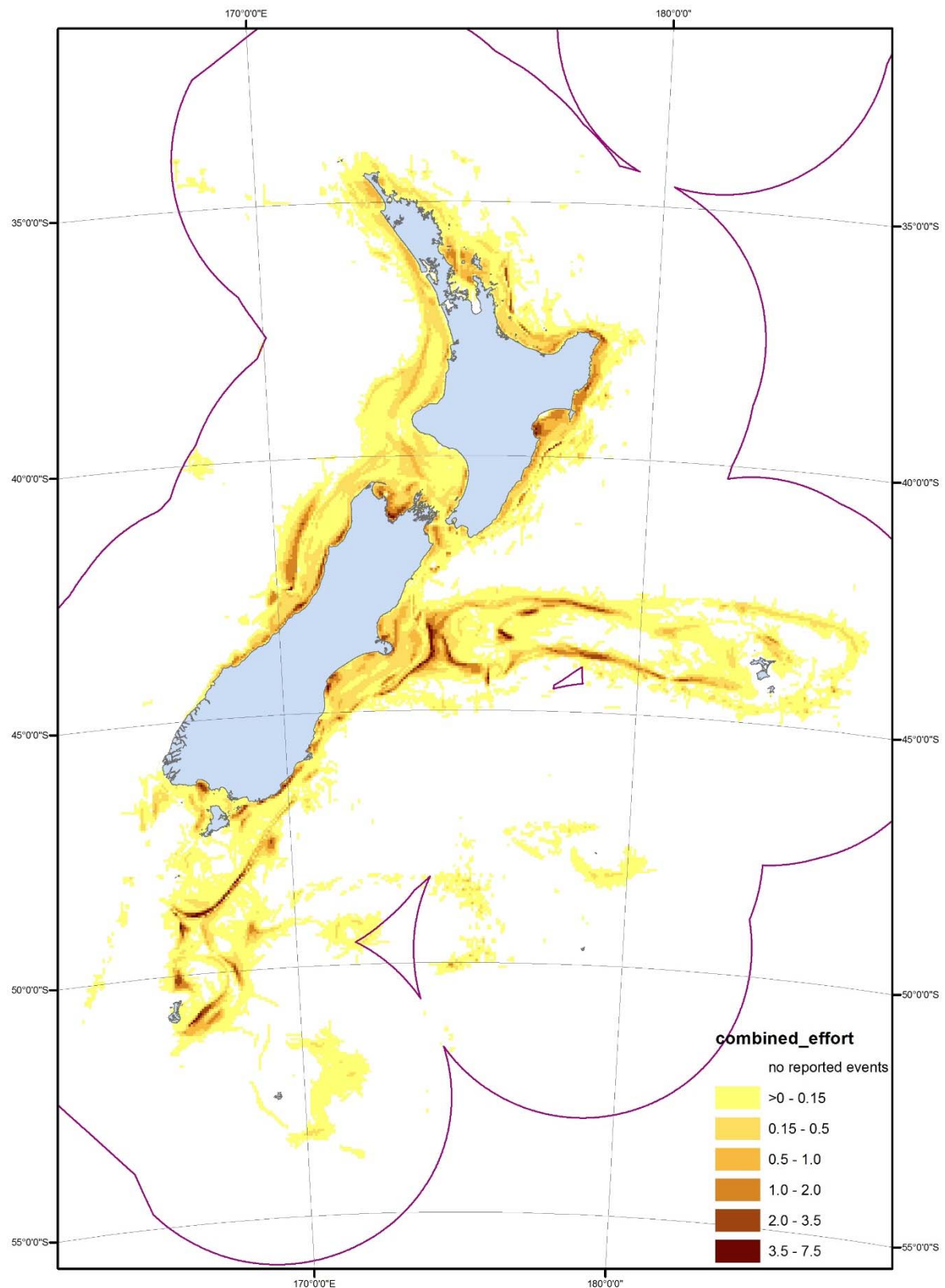


Figure 73: Fishing intensity (number of times fished, averaged over most recent 5 years available), calculated as aggregate swept area (at the cell level) divided by cell area. Map generated by combining data from Black et al. (2013) and Baird et al. (2015). The years for which the data are presented vary depending on the data source, and are 2007/8 – 2011/12 for the coastal data, 2006/7 – 2010/11 for the TCEPR data.

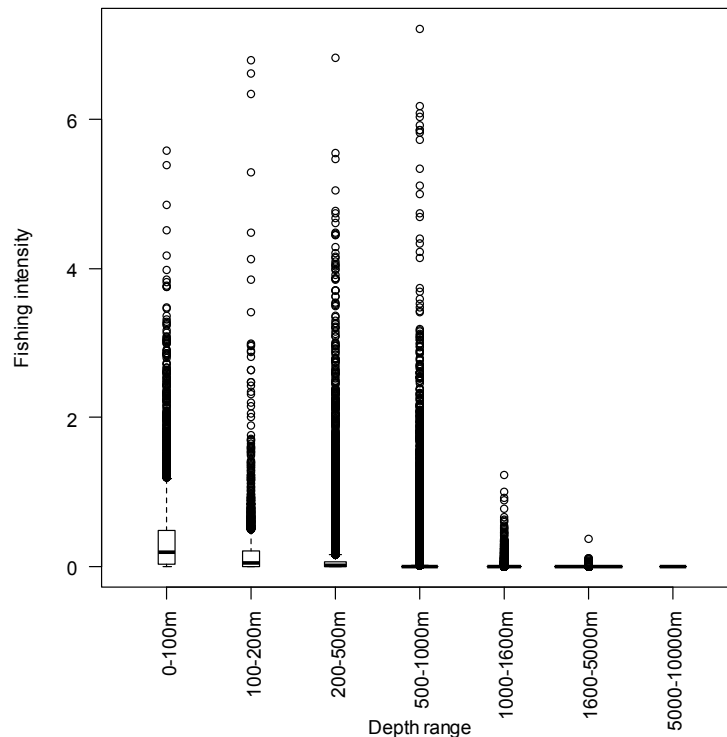


Figure 74: Boxplots of fishing intensity (number of times fished, averaged over most recent 5 years available) by depth band within the New Zealand EEZ. Box widths proportional to the square root of the area within each depth band. The years for which the data are presented vary depending on the data source, and are 2007/8 – 2011/12 for the coastal data, 2006/7 – 2010/11 for the TCEPR data.

Table 21: Fishing intensity (number of times fished, averaged over most recent 5 years available) by depth band within the New Zealand EEZ. Percentage of each depth band by fishing intensity level, estimated on the basis of Figure 73. The years for which the data are presented vary depending on the data source, and are 2007/8 – 2011/12 for the coastal data, 2006/7 – 2010/11 for the TCEPR data.

Depth range (m)	Fishing intensity							
	Not fished	<0.1	0.1–0.5	0.5–1	1–2	2–3.5	3.5–7.5	> 1
0–100	8.84%	38.23%	29.19%	14.51%	6.91%	2.11%	0.21%	9.23%
100–200	26.94%	43.50%	22.83%	4.86%	1.27%	0.43%	0.18%	1.87%
200–500	42.37%	42.32%	8.64%	3.22%	2.17%	0.93%	0.36%	3.46%
500–1000	74.06%	19.62%	3.17%	1.56%	1.15%	0.36%	0.08%	1.59%
1000–1600	90.26%	8.94%	0.75%	0.05%	0.01%			0.01%
1600–5000	99.41%	0.59%						
5000–10000	100.00%							

While the examination of biological traits has proved useful in allocating species to functional groups, the key to fully appreciating the implications of changes in community structure for ecosystem services is to map the biological traits to ecosystem processes and to determine how these processes link to ecosystem services (de Bello et al. 2010). Where they have been examined, multiple links between traits and processes have helped identify predictable trait-service clusters (specific processes are affected by a combination of traits, while particular key traits are simultaneously involved in the control of multiple processes), which depend on several trophic levels (e.g., clusters of traits of plants and soil organisms

control nutrient cycling, herbivory, and fodder and fibre production). The assessment of these trait-service clusters is considered to be a crucial component of understanding ecosystem service delivery (de Bello et al. 2010). The links between traits, processes and services appear to be relatively well understood in some terrestrial ecosystems (e.g., Lavorel 2013), but are less well documented for marine systems, particularly in the deep sea, although progress in this area is likely to be made for New Zealand species within studies undertaken as part of NIWA's Coasts & Oceans core funding and the Sustainable Seas National Science Challenge.

9. ACKNOWLEDGMENTS

This work was completed under Ministry for Primary Industries project BEN200701, and draws on a number of other studies funded by MPI. The Hauraki Gulf Cable Protection Zone study (Morrison et al. 2016) was funded by Auckland Council, through the Sea Change process. Throughout the BEN200701 project we benefitted from advice from Martin Cryer and Rich Ford (MPI), and the Aquatic Environment Working Group. A large number of staff have contributed to fieldwork, sample analysis, taxonomic identification of specimens, and provision of data layers for subsequent numerical analysis including Mike Page, Dan Cairney, Anna Bradley, Stephen Brown, Megan Carter, Mark Hadfield, Richard Gorman, Andrew Palmer, Andy Miller, Keren Spong, Suze Baird and Graham Fenwick. Roland Pitcher (CSIRO) provided a valuable review of the Tasman and Golden Bays study. David Bowden coordinated the MPI project ZBD200701, which shared the responsibility for analysis and identification of the Oceans Survey 20/20 Chatham Rise Challenger Plateau samples, and also provided very thorough review of this final report.

10. REFERENCES

- Anderson, M.J. (2001) A new method for non-parametric multivariate analysis of variance. *Austral Ecology*, 26: 32–46.
- Anderson, M.J., Gorley, R.N., Clarke, K.R. (2008) *PERMANOVA+ for PRIMER: Guide to Software and Statistical Methods*. PRIMER-E Ltd, Plymouth, UK.,
- Anderson, M.J., Willis, T.J. (2003) Canonical analysis of principal coordinates: A useful method of constrained ordination for ecology. *Ecology*, 84: 511–525.
- Anderson, O.F. (2012) Fish and invertebrate bycatch and discards in New Zealand scampi fisheries from 1990–91 until 2009–10. *New Zealand Aquatic Environment and Biodiversity Report*, 100: 65 p.
- Baird, S.J., Hewitt, J.E., Wood, B.A. (2015) Benthic habitats and trawl fishing disturbance in New Zealand waters shallower than 250 m. *New Zealand Aquatic Environment and Biodiversity Report*, 144: 184 p.
- Baird, S.J., Tracey, D., Mormede, S., Clark, M. (2013) The distribution of protected corals in New Zealand waters (DOC12303 / POP2011-06). *NIWA Client Report prepared for Marine Conservation Services (CSP) Department of Conservation*: 93. <http://www.doc.govt.nz/conservation/marine-and-coastal/commercial-fishing/conservation-services-programme/meetings-and-project-updates/27-november-2012/>
- Baird, S.J., Wood, B., Bagley, N. (2011) Nature and extent of commercial fishing effort on or near the seafloor within the New Zealand 200 n. mile Exclusive Economic Zone. *New Zealand Aquatic Environment and Biodiversity Report*, 73: 50 p.
- Ball, B.J., Fox, G., Munday, B.W. (2000) Long- and short-term consequences of a *Nephrops* trawl fishery on the benthos and environment of the Irish Sea. *ICES Journal of Marine Science*, 57: 1315–1320.
- Beck, M.W., Heck, K.L., Able, K.W., Childers, D.L., Eggleston, D.B., Gillanders, B.M., Halpern, B., Hays, C.G., Hoshino, K., Minello, T.J., Orth, R.J., Sheridan, P.F., Weinstein, M.P. (2001) The Identification, Conservation, and Management of Estuarine and Marine Nurseries for Fish and Invertebrates: A better understanding of the habitats that serve as nurseries for marine species and the factors that create site-specific variability in nursery quality will improve conservation and management of these areas. *BioScience*, 51(8): 633–641. 10.1641/0006-3568(2001)051[0633:ticamo]2.0.co;2
- Beentjes, M.P., MacGibbon, D.J., Lyon, W.S. (2013) Inshore trawl survey of Canterbury Bight and Pegasus Bay April–June 2012, (KAH1207). *New Zealand Fisheries Assessment Report*, 2013/36: 139 p.
- Bentley, N., Langley, A., Osborne, T., Lallemande, P. (2011) Segments and summaries of commercial fishing in New Zealand: approaches, data, methods and background analyses. *Final Research Report for Ministry for Primary Industries project SEC200902*: 41 p.

- Bernstein, B., Zalinski, J. (1983) An optimum sampling design and power tests for environmental biologists. *Journal of Environmental Management*, 16: 35–43.
- Black, J., Wood, R., Berthelsen, T., Tilney, R. (2013) Monitoring New Zealand's trawl footprint for deepwater fisheries: 1989–1990 to 2009–2010. *New Zealand Aquatic Environment and Biodiversity Report*, 110: 57 p.
- Blackburn, T.M., Lawton, J.H., Perry, J.N. (1992) A method of estimating the slope of upper bounds of plots of body size and abundance in natural animal assemblages. *Oikos*, 65: 107–112.
- Bowden, D.A. (2011) Benthic invertebrate samples and data from the Ocean Survey 20/20 voyages to Chatham Rise and Challenger Plateau, 2007. *New Zealand Aquatic Environment and Biodiversity Report*, 65: 46 p.
- Bowden, D.A., Hewitt, J.E. (2011) Recommendations for surveys of marine benthic biodiversity: outcomes from the Chatham-Challenger Ocean Survey 20/20 Post-Voyage Analyses Project. *New Zealand Aquatic Environment and Biodiversity Report*: 36 p.
- Bradstock, M., Gordon, D.P. (1983) Coral-like bryozoan growths in Tasman Bay, and their protection to conserve commercial fish stocks. *New Zealand Journal of Marine and Freshwater Research*, 17: 159–163.
- Brown, S.N. (2012) Ecology and enhancement of the flat oyster *Ostrea chilensis* (Philippi, 1845) in central New Zealand. PhD Thesis. University of Canterbury: 218 p.
- Buhl-Mortensen, L., Buhl-Mortensen, P., Dolan, M.F.J., Dannheim, J., Bellec, V., Holte, B. (2012) Habitat complexity and bottom fauna composition at different scales on the continental shelf and slope of northern Norway. *Hydrobiologia*, 685(1): 191–219. 10.1007/s10750-011-0988-6
- Cade, B.S., Terrell, J.W., Schroeder, R.L. (1999) Estimating effects of limiting factors with regression quantiles. *Ecology*, 80: 311–323.
- Chennu, A., Volkenborn, N., de Beer, D., Wetthey, D.S., Woodin, S.A., Polerecky, L. (2015) Effects of Bioadvection by *Arenicola marina* on Microphytobenthos in Permeable Sediments. *PLoS ONE*, 10(7): e0134236. 10.1371/journal.pone.0134236
- Churchman, G.J., Hunt, J.L., Glasby, G.P., Renner, R.M., Griffiths, G.A. (1988) Input of river-derived sediment to the New Zealand continental shelf: II mineralogy and composition. *Estuarine, Coastal and Shelf Science*, 27(4): 397–411. [http://dx.doi.org/10.1016/0272-7714\(88\)90096-0](http://dx.doi.org/10.1016/0272-7714(88)90096-0)
- Clarke, K.R., Green, R.H. (1988) Statistical design and analysis for a 'biological effects' study. *Marine Ecology Progress Series*, 46: 213–226.
- Coco, G., Thrush, S.F., Green, M.O., Hewitt, J.E. (2006) Feedbacks between bivalve (*Atrina zelandica*) density, flow, and suspended sediment concentration on patch stable states. *Ecology*, 87(11): 2862–2870. 10.1890/0012-9658(2006)87[2862:FBBDFA]2.0.CO;2

Cohen, J. (1988) *Statistical power analysis for behavioural sciences*. Hove and London, Hillsdale, New Jersey.

Collie, J.S., Hall, S.J., Kaiser, M.J., Poiner, I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69(5): 785–798.

Compton, T.J., Bowden, D.A., Roland Pitcher, C., Hewitt, J.E., Ellis, N. (2013) Biophysical patterns in benthic assemblage composition across contrasting continental margins off New Zealand. *Journal of Biogeography*, 40: 75–89. 10.1111/j.1365-2699.2012.02761.x

Compton, T.J., Morrison, M.A., Leathwick, J.R., Carbines, G.D. (2012) Ontogenic habitat associations of a demersal fish species, *Pagrus auratus*, identified using boosted regression trees. *Marine Ecology Progress Series*, 462: 219–230.

Cranfield, H.J., Michael, K.P., Doonan, I.J. (1999) Changes in the distribution of epifaunal reefs and oysters during 130 years of dredging for oysters in Foveaux Strait, southern New Zealand. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 9: 461–483.

Cryer, M., Downing, K., Hartill, B., Drury, J., Armiger, H.J., Middleton, C., Smith, M.D. (2005) Digital photography as a stock assessment tool for *Metanephrops challengerii* on New Zealand's continental slope. *Deep Sea 2003: Conference on the Governance and Management of Deep-sea*, Queenstown, New Zealand, 1-5 December 2003.

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.

Cryer, M., O' Shea, O., Gordon, D., Kelly, M., Drury, J., Morrison, M., Hill, A., Saunders, H., Shankar, U., Wilkinson, M., Foster, G. (2000) Distribution and structure of benthic invertebrate communities between North Cape and Cape Reinga. *Final Research Report for Ministry of Research Project ENV98-05*, Objectives 1-4.

Cummings, V.J., Nicholls, P.E., Thrush, S.F. (2003) *Mahurangi Estuary ecological monitoring programme - report on data collected from July 1994 to January 2003*. NIWA Client Report: HAM2003-066, Prepared for ARC: 68.

Currie, D.R., Parry, G.D. (1996) Effects of scallop dredging on a soft sediment community: a large-scale experimental study. *Marine Ecology Progress Series*, 134(1-3): 131–150.

Dahlgren, C.P., Kellison, G.T., Adams, A.J., Gillanders, B.M., Kendall, M.S., Layman, C.A., Ley, J.A., Nagelkerken, I., Serafy, J.E. (2006) Marine nurseries and effective juvenile habitats: concepts and applications. *Marine Ecology Progress Series*, 312: 291–295. http://gateway.isiknowledge.com/gateway/Gateway.cgi?GWVersion=2&SrcAuth=Alerting&SrcApp=Alerting&DestApp=CCC&DestLinkType=FullRecord&KeyUT=000237918300025*Order Full Text []

Dayton, P.K., Thrush, S., Coleman, F.C. (2002) *The ecological effects of fishing in marine ecosystems of the United States*. Pew Oceans Commission, Arlington, Virginia: 45 p.

Dayton, P.K., Thrush, S.F., Agardy, T.M., Hofman, R.J. (1995) Environmental effects of fishing. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 5: 205–232.

de Bello, F., Lavorel, S., Díaz, S., Harrington, R., Cornelissen, J.C., Bardgett, R., Berg, M., Cipriotti, P., Feld, C., Hering, D., Martins da Silva, P., Potts, S., Sandin, L., Sousa, J., Storkey, J., Wardle, D., Harrison, P. (2010) Towards an assessment of multiple ecosystem processes and services via functional traits. *Biodiversity and Conservation*, 19(10): 2873-2893. 10.1007/s10531-010-9850-9

DFO (2006) Impacts of trawl gears and scallop dredges on benthic habitats, populations and communities. . *Canadian Science Advisory Secretariat Science Advisory Report*, 2006/025.

E. Bros, W., C. Cowell, B. (1987) A technique for optimizing sample size (replication). *Journal of Experimental Marine Biology and Ecology*, 114(1): 63–71. [http://dx.doi.org/10.1016/0022-0981\(87\)90140-7](http://dx.doi.org/10.1016/0022-0981(87)90140-7)

Eberhardt, L.L., Thomas, J.M. (1991) Designing Environmental Field Studies. *Ecological Monographs*, 61(1): 53–73. 10.2307/1942999

Eleftheriou, A., Robertson, M.R. (1992) The effects of experimental scallop dredging on the fauna and physical environment of a shallow sandy community. *Netherlands Journal of Sea Research*, 30: 289–299.

Ellis, J.I., Schneider, D.C. (1997) Evaluation of a gradient sampling design for environmental impact assessment. *Environmental Monitoring and Assessment*, 48(2): 157–172.

Ellis, J.I., Schneider, D.C., Thrush, S.F. (2000) Detecting anthropogenic disturbance in an environment with multiple gradients of physical disturbance, Manukau Harbour, New Zealand. *Hydrobiologia*, 440(1-3): 379–391.

Ellis, N., Pantus, F. (2001) Management strategy modelling: tools to evaluate trawl management strategies with respect to impacts on benthic biota within the Great Barrier Reef Marine Park area.: 132. http://www.cmar.csiro.au/e-print/open/ellisn_2001.pdf

Floorl, O., Hewitt, J.E., Bowden, D.A. (2012) Chatham-Challenegr Ocean Survey 20/20 Post Voyage analyses: Objective 9 - Patterns in Species Composition. *New Zealand Aquatic Environment and Biodiversity Report*, 2012/97: 40.

Gibbs, M., Funnell, G., Pickmere, S., Norkko, A., Hewitt, J. (2005) Benthic nutrient fluxes along an estuarine gradient: influence of the pinnid bivalve *Atrina zelandica* in summer. *Marine Ecology Progress Series*, 288: 151-164. http://gateway.isiknowledge.com/gateway/Gateway.cgi?GWVersion=2&SrcAuth=Alerting&SrcApp=Alerting&DestApp=CCC&DestLinkType=FullRecord&KeyUT=ISI:000228394400013*Order Full Text []

Gislason, H. (1994) Ecosystem effects of fishing activities in the North Sea. *Marine Pollution Bulletin*, 29: 520–527.

Gislason, H., Sinclair, M., Sainsbury, K., O'Boyle, R. (2000) Symposium overview: incorporating ecosystem objectives within fisheries management. *ICES Journal of Marine Science*, 55: 362–370.

Gorman, R., Bryan, K.R., Laing, A.K. (2003) Wave hindcast for the New Zealand region: nearshore validation and coastal wave climate. *New Zealand Journal of Marine and Freshwater Research*, 37: 567–588.

Grange, K.R., Tovey, A., Hill, A.F. (2003) The spatial extent and nature of the bryozoan communities at Separation Point, Tasman Bay. *New Zealand Marine Biodiversity Biosecurity Report*, 4: 22 pp.

Gray, J.S., Dayton, P.K., Thrush, S.F., Kaiser, M.J. (2006) On effects of trawling, benthos and sampling design. *Marine Pollution Bulletin*, 52: 840–843.

Green, M.O., Vincent, C.E., McCave, I.N., Dickson, R.R., rees, J.M., Pearson, N.D. (1995) Storm sediment transport: observations from the British North sea shelf. *Continental Shelf Research*, 15: 889–912.

Greenstreet, S.P.R., Spence, F.E., McMillan, J.A. (1999) Fishing effects in northeast Atlantic shelf seas: patterns in fishing effort, diversity and community structure. V. Changes in structure of the North Sea groundfish species assemblage between 1925 and 1996. *Fisheries Research*, 40: 153–183.

Griffiths, G.A., Glasby, G.P. (1985) Input of river-derived sediment to the New Zealand continental shelf: I. Mass. *Estuarine, Coastal and Shelf Science*, 21(6): 773–787. [http://dx.doi.org/10.1016/0272-7714\(85\)90072-1](http://dx.doi.org/10.1016/0272-7714(85)90072-1)

Hadfield, M., Goring, D., Gorman, R., Wild, M., Stevens, S., Shankar, U., Niven, K., Snelder, T. (2002) Physical Variables for the New Zealand Marine Environment Classification System: Development and Description of Data Layers. *NIWA Report*, CHC2002-043: 59 p.

Hadfield, M., Zeldis, J. (2012) *Freshwater dilution and transport in Canterbury Bight*. : 39 p.

Haidvogel, D.B., Arango, H., Budgell, W.P., Cornuelle, B.D., Curchitser, E., Di Lorenzo, E., Fennel, K., Geyer, W.R., Hermann, A.J., Lanerolle, L., Levin, J., McWilliams, J.C., Miller, A.J., Moore, A.M., Powell, T.M., Shchepetkin, A.F., Sherwood, C.R., Signell, R.P., Warner, J.C., Wilkin, J. (2008) Ocean forecasting in terrain-following coordinates: Formulation and skill assessment of the Regional Ocean Modeling System. *Journal of Computational Physics*, 227(7): 3595–3624. <http://dx.doi.org/10.1016/j.jcp.2007.06.016>

Hall-Spencer, J.M., Frogia, C., Atkinson, R.J.A., Moore, P.G. (1999) The impact of Rapido trawling for scallops, *Pecten jacobaeus* (L.), on the benthos of the Gulf of Venice. *ICES Journal of Marine Science*, 56(1): 111–124.

Hall, S.J. (1994) Physical disturbance and marine benthic communities: life in unconsolidated sediments. *Oceanography and Marine Biology Annual Review*, 32: 179–239.

Hall, S.J. (1999) *The effects of fisheries on ecosystems and communities*. Blackwell Scientific, Oxford, England: 274 p.

Hall, S.J., Raffaelli, D., Thrush, S.F. (1994) Patchiness and disturbance in shallow water benthic assemblages. In: A.G. Hildrew, P.S. Giller & D. Raffaelli (Eds). *Aquatic ecology: Scale, pattern and processes*. Blackwell Scientific: 333–375.

Handley, S.J., Willis, T.J., Cole, R.G., Bradley, A., Cairney, D.J., Brown, S.N., Carter, M.E. (2014) The importance of benchmarking habitat structure and composition for understanding the extent of fishing impacts in soft sediment ecosystems. *Journal of Sea Research*, 86: 58–68.

Hewitt, J.E., Anderson, M.J., Thrush, S.F. (2005a) Assessing and monitoring ecological community health in marine systems. *Ecological Applications*, 15(3): 942–953. http://gateway.isiknowledge.com/gateway/Gateway.cgi?GWVersion=2&SrcAuth=Alerting&SrcApp=Alerting&DestApp=CCC&DestLinkType=FullRecord&KeyUT=ISI:000229723400013*Order Full Text []

Hewitt, J.E., Julian, K., Bone, E.K. (2010) Chatham–Challenger Ocean Survey 20/20 Post-Voyage Analyses: Objective 10 – Biotic habitats and their sensitivity to physical disturbance. *New Zealand Aquatic Environment and Biodiversity Report*, 81: 37 p.

Hewitt, J.E., Lundquist, C., Bowden, D.A. (2011) Chatham-Challenger Ocean Survey 20/20 Post Voyage Analyses: Diversity Metrics. *New Zealand Aquatic Environment and Biodiversity Report*, 83: 64 p.

Hewitt, J.E., McBride, G.B., Pridmore, R.D., Thrush, S.F. (1993) Patchy distributions: Optimizing sample size. *Environmental Monitoring and Assessment*, 27: 95–105.

Hewitt, J.E., Pilditch, C.A. (2004) Short-term feeding responses of *Atrina zelandica* to suspended sediment concentrations: effects of environmental history and physiological state. *Journal of Experimental Marine Biology and Ecology*, 306: 95–112.

Hewitt, J.E., Thrush, S., Gibbs, M., Lohrer, D., Norkko, A. (2006) Indirect effects of *Atrina zelandica* on water column nitrogen and oxygen fluxes: The role of benthic macrofauna and microphytes. *Journal of Experimental Marine Biology and Ecology*, 330(1): 261–273. http://gateway.isiknowledge.com/gateway/Gateway.cgi?GWVersion=2&SrcAuth=Alerting&SrcApp=Alerting&DestApp=CCC&DestLinkType=FullRecord&KeyUT=000236058900022*Order Full Text []

Hewitt, J.E., Thrush, S.F., Cummings, V.J. (2001) Assessing environmental impacts: effects of spatial and temporal variability at the scale of likely impacts. *Ecological Applications*, 11: 1502–1516.

Hewitt, J.E., Thrush, S.F., Halliday, J., Duffy, C. (2005b) The importance of small-scale biogenic habitat structure for maintaining beta diversity. *Ecology*, 86: 1619–1626.

Hewitt, J.E., Thrush, S.F., Legendre, P., Funnell, G.A., Ellis, J., Morrison, M. (2004) Mapping of marine soft-sediment communities: integrated sampling for ecological interpretation. *Ecological Applications*, 14 (4): 1203–1216.

Hiddink, J.G., Jennings, S., Kaiser, M.J., Queiros, A.M., Duplisea, D.E., Piet, G.J. (2006) Cumulative impacts of seabed trawl disturbance on benthic biomass, production, and species richness in different habitats. *Canadian Journal Fisheries and Aquatic Science*, 63: 721–736.

Hiddink, J.G., Johnson, A.F., Kingham, R., Hinz, H. (2011) Could our fisheries be more productive? Indirect negative effects of bottom trawl fisheries on fish condition. *Journal of Applied Ecology*, 48(6): 1441-1449. 10.1111/j.1365-2664.2011.02036.x

Huettel, M., Berg, P., Kostka, J.E. (2014) Benthic Exchange and Biogeochemical Cycling in Permeable Sediments. *Annual Review of Marine Science*, 6(1): 23-51. doi:10.1146/annurev-marine-051413-012706

Humborstad, O.-B., Nøttestad, L., Løkkeborg, S., Rapp, H.T. (2004) RoxAnn bottom classification system, sidescan sonar and video-sledge: spatial resolution and their use in assessing trawling impacts. *ICES Journal of Marine Science: Journal du Conseil*, 61(1): 53–63. 10.1016/j.icesjms.2003.10.001

Hurlbert, S.H. (1984) Pseudoreplication and the design of ecological field experiments. *Ecological Monographs*, 54: 197–211.

Hurst, R.J., Bagley, N.W. (1994) Trawl survey of middle depths and inshore bottom species off Southland, February-March 1993 (TAN9301). *New Zealand Fisheries Data Report*, 52: 61 p.

Jennings, S., Dinmore, T.A., Duplisea, D.E., Warr, K.J., Lancaster, J.E. (2001) Trawling disturbance can modify benthic production processes. *Journal of Animal Ecology*, 70: 459–475.

Jennings, S., Kaiser, M.J. (1998) The effects of fishing on marine ecosystems. *Advances in Marine Biology*, 34: 203–314.

Kaiser, M.J., Clarke, K.R., Hinz, H., Austen, M.C., Somerfield, P.J., Karakassis, I. (2006) Global analysis of response and recovery of benthic biota to fishing. *Marine Ecology Progress Series*, 311: 1–14.

Kaiser, M.J., de Groot, S.J. (2000) *The effects of fishing on non-target species and habitats*. Oxford, England, Blackwell Science.

Kaiser, M.J., Ramsay, K. (1997) Opportunistic feeding by dabs within areas of trawl disturbance: Possible implications for increased survival. *Marine Ecology Progress Series*, 152(1-3): 307–310.

Kaiser, M.J., Spencer, B.E. (1994) Fish scavenging behaviour in recently trawled areas. *Marine Ecology Progress Series*, 112: 41–49.

Lambert, G.I., Jennings, S., Kaiser, M.J., Davies, T.W., Hiddink, J.G. (2014) Quantifying recovery rates and resilience of seabed habitats impacted by bottom fishing. *Journal of Applied Ecology*, 51: 1326–1336. 10.1111/1365-2664.12277

Lavorel, S. (2013) Plant functional effects on ecosystem services. *Journal of Ecology*, 101(1): 4-8. 10.1111/1365-2745.12031

Leathwick, J.R., Rowden, A., Nodder, S., Gorman, R., Bardsley, S., Pinkerton, M., Baird, S.J., Hadfield, M., Currie, K., Goh, A. (2012) A Benthic-optimised Marine Environment

Classification (BOMEC) for New Zealand waters. *New Zealand Aquatic Environment and Biodiversity Report*, 88: 54 p.

Lindeboom, H.J., de Groot, S.J. (1998) *Impact-II: The effect of different types of fisheries on the North Sea and Irish Sea benthic ecosystems*. Texel, The Netherlands, Netherlands Institute for Sea Research: 404 p.

Lohrer, A.M., Thrush, S.F., Gibbs, M.M. (2004) Bioturbators enhance ecosystem function through complex biogeochemical interactions. *Nature*, 431(7012): 1092–1095.

Lohrer, A.M., Thrush, S.F., Hewitt, J.E., Kraan, C. (2015) The up-scaling of ecosystem functions in a heterogeneous world. *Scientific Reports*, 5: 10349. 10.1038/srep10349
<http://www.nature.com/articles/srep10349#supplementary-information>

Lokkeborg, S. (2005) Impacts of trawling and scallop dredging on benthic habitats and communities. *FAO Fisheries Technical Paper*, 472.

Lundquist, C.J., Pinkerton, M.H. (2008) Collation of data for ecosystem modelling of Te Tapuwae o Rongokako Marine Reserve. *Science for Conservation*, 288: 105 p.

Lundquist, C.J., Pritchard, M., Thrush, S.F., Hewitt, J.E., Greenfield, B.L., Halliday, J., Lohrer, A.M. (2013) Bottom disturbance and seafloor community dynamics: Development of a model of disturbance and recovery dynamics for marine benthic ecosystems. *New Zealand Aquatic Environment and Biodiversity Report*, 118: 58 p.

Lundquist, C.J., Thrush, S.F., Coco, G., Hewitt, J.E. (2010) Interactions between disturbance and dispersal reduce persistence thresholds in a benthic community. *Marine Ecology Progress Series*, 413: 217–228. 10.3354/meps08578

MacCready, P., Banas, N.S., Hickey, B.M., Dever, E.P., Liu, Y. (2009) A model study of tide- and wind-induced mixing in the Columbia River Estuary and plume. *Continental Shelf Research*, 29(1): 278–291. <http://dx.doi.org/10.1016/j.csr.2008.03.015>

MacDiarmid, A., Bowden, D.A., Cummings, V., Morrison, M.A., Jones, E.G., Kelly, M., Neil, H.L., Nelson, W., Rowden, A. (2013) Sensitive marine benthic habitats defined, WLG2013-18: 72 p.

MacDiarmid, A., McKenzie, A., Sturman, J., Beaumont, J., Mikaloff-Fletcher, S., Dunne, J. (2012) Assessment of anthropogenic threats to New Zealand marine habitats. *New Zealand Aquatic Environment and Biodiversity Report*, 93: 255 p.

MacGibbon, D.J., Stevenson, M.L. (2013) Inshore trawl survey of the west coast South Island and Tasman and Golden Bays, March-April 2013 (KAH1305). *New Zealand Fisheries Assessment Report*, 2013/66: 119 p.

Malik, M.A., Mayer, L.A. (2007) Investigation of seabed fishing impacts on benthic structure using multi-beam sonar, sidescan sonar, and video. *ICES Journal of Marine Science: Journal du Conseil*, 64(5): 1053–1065. 10.1093/icesjms/fsm056

McArdle, B.H., Anderson, M.J. (2001) Fitting multivariate models to community data: a comment on distance-based redundancy analysis. *Ecology*, 82: 290–297.

McConnaughey, R.A., Mier, K.L., Dew, C.B. (2000) An examination of chronic trawling effects on soft-bottom benthos of the eastern Bering Sea. *ICES Journal of Marine Science: Journal du Conseil*, 57(5): 1377–1388. 10.1006/jmsc.2000.0906

McKnight, D.G. (1969) Infaunal benthic communities of the New Zealand continental shelf. *New Zealand journal of Marine and Freshwater Research*, 3: 409–444.

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.

Miller, R.J., Hocevar, J., Stone, R.P., Fedorov, D.V. (2012) Structure-Forming Corals and Sponges and Their Use as Fish Habitat in Bering Sea Submarine Canyons. *PLoS ONE*, 7(3): e33885. 10.1371/journal.pone.0033885

Ministry for Primary Industries (2014) Fisheries Assessment Plenary, May 2014: stock assessments and stock status: 1381 p.

Ministry of Fisheries (2005) Strategy for Managing the Environmental Effects of Fishing: 26 p.

Mitchell, J.S. (1987) Tasman sediments, 1:200,000 at Lat 41 deg S. *Coastal chart series*. Coastal chart series.

Mitchell, J.S., Carter, L., McFougall, J.C. (1989) New Zealand Regional Sediments (1:6 000 000). *NZOI Miscellaneous Chart Series 67*.

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*, 106: 20 p.

Mormul, R.P., Thomaz, S.M., Takeda, A.M., Behrend, R.D. (2011) Structural Complexity and Distance from Source Habitat Determine Invertebrate Abundance and Diversity. *Biotropica*, 43(6): 738-745. 10.1111/j.1744-7429.2011.00762.x

Morrison, M.A., Ching, N., Robertson, D., Nodder, S., Hewitt, J.E., Willis, T. (2010) *Bay of Islands OS20/20 survey report, Chapter 1: Introduction*. NIWA Client Report WLG2010-38: 12 p.

Morrison, M.A., Jones, E.G., Consalvey, M., Berkenbusch, K. (2014a) Linking marine fisheries species to biogenic habitats in New Zealand: a review and synthesis of knowledge. *New Zealand Aquatic Environment and Biodiversity Report*, 130: 160 p.

Morrison, M.A., Jones, E.G., Parsons, D.P., Grant, C.M. (2014b) Habitats and areas of particular significance for coastal finfish fisheries management in New Zealand: A review of concepts and life history knowledge, and suggestions for future research. *New Zealand Aquatic Environment and Biodiversity Report*, 125: 205 p.

Morrison, M.A., Tuck, I.D., Taylor, R.B., Miller, A. (2016) *An assessment of the Hauraki Gulf Cableway Protection Zone (CPZ), relative to adjacent seafloor*. Auckland Council Technical report 2016/004: 54 p.

Nelson, W.A., Gordon, D.P. (1997) Assessing New Zealand's marine biological diversity - a challenge for policy makers and systematists. *New Zealand Science Review*, 54: 58–66.

Nikora, V., Green, M.O., Thrush, S.F., Hume, T.M., Goring, D. (2002) Structure of the internal boundary layer over a patch of horse mussels (*Atrina zelandica*) in an estuary. *Journal of Marine Research*, 60: 121–150.

Nodder, S., Maas, E., Bowden, D.A., Pildich, C. (2011) Physical, biogeochemical, and microbial characteristics of sediment samples from the Chatham Rise and Challenger Plateau. *New Zealand Aquatic Environment and Biodiversity Report*, 70: 40 p.

Parsons, D.M., Middleton, C., Spong, K.T., Mackay, G., Smith, M.D., Buckthought, D. (2015) Mechanisms Explaining Nursery Habitat Association: How Do Juvenile Snapper (*Chrysophrys auratus*) Benefit from Their Nursery Habitat? *PLoS ONE*, 10(3): e0122137. 10.1371/journal.pone.0122137

Parsons, D.M., Morrison, M., Thrush, S.F., Middleton, C., Smith, M., Spong, K.T., Buckthought, D. (2013) The influences of habitat structure on juvenile fish in a New Zealand estuary. *Marine Ecology*, 34: 492–500.

Pinkerton, M.H., Lundquist, C., Duffy, C.A.J., Freeman, D.J. (2008) Trophic modelling of a New Zealand rocky reef ecosystem using simultaneous adjustment of diet, biomass and energetic parameters. *Journal of Experimental Marine Biology and Ecology*, 367: 189–203.

Puig, P., Canals, M., Company, J.B., Martin, J., Amblas, D., Lastras, G., Palanques, A., Calafat, A. (2012) Ploughing the deep sea floor. *Nature*, 489: 286–289.

Pusceddu, A., Bianchelli, S., Martín, J., Puig, P., Palanques, A., Masqué, P., Danovaro, R. (2014) Chronic and intensive bottom trawling impairs deep-sea biodiversity and ecosystem functioning. *Proceedings of the National Academy of Sciences*. 10.1073/pnas.1405454111

Ramsay, K., Kaiser, M.J. (1998) Demersal fishing disturbance increases predation risk for whelks (*Buccinum undatum* L.). *Journal of Sea Research*, 39: 299–304.

Ramsay, K., Kaiser, M.J., Hughes, R.N. (1996) Changes in hermit crab feeding patterns in response to trawling disturbance. *Marine Ecology Progress Series*, 144: 63–72.

Ramsay, K., Kaiser, M.J., Hughes, R.N. (1998) Responses of benthic scavengers to fishing disturbance by towed gears in different habitats. *Journal of Experimental Marine Biology and Ecology*, 224(1): 73–89.

Reiss, H., Greenstreet, S.P.R., Sieben, K., Ehrich, S., Piet, G.J., Quirijns, F., Robinson, L., Wolff, W.J., Kroncke, I. (2009) Effects of fishing disturbance on benthic communities and secondary production within an intensively fished area. *Marine Ecology Progress Series*, 394: 201–213. 10.3354/meps08243

Rice, J.C. (2006) Impacts of Mobile Bottom Gears on Seafloor Habitats, Species, and Communities: A Review and Synthesis of Selected International Reviews. *DFO Can. Sci. Advis. Sec. Res.*: 35.

Schwarz, J.N., Pinkerton, M.H., Wood, S., Zeldis, J. (2009) Remote sensing of river plumes in the Canterbury Bight: Stage IIA Progress Report. *NIWA Client Report CHC2009-207*.

Schwarz, J.N., Pinkerton, M.H., Wood, S., Zeldis, J. (2010) Remote sensing of river plumes in the Canterbury Bight: Stage IIA Progress Report. *NIWA Client Report CHC2010-048*.

Snelder, T.H., Leathwick, J., Image, K., Weatherhead, M., Wild, M. (2004) *The New Zealand Marine Environment Classification*. NIWA Client Report CHC2004–071: 86 p.

Snelder, T.H., Leathwick, J.R., Dey, K.L., Rowden, A.A., Weatherhead, M.A., Fenwick, G.D., Francis, M.P., Gorman, R.M., Grieve, J.M., Hadfield, M.G., Hewitt, J.E., Richardson, K.M., Uddstrom, M.J., Zeldis, J.R. (2006) Development of an ecologic marine classification in the New Zealand region. *Environmental Management*, 39: 12–29.

Stevens, D.W., Hurst, R.J., Bagley, N.W. (2011) Feeding habits of New Zealand fishes: a literature review and summary of research trawl database records 1960 to 2000. *New Zealand Aquatic Environment and Biodiversity Report*, 85: 218 p.

Stewart-Oaten, A., Murdoch, W.M., Parker, K.R. (1986) Environmental impact assessment: Pseudoreplication in time? *Ecology*, 67: 929–940.

Talman, S.G., Norkko, A., Thrush, S.F., Hewitt, J.E. (2004) Habitat structure and the survival of juvenile scallops (*Pecten novaezealandiae*): comparing predation in habitats with varying complexity. *Marine Ecology Progress Series*, 269: 197–207.

ter Braak, C.J.F. (1986) Canonical correspondence analysis: a new eigenvector method for multivariate direct gradient analysis. *Ecology*, 67: 1167–1179.

Thrush, S.F., Dayton, P.K. (2002) Disturbance to marine benthic habitats by trawling and dredging - Implications for marine biodiversity. *Annual Review of Ecology and Systematics*, 33: 449–473.

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(1–3): 141–150.

Thrush, S.F., Hewitt, J.E., Cummings, V.J., Dayton, P.K., Cryer, M., Turner, S.J., Funnell, G., Budd, R., Milburn, C., 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.

Thrush, S.F., Lundquist, C.J., Hewitt, J.E. (2005) Spatial and temporal scales of disturbance to the seafloor: A generalised framework for active habitat management. In: P.W. Barnes & J.P. Thomas (Eds). *Benthic habitats and the effects of fishing*. American Fisheries Society, Symposium Series, Bethesda, Maryland: 639–649.

Thrush, S.F., Whitlatch, R.B., Pridmore, R.D., Hewitt, J.E., Cummings, V.J., Maskery, M. (1996) Scale-dependent recolonization: the role of sediment stability in a dynamic sandflat habitat. *Ecology*, 77: 2472–2487.

Tracey, D.M., Anderson, O.F., Naylor, R.J. (2011a) A guide to common deepsea invertebrates in New Zealand waters. *New Zealand Aquatic Environment and Biodiversity Report*, 86: 317 p.

Tracey, D.M., Baird, S.J., Sanders, B.M., Smith, M.H. (2011b) *Distribution of protected corals in relation to fishing effort and assessment of accuracy of observer identification*. NIWA Client Report WLG2011-33: 74 p.

Tracey, D.M., Clark, M.R., Anderson, O.F., Kim, S.W. (2012) Deep-Sea Fish Distribution Varies between Seamounts: Results from a Seamount Complex off New Zealand. *PLoS ONE*, 7(6): e36897. 10.1371/journal.pone.0036897

Tuck, I.D. (2015) Characterisation and length-based population model for scampi (*Metanephrops challengeri*) at the Auckland Islands (SCI 6A). *New Zealand Fisheries Assessment Report*, 2015/21: 164 p.

Tuck, I.D., Drury, J., Kelly, M., Geering, P. (2010) Designing a programme to monitor the recovery of the benthic community between North Cape and Cape Reinga. *New Zealand Aquatic Environment and Biodiversity Report*, 53: 78 p.

Tuck, I.D., Hall, S.J., Robertson, M.R., Armstrong, E., Basford, D.J. (1998) Effects of physical trawling disturbance in a previously unfished sheltered Scottish sea loch. *Marine Ecology Progress Series*, 162: 227–242.

Tuck, I.D., Hewitt, J. (2013) Monitoring change in benthic communities in Spirits Bay. *New Zealand Aquatic Environment and Biodiversity Report*, No. 111: 52 p.

Tuck, I.D., Hewitt, J., Handley, S., Willis, T., Carter, M., Hadfield, M., Gorman, R., Cairney, D., Brown, S., Palmer, A. (2011) Assessing the effects of fishing on soft sediment habitat, fauna and processes. *Progress Report for Ministry of Fisheries research project BEN200701*: 30 p.

Tuck, I.D., Parkinson, D., Armiger, H., Smith, M., Miller, A., Rush, N., Spong, K. (2015a) Estimating the abundance of scampi in SCI 3 (Mernoo Bank) in 2013. *New Zealand Fisheries Assessment Report*, 2015/23: 49 p.

Tuck, I.D., Parkinson, D., Drury, J., Armiger, H., Miller, A., Rush, N., Smith, M., Hartill, B. (2012) Estimating the abundance of scampi in SCI 1 and SCI 2 (2012). *Final Research Report for Ministry for Primary Industries research project SCI201002A*: 53.

Tuck, I.D., Parsons, D.M., Hartill, B.W., Chiswell, S.M. (2015b) Scampi (*Metanephrops challengeri*) emergence patterns and catchability. *ICES Journal of Marine Science*, 72 (Supplement 1): i199–i210.
<http://icesjms.oxfordjournals.org/content/early/2015/01/08/icesjms.fsu244.abstract>

Tuck, I.D., Pinkerton, M.H., Tracey, D.M., Anderson, O.F., Chiswell, S.M. (2014) Ecosystem and environmental indicators for deepwater fisheries. *New Zealand Aquatic Environment and Biodiversity Report*, 127: 149 p.

Underwood, A.J. (1992) Beyond BACI: the detection of environmental impact on populations in the real, but variable, world. *Journal of Experimental Marine Biology and Ecology*, 161: 145–178.

University of Auckland (1975) *Marine environment of Golden Bay*. Report prepared for Shell BP and Todd Oil Services (NZ): 98 p.

Van Den Wollenberg, A.L. (1977) Redundancy analysis an alternative for canonical correlation analysis. *Psychometrika*, 42: 207–219.

van der Linden, W.J.M. (1969) Off-shore sediments, north-west Nelson, South Island, New Zealand. . *New Zealand Journal of Geology and Geophysics*, 12: 87–103.

Walters, R.A., Goring, D.G., Bell, R.G. (2001) Ocean tides around New Zealand. *New Zealand Journal of Marine and Freshwater Research*, 35(3): 567–579. 10.1080/00288330.2001.9517023

Warner, J.C., Sherwood, C.R., Signell, R.P., Harris, C.K., Arango, H.G. (2008) Development of a three-dimensional, regional, coupled wave, current, and sediment-transport model. *Computers & Geosciences*, 34(10): 1284–1306. <http://dx.doi.org/10.1016/j.cageo.2008.02.012>

Watling, L., Hoofd, M., Boulanger, M., Ferguson, N., Nouvian, C. (2014) The bottom line on trawling: How much more science do we need? *ICES Symposium on the Effects of fishing on benthic fauna, habitat and ecosystem function*, Tromso, Norway.

Zeldis, J., Hadfield, M., Morrissey, D., Broekhuizen, N., Stenton-Dozey, J. (2011a) *Tasman aquaculture: guidance on farming additive species – Stage 1*. NIWA Client Report CHC2011-005: 74.

Zeldis, J., Hadfield, M., Morrissey, D., Broekhuizen, N., Stenton-Dozey, J. (2011b) *Tasman aquaculture: guidance on farming additive species – Stage 2*. NIWA Client Report CHC2011-029: 69.

11. APPENDIX 1: Marginal tests for DistLM models for Tasman and Golden Bays case study

These show how much of the total variance each variable explains when taken alone, ignoring all other variables. All variables available to the full model are presented, with those retained in the final model identified. P represents significance of term, and Prop. represents the proportion of total variance explained by that term.

Tasman and Golden Bays, epifaunal community composition

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df	Final model
Teffort	3342	1.8041	0.072	0.047723	36	2	
OEffort	3892.6	2.1189	0.044	0.055586	36	2	*
SEffort	1851.7	0.97779	0.445	0.026443	36	2	*
SEav6yr	5926.9	3.3286	0.006	0.084636	36	2	*
Tmarks	1321.2	0.69227	0.71	0.018867	36	2	*
salmin	3302.8	1.7819	0.087	0.047164	36	2	*
salmx	7389	4.2466	0.003	0.10551	36	2	*
salql	5240.9	2.9122	0.009	0.07484	36	2	*
Depth	4153.1	2.2696	0.028	0.059306	36	2	*
Mud	5193.1	2.8835	0.014	0.074156	36	2	*
Chla	1024.8	0.53464	0.802	0.014634	36	2	
MxCURRENTS	3519.4	1.905	0.078	0.050257	36	2	
mxWaves	7412.1	4.2614	0.003	0.10584	36	2	*
East	8757.6	5.1456	0.001	0.12506	36	2	
North	7478.7	4.3043	0.001	0.1068	36	2	*

Tasman and Golden Bays, epifaunal species richness

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df	Final model
Teffort	225.73	0.50954	0.518	0.013956	36	2	
OEffort	1682.3	4.1792	0.034	0.10401	36	2	
SEffort	1717.2	4.2762	0.036	0.10617	36	2	*
SEav6yr	1281.3	3.0973	0.075	0.079219	36	2	
Tmarks	681.6	1.5839	0.21	0.042142	36	2	*
salmin	995.5	2.3612	0.122	0.061551	36	2	
salmx	3023.2	8.2763	0.007	0.18692	36	2	
salql	1983.9	5.0333	0.023	0.12266	36	2	*
Depth	1705.7	4.2442	0.042	0.10546	36	2	
Mud	1216.9	2.9289	0.083	0.075237	36	2	*
Chla	15.608	0.034774	0.958	0.000965	36	2	*
MxCURRENTS	175.82	0.39565	0.617	0.010871	36	2	
mxWaves	4491.4	13.841	0.002	0.2777	36	2	
East	5105.4	16.605	0.001	0.31566	36	2	
North	2594.8	6.8791	0.008	0.16043	36	2	*

Tasman and Golden Bays, epifaunal diversity (Shannon-Weiner)

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df	Final model
Teffort	69.898	0.19587	0.767	0.005411	36	2	*
OEffort	110.31	0.31009	0.69	0.00854	36	2	
SEffort	598.54	1.7492	0.169	0.046338	36	2	
SEav6yr	101.58	0.28537	0.789	0.007865	36	2	
Tmarks	449.03	1.2966	0.214	0.034763	36	2	
salmin	290.41	0.82801	0.382	0.022483	36	2	*
salmx	1259.3	3.8888	0.034	0.097492	36	2	
salql	165.74	0.46794	0.549	0.012832	36	2	
Depth	336.64	0.96336	0.378	0.026063	36	2	*
Mud	199.92	0.56596	0.552	0.015478	36	2	
Chla	79.163	0.22199	0.772	0.006129	36	2	
MxCURRENTS	77.003	0.2159	0.808	0.005961	36	2	*
mxWaves	240.97	0.68437	0.507	0.018656	36	2	
East	874.04	2.6128	0.084	0.067667	36	2	
North	1601.5	5.0953	0.011	0.12399	36	2	*

Tasman and Golden Bays, epifaunal evenness

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df	Final model
Teffort	11.65	0.057628	0.89	0.001598	36	2	*
OEffort	414.18	2.1688	0.108	0.056822	36	2	
Seffort	44.729	0.22227	0.706	0.006136	36	2	*
SEav6yr	146.4	0.73788	0.439	0.020085	36	2	
Tmarks	50.007	0.24868	0.651	0.006861	36	2	
salmin	885.11	4.9756	0.019	0.12143	36	2	*
salmx	57.575	0.28662	0.648	0.007899	36	2	
salql	1165	6.848	0.021	0.15982	36	2	
Depth	1460	9.0164	0.003	0.20029	36	2	*
Mud	140.54	0.70777	0.419	0.019281	36	2	
Chla	66.05	0.32919	0.612	0.009061	36	2	*
MxCURRENTS	99.832	0.4999	0.517	0.013696	36	2	*
mxWaves	489.95	2.5942	0.115	0.067217	36	2	
East	71.551	0.35688	0.604	0.009816	36	2	
North	223.15	1.1369	0.313	0.030614	36	2	*

Tasman and Golden Bays, epifaunal biomass

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df	Final model
Teffort	245.58	0.2033	0.81	0.005616	36	2	
OEffort	2857.5	2.5167	0.092	0.06534	36	2	
Seffort	657.28	0.54931	0.594	0.015029	36	2	
SEav6yr	238.79	0.19764	0.855	0.00546	36	2	*
Tmarks	536.94	0.44749	0.634	0.012278	36	2	*
salmin	1348.9	1.1457	0.284	0.030844	36	2	
salmx	4709.6	4.3447	0.021	0.10769	36	2	*
salql	1718.5	1.4725	0.217	0.039294	36	2	
Depth	2987.4	2.6394	0.065	0.06831	36	2	*
Mud	4289.8	3.9153	0.02	0.098091	36	2	*
Chla	200.53	0.16583	0.865	0.004585	36	2	
MxCURRENTS	609.91	0.50916	0.588	0.013946	36	2	
mxWaves	4754.3	4.391	0.015	0.10871	36	2	
East	5991.4	5.715	0.008	0.137	36	2	*
North	3735.8	3.3624	0.041	0.085423	36	2	

Tasman and Golden Bays, epifaunal prey biomass

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df	Final model
Teffort	292.79	0.21747	0.84	0.006005	36	2	
OEffort	3018.3	2.3755	0.087	0.061901	36	2	
Seffort	392.03	0.29178	0.792	0.00804	36	2	
SEav6yr	239.48	0.17768	0.904	0.004911	36	2	*
Tmarks	1119.8	0.84622	0.386	0.022966	36	2	*
salmin	1227.6	0.92974	0.385	0.025176	36	2	
salmx	4717.5	3.856	0.024	0.096748	36	2	*
salql	1418.5	1.0786	0.335	0.029091	36	2	
Depth	3387.5	2.6877	0.087	0.069472	36	2	*
Mud	4332.5	3.5106	0.036	0.088852	36	2	*
Chla	556.78	0.41582	0.685	0.011419	36	2	*
MxCURRENTS	1036.1	0.78159	0.428	0.021249	36	2	
mxWaves	3926.3	3.1526	0.056	0.080521	36	2	*
East	4930.8	4.0499	0.023	0.10112	36	2	*
North	3191.9	2.5216	0.089	0.06546	36	2	

Tasman and Golden bays, epifaunal production

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df	Final model
Teffort	161.92	0.15474	0.867	0.00428	36	2	
OEffort	2681.2	2.7461	0.072	0.070875	36	2	
SEffort	443.43	0.42698	0.66	0.011722	36	2	
SEav6yr	268.9	0.25772	0.806	0.007108	36	2	*
Tmarks	518.6	0.50036	0.574	0.013708	36	2	*
salmin	1546.5	1.5343	0.215	0.040878	36	2	
salmx	4071.7	4.3419	0.02	0.10763	36	2	
salql	1697.7	1.6915	0.175	0.044877	36	2	
Depth	2664.5	2.7277	0.085	0.070433	36	2	*
Mud	3349.8	3.4974	0.052	0.088547	36	2	*
Chla	198.4	0.18979	0.83	0.005244	36	2	*
MxCURRENTS	651.11	0.63045	0.53	0.017211	36	2	
mxWaves	3893	4.1295	0.031	0.1029	36	2	*
East	5114.8	5.6282	0.007	0.1352	36	2	*
North	3147.8	3.2673	0.058	0.083207	36	2	

Tasman and Golden Bays, epifaunal prey production

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df	Final model
Teffort	298.16	0.25289	0.793	0.006976	36	2	
OEffort	2818.6	2.5415	0.1	0.065943	36	2	
SEffort	399.42	0.33958	0.746	0.009345	36	2	
SEav6yr	300.51	0.25489	0.822	0.007031	36	2	*
Tmarks	855.14	0.73492	0.45	0.020006	36	2	*
salmin	1152.6	0.99764	0.328	0.026965	36	2	
salmx	4848.3	4.6058	0.014	0.11343	36	2	
salql	1259.5	1.093	0.307	0.029467	36	2	
Depth	2752.7	2.478	0.091	0.064401	36	2	*
Mud	3411.3	3.1223	0.046	0.079809	36	2	*
Chla	568.56	0.48531	0.617	0.013302	36	2	*
MxCURRENTS	1225.2	1.0623	0.344	0.028664	36	2	
mxWaves	3294.6	3.0065	0.048	0.077078	36	2	*
East	4679.2	4.4254	0.014	0.10947	36	2	*
North	3418	3.129	0.044	0.079966	36	2	

Tasman and Golden Bays, infaunal community composition

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df	Final model
salmin	3832.6	2.2827	0.011	0.059628	36	2	*
salmx	2839	1.6636	0.06	0.04417	36	2	*
salql	6285.6	3.9022	0.001	0.097793	36	2	*
Depth	9841.5	6.5088	0.001	0.15312	36	2	
Mud	6085.6	3.765	0.001	0.09468	36	2	*
Chla	1807.8	1.0418	0.377	0.028126	36	2	
MxCURRENTS	5230.4	3.189	0.001	0.081376	36	2	
East	4729.8	2.8596	0.004	0.073587	36	2	*
North	3745.8	2.2278	0.01	0.058278	36	2	*
mxWaves	6627.7	4.1389	0.002	0.10312	36	2	*
Teffort	3384.4	2.0009	0.024	0.052655	36	2	*
OEffort	3474.5	2.0572	0.015	0.054056	36	2	*
SEffort	4239.1	2.542	0.005	0.065953	36	2	
SEav6yr	4035.6	2.4117	0.01	0.062787	36	2	*
Tmarks	1426.8	0.81727	0.649	0.022198	36	2	
FG6	3079.3	1.8115	0.033	0.047908	36	2	*

Tasman and Golden Bays, infaunal species richness

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df	Final model
Teffort	1861.5	20.195	0.001	0.35937	36	2	*
OEffort	106.23	0.75372	0.402	0.020507	36	2	*
SEffort	109.68	0.77874	0.374	0.021174	36	2	
SEav6yr	772.57	6.3103	0.019	0.14914	36	2	*
Tmarks	114.84	0.81618	0.406	0.022169	36	2	
FG6	102.95	0.72998	0.372	0.019874	36	2	*
mxWaves	351.9	2.6239	0.088	0.067934	36	2	
salmin	859.35	7.1601	0.008	0.1659	36	2	
salmx	106.11	0.75287	0.422	0.020485	36	2	
salql	1080.8	9.4913	0.005	0.20864	36	2	*
Depth	759.91	6.1891	0.018	0.1467	36	2	
Mud	7.8411	0.054576	0.935	0.001514	36	2	
Chla	17.059	0.11895	0.82	0.003293	36	2	
MxCURRENTS	504.57	3.8851	0.041	0.097407	36	2	*
East	130.66	0.93158	0.342	0.025225	36	2	*
North	128.75	0.91758	0.368	0.024855	36	2	

Tasman and Golden Bays, infaunal diversity (Shannon-Weiner)

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df	Final model
Teffort	119.56	2.4925	0.111	0.064753	36	2	*
OEffort	45.632	0.91223	0.319	0.024714	36	2	*
SEffort	9.0065	0.17646	0.67	0.004878	36	2	*
SEav6yr	46.343	0.92683	0.331	0.025099	36	2	*
Tmarks	5.1697	0.10108	0.772	0.0028	36	2	*
FG6	85.894	1.7564	0.204	0.046519	36	2	*
mxWaves	57.401	1.1551	0.301	0.031087	36	2	
salmin	15.68	0.30833	0.588	0.008492	36	2	
salmx	2.1737	0.042431	0.879	0.001177	36	2	
salql	4.2238	0.08254	0.81	0.002288	36	2	*
Depth	12.804	0.25139	0.638	0.006935	36	2	
Mud	22.363	0.44135	0.529	0.012111	36	2	*
Chla	0.76781	0.014976	0.948	0.000416	36	2	
MxCURRENTS	0.59173	0.011541	0.957	0.00032	36	2	
East	52.355	1.0506	0.298	0.028355	36	2	*
North	23.418	0.46246	0.525	0.012683	36	2	

Tasman and Golden Bays, infaunal evenness

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df	Final model
Teffort	8.5853	0.21106	0.663	0.005829	36	2	*
OEffort	98.436	2.5781	0.128	0.066828	36	2	*
SEffort	0.98449	0.024077	0.901	0.000668	36	2	*
SEav6yr	242	7.0774	0.009	0.1643	36	2	*
Tmarks	1.5819	0.038704	0.875	0.001074	36	2	*
FG6	46.446	1.1721	0.261	0.031532	36	2	*
mxWaves	171.01	4.7284	0.036	0.1161	36	2	
salmin	29.995	0.74831	0.398	0.020363	36	2	*
salmx	14.607	0.36058	0.534	0.009917	36	2	
salql	68.433	1.754	0.209	0.046459	36	2	
Depth	143.27	3.8787	0.044	0.097263	36	2	
Mud	21.859	0.54229	0.479	0.01484	36	2	
Chla	0.26439	0.006463	0.983	0.000179	36	2	
MxCURRENTS	59.034	1.503	0.237	0.040078	36	2	
East	107.26	2.8274	0.093	0.072819	36	2	
North	1.2275	0.030026	0.896	0.000833	36	2	*

Tasman and Golden Bays, infaunal biomass

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df	Final model
Teffort	343.86	1.5613	0.213	0.042703	35	2	
OEffort	1104	5.5609	0.021	0.1371	35	2	
SEffort	966.24	4.7725	0.035	0.12	35	2	
SEav6yr	1206	6.1655	0.021	0.14977	35	2	
Tmarks	992.02	4.9177	0.024	0.1232	35	2	*
FG6	267.28	1.2017	0.276	0.033193	35	2	*
mxWaves	2620.8	16.888	0.001	0.32547	35	2	*
salmin	625.35	2.947	0.09	0.077661	35	2	
salmx	105.91	0.46648	0.519	0.013153	35	2	*
salql	2059.8	12.031	0.001	0.25581	35	2	*
Depth	3330.8	24.691	0.001	0.41365	35	2	*
Mud	393.01	1.7959	0.185	0.048807	35	2	*
Chla	778.98	3.7485	0.053	0.096739	35	2	*
MxCURRENTS	1211.8	6.2001	0.02	0.15049	35	2	
East	1751.5	9.729	0.001	0.21751	35	2	
North	41.959	0.18333	0.713	0.005211	35	2	*

Tasman and Golden Bays, infaunal prey

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df	Final model
Teffort	1427.2	5.851	0.017	0.13981	36	2	*
OEffort	1059.7	4.17	0.047	0.10381	36	2	
SEffort	817.69	3.1346	0.078	0.080098	36	2	
SEav6yr	2024.6	8.9056	0.003	0.19832	36	2	
Tmarks	877.58	3.3858	0.065	0.085964	36	2	*
FG6	183.91	0.66043	0.444	0.018015	36	2	*
mxWaves	2484	11.576	0.002	0.24332	36	2	*
salmin	1522.2	6.3087	0.018	0.14911	36	2	*
salmx	13.428	0.047413	0.934	0.001315	36	2	*
salql	3230.7	16.668	0.002	0.31647	36	2	*
Depth	4602.5	29.555	0.001	0.45084	36	2	*
Mud	859.85	3.3111	0.061	0.084227	36	2	*
Chla	2013.2	8.8434	0.007	0.19721	36	2	*
MxCURRENTS	2897	14.264	0.001	0.28378	36	2	*
East	1274.7	5.1366	0.026	0.12487	36	2	*
North	98.747	0.35162	0.58	0.009673	36	2	*

Tasman and Golden Bays, infaunal production

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df	Final model
Teffort	971.5	9.0963	0.002	0.20171	36	2	*
OEffort	686.79	5.9871	0.019	0.14259	36	2	
SEffort	341.93	2.7511	0.106	0.070994	36	2	
SEav6yr	868.92	7.9244	0.014	0.18041	36	2	
Tmarks	414.2	3.3872	0.066	0.085998	36	2	*
FG6	47.965	0.36213	0.59	0.009959	36	2	*
mxWaves	1277.5	12.995	0.001	0.26524	36	2	
salmin	1003.8	9.4779	0.004	0.20841	36	2	
salmx	76.021	0.57734	0.442	0.015784	36	2	*
salql	1871.4	22.877	0.001	0.38856	36	2	
Depth	1874.2	22.932	0.001	0.38913	36	2	*
Mud	580.39	4.9326	0.025	0.12051	36	2	*
Chla	728.68	6.4175	0.016	0.15129	36	2	*
MxCURRENTS	1063.2	10.198	0.004	0.22074	36	2	
East	772.96	6.882	0.014	0.16049	36	2	
North	4.0348	0.030184	0.925	0.000838	36	2	

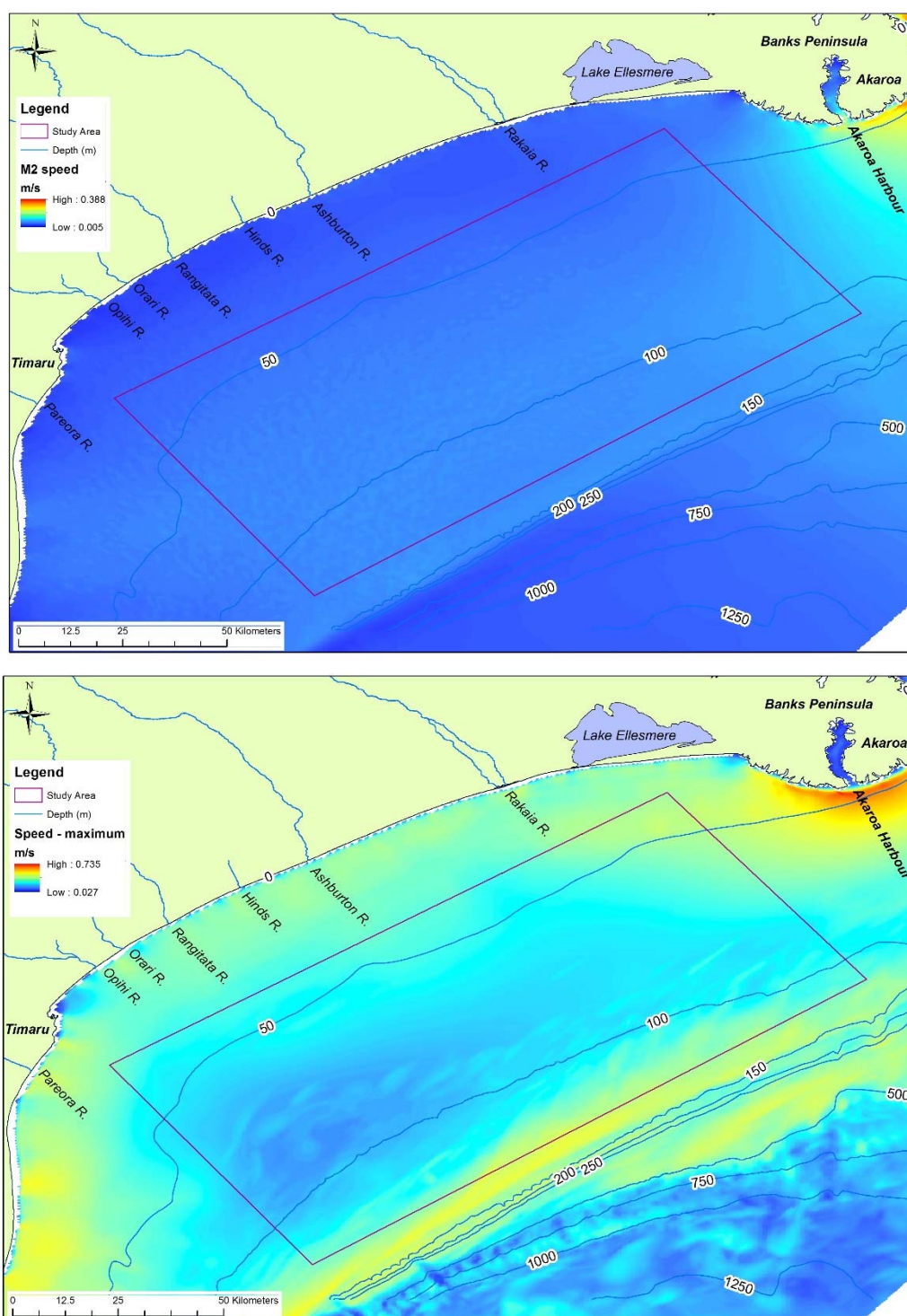
Tasman and Golden Bays, infaunal prey production

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df	Final model
Teffort	1118.4	9.5828	0.003	0.21023	36	2	*
OEffort	626.19	4.8026	0.029	0.1177	36	2	
SEffort	345.04	2.4968	0.123	0.064856	36	2	*
SEav6yr	935.86	7.6846	0.007	0.17591	36	2	
Tmarks	401.21	2.9364	0.097	0.075414	36	2	*
FG6	57.599	0.39403	0.54	0.010827	36	2	
mxWaves	1127.6	9.6829	0.003	0.21196	36	2	
salmin	1211.3	10.613	0.003	0.22768	36	2	*
salmx	12.89	0.087436	0.85	0.002423	36	2	*
salql	2076.6	23.049	0.001	0.39033	36	2	*
Depth	2131.3	24.061	0.001	0.40061	36	2	*
Mud	589.14	4.4831	0.034	0.11074	36	2	*
Chla	1157.8	10.014	0.003	0.21763	36	2	*
MxCURRENTS	1588.5	15.324	0.002	0.29858	36	2	*
East	582.05	4.4225	0.035	0.10941	36	2	*
North	46.637	0.31837	0.611	0.008766	36	2	*

12. APPENDIX 2: Environmental data used to stratify South Canterbury Bight study area

Currents

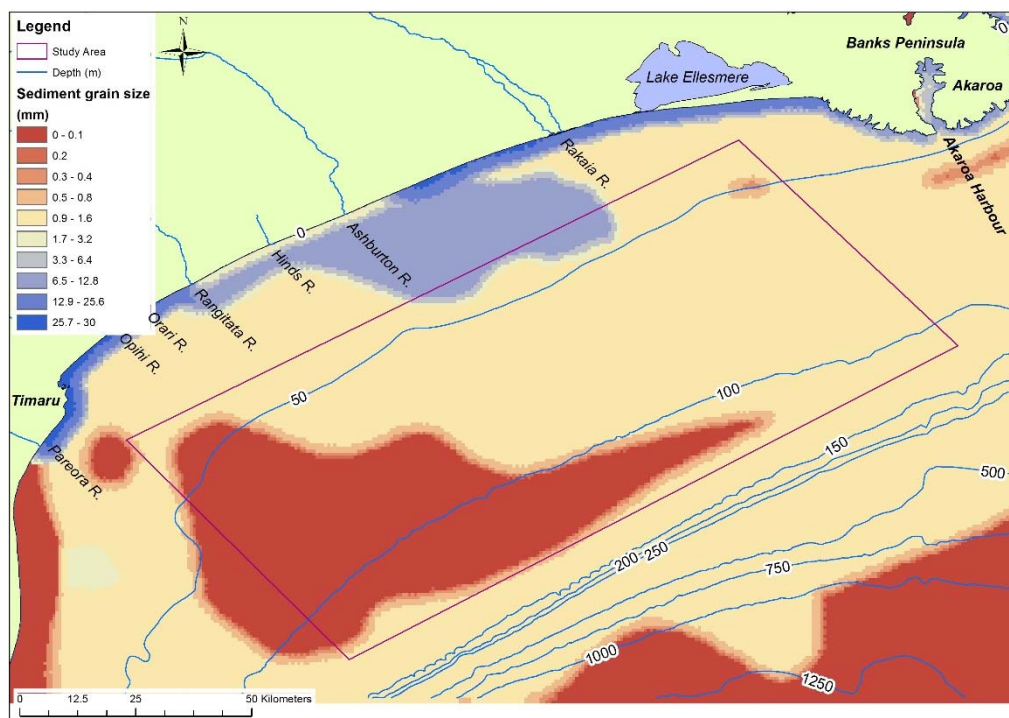
A 3-dimensional tidal model with 20 levels (but neutrally stratified and with no drivers other than the M2 or principal lunar semi-diurnal constituent tide) was run for 12 lunar tidal cycles (6.21 days). The tidal currents are the maximum M2 and maximum speed (A1. 1) at each grid point (spacing is 500 m) at the level nearest the bottom over the last 6 tidal cycles. In the Canterbury Bight region the S2 (solar) constituent is weak, so there is not the normal semi-monthly (M2+S2) amplitude variation. The speed of the M2 constituent at the South Canterbury Bight is 0.28 m.s^{-1} .



A1. 1: Tidal model output of maximum M2 current (upper plot) and maximum tidal current (lower plot).

Sediment

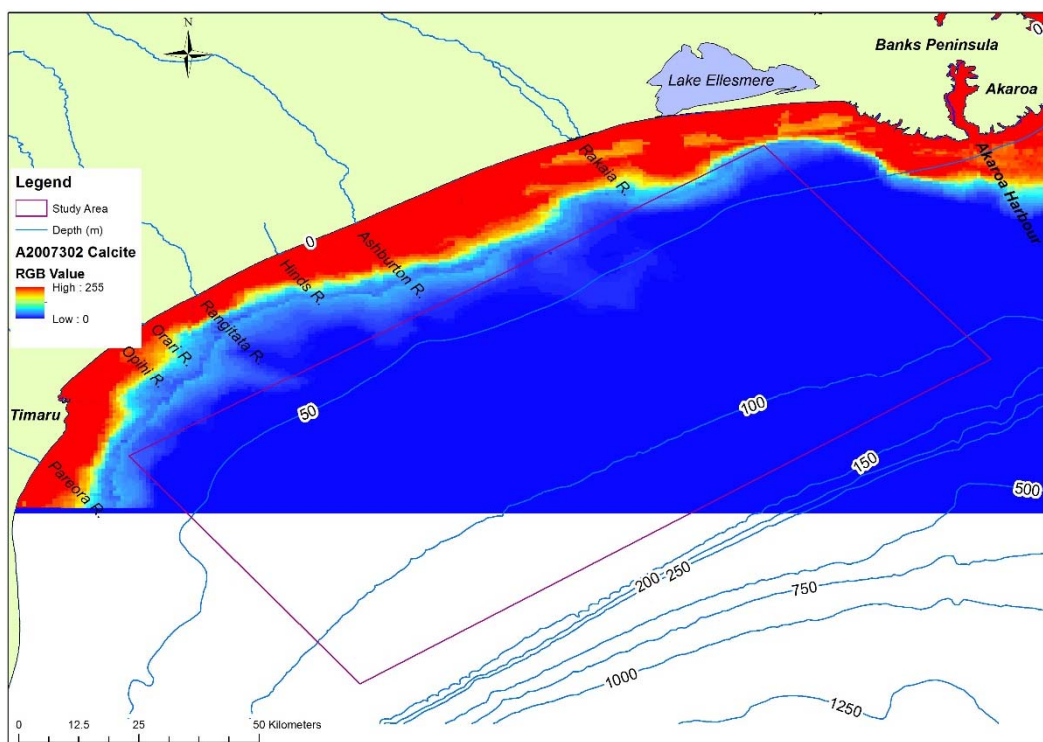
The sediment characteristics were derived from digitization of Mitchell et al. (1989), a regional sediment chart as described in Appendix 1 of Leathwick et al. (2012). Median grain size was extracted across a 1 km grid placed atop, and interpolated using kriging in ArcMap™ 10.0 (ESRI®). Coarser sediments are present along the coastal fringe, but only extend into the margins of the study area between the Ashburton and Rakaia rivers. The finest sediments in the study area are found towards the southwest (A1. 2).



A1. 2: Mean grain size derived from digitisation of Mitchell et al. (1989).

Suspended sediments

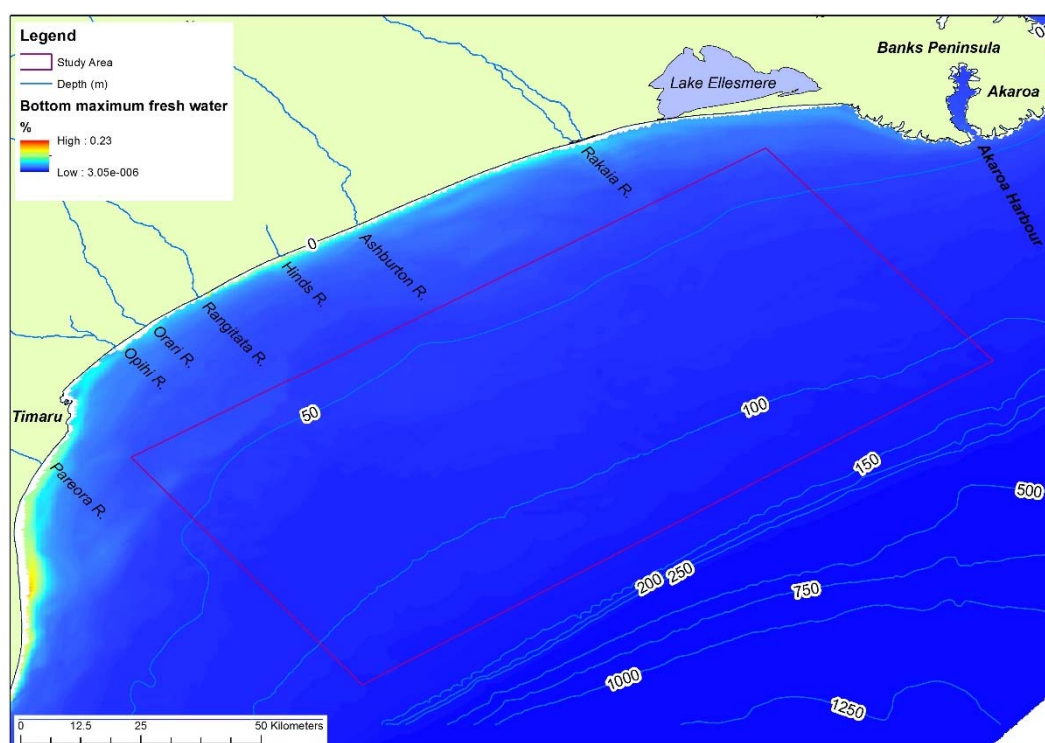
Ten rivers in the Canterbury Bight, as well as Lake Ellesmere, were studied using a core, cloud-free dataset of colour satellite imagery from the NASA MODIS-Aqua sensor (Schwarz et al. 2009; 2010). Freshwater extent was measured wherever neighbouring river plumes could be distinguished from one another based on gradients in light scattering derived from the satellite data. Optical and biogeochemical products were derived using NASA algorithms, specifically calcite, chlorophyll, particulate backscatter at 555 nm. Upon closer examination, only the calcite and chlorophyll algorithms were found to function robustly near-shore. The calcite product (A1. 3) was chosen as a potential means of distinguishing recently arrived riverine sediments from re-suspended coastal sediments (Schwarz et al. 2009), and generally shows uniformly very low levels within the study area. As the area of the satellite image did not cover the full study area, sampling stations that did not overlap with the image were given a nominal zero value. This was considered appropriate as calcite values for offshore areas were zero (A1. 3).



A1. 3: Satellite colour imagery analysis of calcite (RGB values), interpreted as a measure of recently arrived riverine sediments.

Bottom salinity

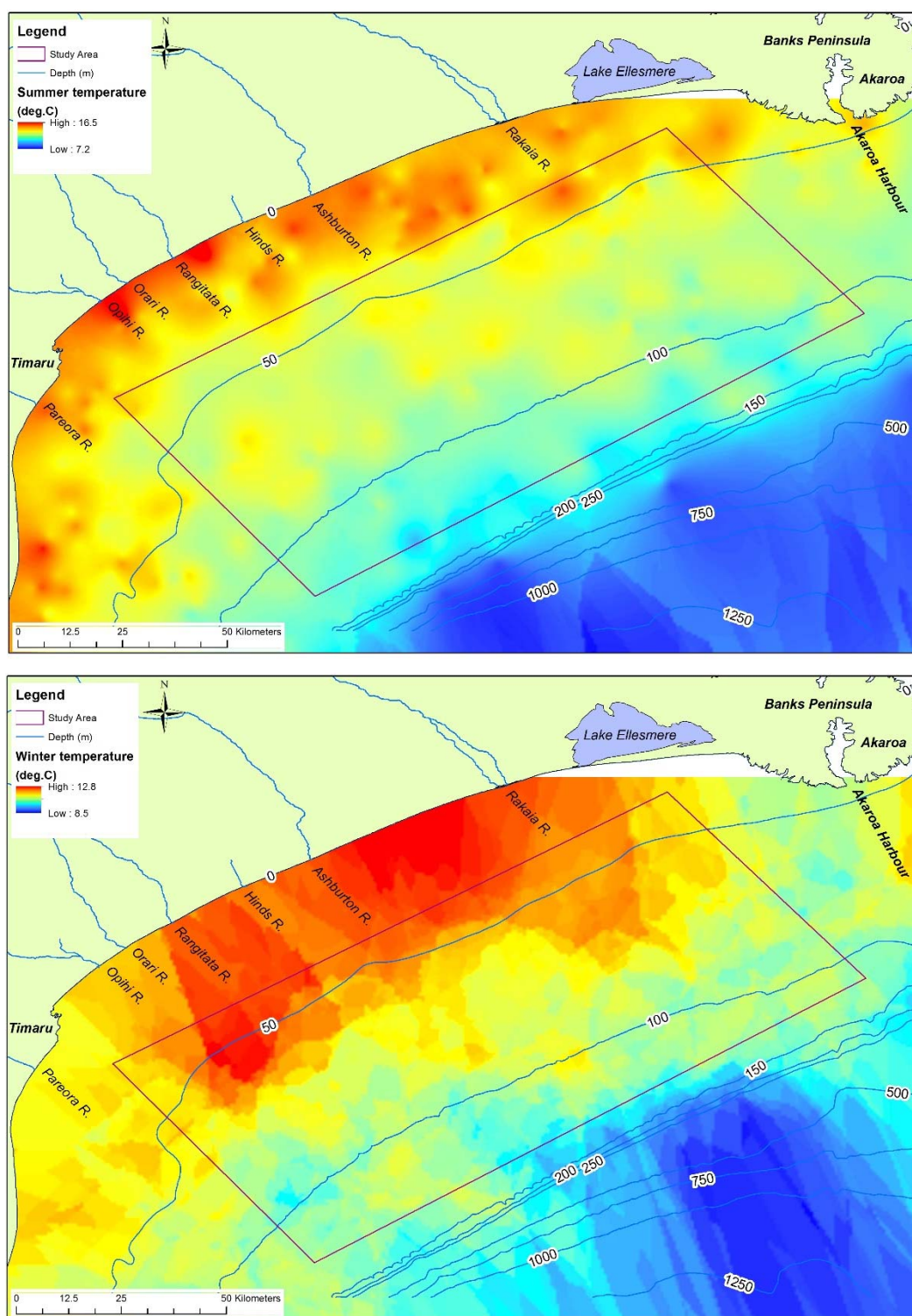
Bottom salinity (A1. 4) was estimated from a model of freshwater dispersion and transport in the Canterbury Bight was run for a period of one year (April 2009–April 2010), forced with ocean currents, winds, tides and freshwater inputs from 10 rivers in Southland, Otago and Canterbury, along with Lake Ellesmere/Te Waihora (Hadfield & Zeldis 2012). The freshwater sources in Canterbury were each labelled with a separate tracer, allowing the concentration of river-derived freshwater to be evaluated and attributed to the specific source. The model was validated in July and December 2011 using an Acoustic Doppler Current Profiler (ADCP) time series at two locations and two surveys with conductivity-temperature-depth (CTD) instruments. The model validation confirmed that the model was simulating currents in the area very well and that the behaviour of the surface freshwater layer was realistic, though it did not reproduce the small-scale variability on the in-shore transect during the CTD surveys. The coastal freshwater band, and the river plumes within it, generally move north-eastward but with substantial fluctuations that are largely wind driven. Under winds from the south or southwest, the coastal freshwater band can move quite quickly (within a few days) north-eastward along the coast of Canterbury Bight and around the end of Banks Peninsula. The model indicated substantially higher freshwater concentrations on the southern and eastern sides of Banks Peninsula than on the northern side.



A1. 4: Bottom salinity estimated from modelled freshwater dispersion (Hadfield & Zeldis 2012).

Temperature

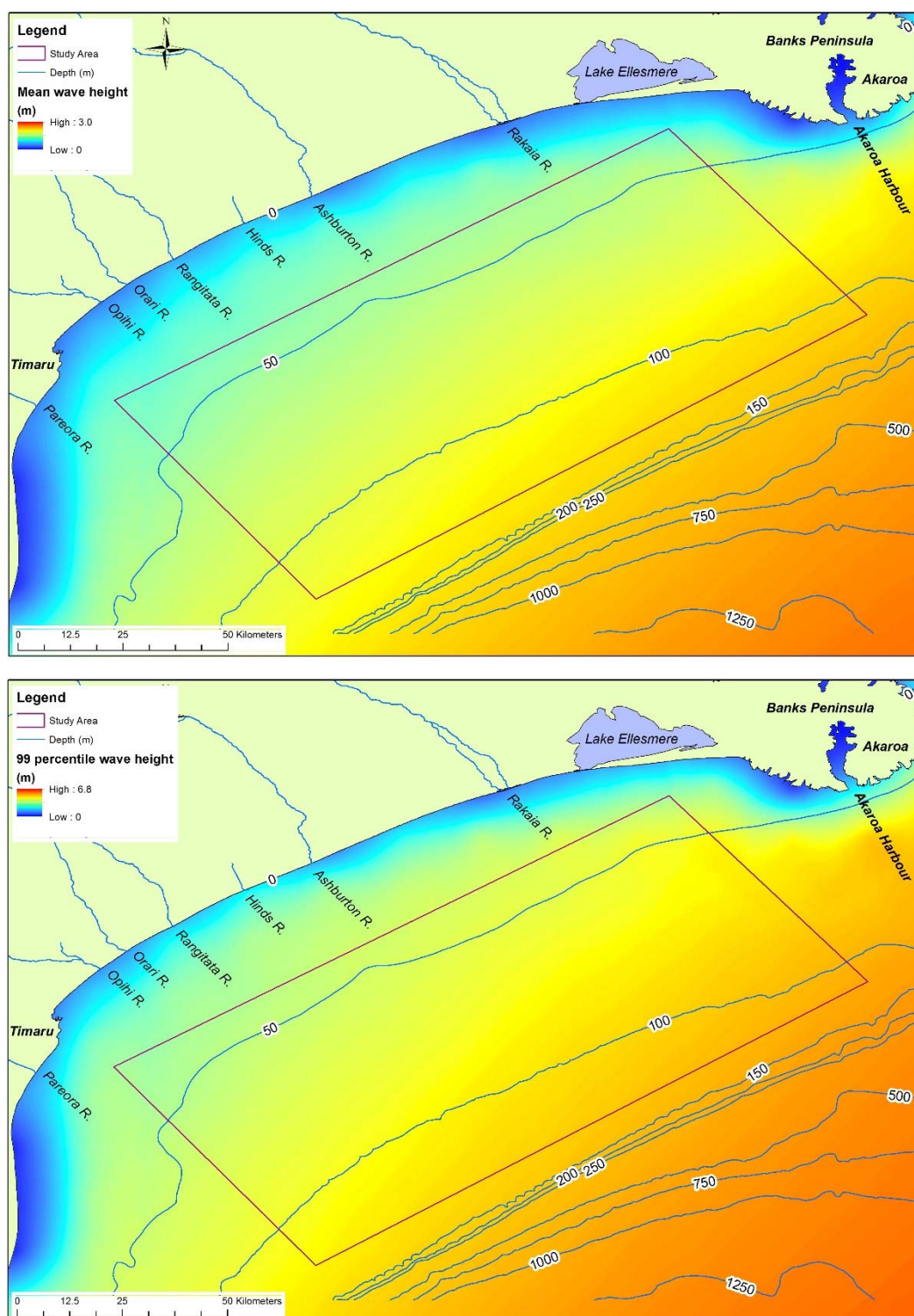
Bottom temperature from NIWA *Kaharoa* trawl survey database were extracted between May 1991 and June 2012. Summer and winter measurements were interpolated separately using kriging in ArcMap™ 10.0 (ESRI®). While temperatures are generally warmer in summer, the relatively warmer coastal temperatures appear to extend further offshore in winter (A1. 5).



A1. 5: Summer (upper plot) and winter (lower plot) bottom water temperature interpolated from observations made during Kaharoa research trawl surveys, 1991–2012. Note, the two plots have different colour scales.

Waves

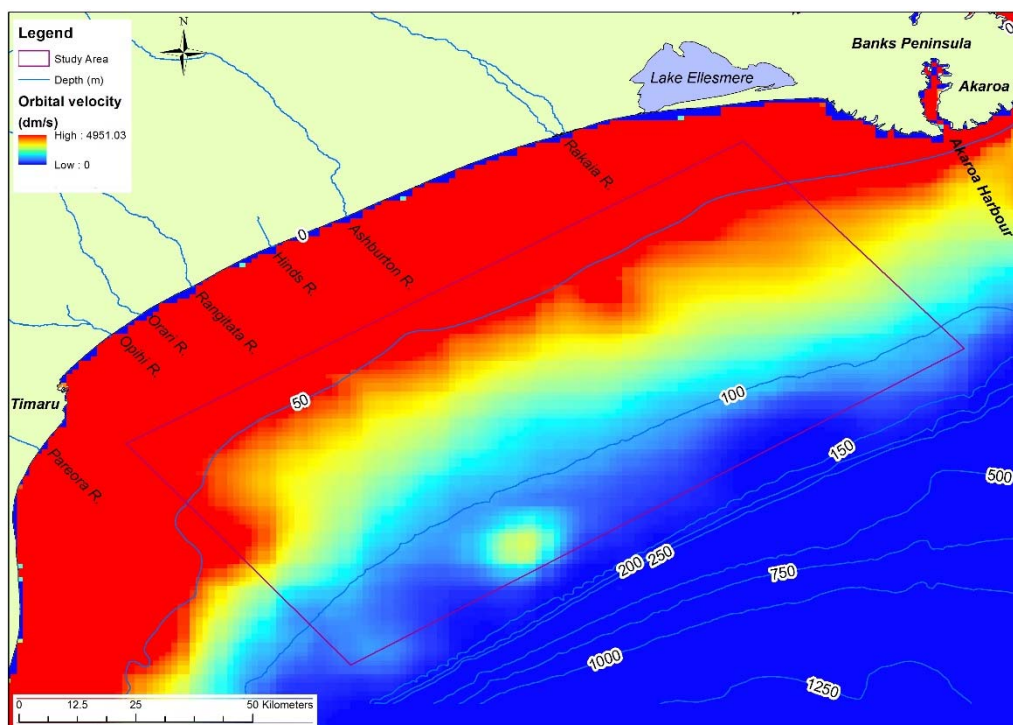
Wave modelling was carried out using NIWA’s operational forecasting system NZWAVE_12, which incorporates wind inputs from the weather forecasting model NZLAM_12, both of which have a horizontal grid spacing of 12 km, and both are nested in coarser-scale global models. The output are the mean and 99th percentile modelled wave height, and both data sets show very similar patterns (A1. 6).



A1. 6: Mean (upper plot) and 99th percentile of modelled wave height.

Orbital velocity

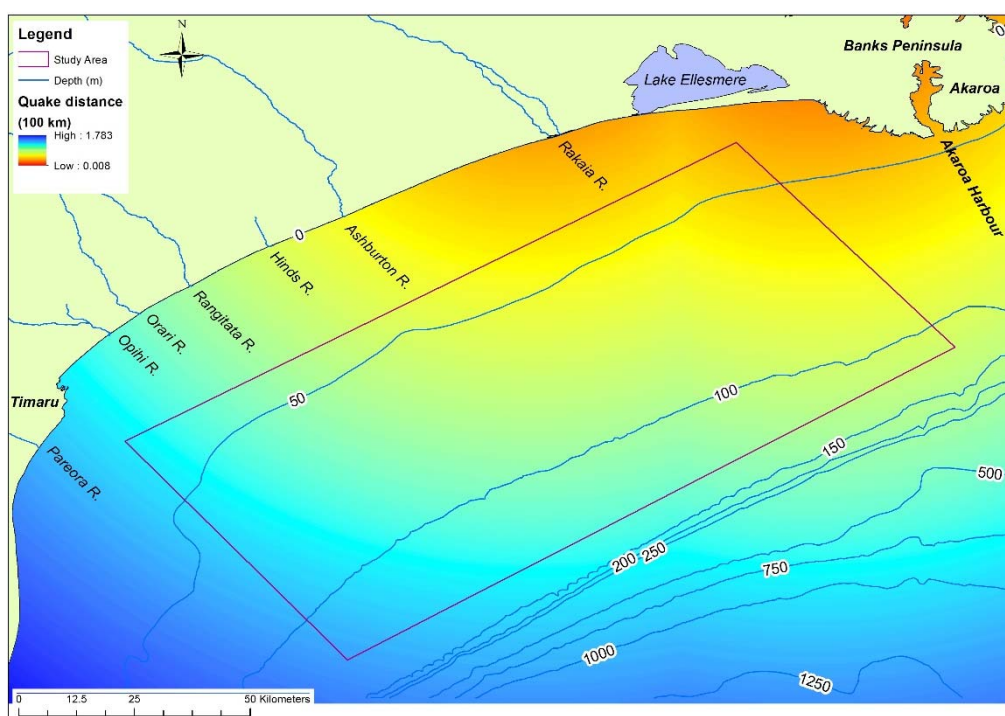
Orbital velocity at the seabed (A1. 7) was based on a 250 m grid wave climatology model (Hadfield et al. 2002), derived from a 20 year hindcast (1979–1998) of mean significant wave height for the New Zealand region (Gorman et al. 2003).



A1. 7: Modelled seabed orbital wave velocity.

Distance to quake epicentres

Following the swarm of earthquakes in the Canterbury region in 2010–2011, there was concern that this seismic activity may have affected fishing and seabed communities in the Canterbury Bight, potentially confounding the results. This theory was put to an experienced fisherman in the region, and his response was that “if anything, fishing had improved over the last couple of years. English sole appears to have disappeared, whereas flounder have increased, with an average of 15 bins per day” (Raymond Mitchel, pers. comm.). To cover any potential unmeasured effect on benthic species, the epicentres of the magnitude 7.1 earthquake at Darfield on 4th September 2010 and the 6.3 magnitude earthquake 5 km south east of Christchurch on 22 February 2011 were plotted. The distance of each of these epicentres to a 1 km grid overlaid over the study area were then calculated. These distances were then interpolated using kriging in ArcMap™ 10.0 (ESRI®). This plot shows a gradient increasing towards the epicentres (A1. 8).



A1. 8: Distance from major earthquake epicentres.

13. APPENDIX 3: Marginal tests for DistLM models for the South Canterbury Bight case study

These show how much of the total variance each variable explains when taken alone, ignoring all other variables. All variables available to the full model are presented, with those retained in the final model identified. P represents significance of term, and Prop. represents the proportion of total variance explained by that term.

South Canterbury Bight, epifaunal community composition

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df	Final model
Fish	4170.4	1.5384	0.086	0.05208	28	2	*
Depth	18389	8.3468	0.001	0.22964	28	2	*
M2	13167	5.51	0.001	0.16443	28	2	*
mxcurr	5955.2	2.2496	0.02	0.074369	28	2	*
median	4840.4	1.8014	0.037	0.060447	28	2	*
Wmean	15181	6.5502	0.001	0.18959	28	2	*
W99	12928	5.3906	0.001	0.16144	28	2	*
FW	14422	6.1506	0.001	0.1801	28	2	*
Quake	8829	3.4698	0.004	0.11026	28	2	*
WinT	16156	7.0769	0.001	0.20175	28	2	
SumT	16247	7.127	0.001	0.20289	28	2	*
vel	14440	6.1599	0.001	0.18032	28	2	
Moisture	8699.5	3.4127	0.003	0.10864	28	2	*
LOI	12030	4.95	0.001	0.15023	28	2	*
Mud	11252	4.5777	0.001	0.14052	28	2	*
Sand	7735.7	2.9942	0.004	0.096604	28	2	*
Chla	6810.3	2.6027	0.005	0.085048	28	2	*
North	11222	4.5634	0.001	0.14014	28	2	*
East	5521.4	2.0736	0.021	0.068952	28	2	*

South Canterbury Bight, epifaunal species richness

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df	Final model
Fish	655.25	1.3059	0.249	0.044561	28	2	*
Depth	1646.9	3.5316	0.06	0.112	28	2	
M2	1521.3	3.231	0.073	0.10345	28	2	*
mxcurr	279.44	0.5424	0.533	0.019003	28	2	
median	967.26	1.9715	0.154	0.06578	28	2	*
Wmean	1194.7	2.476	0.118	0.081245	28	2	*
W99	877.16	1.7762	0.191	0.059652	28	2	*
FW	859.13	1.7374	0.19	0.058426	28	2	*
Quake	1308.7	2.7354	0.1	0.088997	28	2	*
WinT	2120.5	4.7183	0.027	0.14421	28	2	*
SumT	1787.7	3.8752	0.045	0.12158	28	2	*
vel	669.62	1.3359	0.259	0.045538	28	2	*
Moisture	1465.3	3.099	0.073	0.09965	28	2	
LOI	1003.1	2.0499	0.118	0.068216	28	2	*
Mud	886.66	1.7967	0.185	0.060298	28	2	
Sand	102.1	0.19577	0.765	0.006943	28	2	*
Chla	171.79	0.33099	0.638	0.011683	28	2	
North	1112.5	2.2917	0.093	0.075654	28	2	*
East	35.269	0.067319	0.915	0.002399	28	2	*

South Canterbury Bight, epifaunal diversity (Shannon-Weiner)

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df	Final model
Fish	1332	7.5465	0.005	0.2123	28	2	*
Depth	74.543	0.33666	0.586	0.011881	28	2	*
M2	24.144	0.10816	0.834	0.003848	28	2	*
mxcurr	376.15	1.7857	0.198	0.059951	28	2	*
median	185.96	0.85525	0.348	0.029639	28	2	*
Wmean	189.3	0.87105	0.364	0.03017	28	2	*
W99	233.64	1.083	0.295	0.037238	28	2	*
FW	364.56	1.7273	0.173	0.058104	28	2	*
Quake	71.402	0.32231	0.631	0.01138	28	2	
WinT	82.606	0.37356	0.599	0.013166	28	2	
SumT	48.269	0.21708	0.725	0.007693	28	2	*
vel	661.02	3.2973	0.069	0.10535	28	2	*
Moisture	215.75	0.99713	0.327	0.034387	28	2	
LOI	65.707	0.29633	0.649	0.010473	28	2	*
Mud	26.499	0.11876	0.825	0.004223	28	2	*
Sand	31.425	0.14094	0.841	0.005009	28	2	*
Chla	1442.2	8.3571	0.005	0.22986	28	2	*
East	18.539	0.082979	0.884	0.002955	28	2	*
North	44.872	0.20169	0.719	0.007152	28	2	*

South Canterbury Bight, epifaunal evenness

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df	Final model
Fish	646.86	4.2505	0.043	0.1318	28	2	
Depth	435.36	2.7254	0.078	0.088703	28	2	*
M2	397.93	2.4704	0.123	0.081077	28	2	
mxcurr	732.68	4.9133	0.032	0.14928	28	2	*
median	0.85307	0.004868	0.997	0.000174	28	2	*
Wmean	686.67	4.5546	0.031	0.13991	28	2	
W99	667.57	4.4079	0.035	0.13601	28	2	*
FW	909.52	6.3689	0.01	0.18531	28	2	
Quake	108.38	0.63225	0.461	0.022082	28	2	*
WinT	131.03	0.76803	0.401	0.026697	28	2	
SumT	344.43	2.1132	0.15	0.070176	28	2	
vel	1230.6	9.3695	0.002	0.25073	28	2	*
Moisture	66.133	0.38243	0.599	0.013474	28	2	*
LOI	57.55	0.33221	0.677	0.011726	28	2	*
Mud	107.01	0.62408	0.495	0.021803	28	2	*
Sand	62.44	0.3608	0.682	0.012722	28	2	
Chla	982.18	7.0051	0.012	0.20012	28	2	*
North	175.76	1.0399	0.325	0.03581	28	2	*
East	60.939	0.35202	0.654	0.012416	28	2	*

South Canterbury Bight, epifaunal biomass

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df	Final model
Depth	358.75	9.5214	0.009	0.25376	28	2	
M2	188.83	4.3165	0.04	0.13357	28	2	*
mxcurr	343.88	9	0.004	0.24324	28	2	*
median	77.161	1.6165	0.231	0.05458	28	2	*
Wmean	239.93	5.7233	0.018	0.16971	28	2	*
W99	137.32	3.0123	0.088	0.097133	28	2	*
FW	294.25	7.3597	0.011	0.20814	28	2	*
Quake	411.9	11.512	0.002	0.29135	28	2	
WinT	293.98	7.3511	0.012	0.20794	28	2	*
SumT	471.7	14.02	0.002	0.33365	28	2	*
vel	404.58	11.226	0.002	0.28618	28	2	*
Fish	7.1018	0.14137	0.745	0.005023	28	2	
Moisture	168.55	3.7901	0.058	0.11922	28	2	
LOI	221.29	5.1961	0.031	0.15653	28	2	*
Mud	269.47	6.5939	0.009	0.19061	28	2	*
Sand	88.615	1.8724	0.204	0.062681	28	2	*
Chla	1.6637	0.03299	0.892	0.001177	28	2	
East	27.322	0.55179	0.44	0.019326	28	2	
North	450.48	13.095	0.001	0.31865	28	2	*

South Canterbury Bight, epifaunal prey

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df	Final model
Fish	9.048	0.14703	0.731	0.005224	28	2	
Depth	628.42	15.942	0.002	0.36279	28	2	*
M2	416.31	8.8587	0.006	0.24034	28	2	*
mxcurr	385.86	8.025	0.008	0.22276	28	2	*
median	119.91	2.0825	0.151	0.069225	28	2	*
Wmean	482.39	10.807	0.003	0.27849	28	2	*
W99	319.21	6.3256	0.019	0.18428	28	2	*
FW	495.08	11.205	0.003	0.28581	28	2	
Quake	459.44	10.108	0.006	0.26524	28	2	*
WinT	485.44	10.902	0.005	0.28025	28	2	*
SumT	694.72	18.75	0.001	0.40107	28	2	*
vel	632.78	16.116	0.002	0.36531	28	2	*
Moisture	242.63	4.561	0.035	0.14008	28	2	*
LOI	367.69	7.5452	0.011	0.21227	28	2	*
Mud	391.12	8.1664	0.006	0.2258	28	2	*
Sand	140.6	2.4735	0.121	0.081169	28	2	*
Chla	1.114	0.018019	0.938	0.000643	28	2	*
East	4.6942	0.076087	0.826	0.00271	28	2	
North	530.36	12.356	0.002	0.30618	28	2	*

South Canterbury Bight, epifaunal productivity

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df	Final model
Fish	0.76123	0.017912	0.949	0.000639	28	2	
Depth	292.25	9.1078	0.004	0.24544	28	2	
M2	163.54	4.4579	0.054	0.13735	28	2	*
mxcurr	279.73	8.5976	0.007	0.23492	28	2	*
median	72.637	1.8191	0.185	0.061003	28	2	*
Wmean	190.72	5.3402	0.034	0.16017	28	2	*
W99	101.94	2.6217	0.106	0.085617	28	2	*
FW	221.51	6.3993	0.017	0.18603	28	2	
Quake	372.27	12.736	0.004	0.31264	28	2	*
WinT	257.08	7.7098	0.009	0.2159	28	2	*
SumT	374.33	12.839	0.003	0.31437	28	2	*
vel	316.43	10.134	0.006	0.26575	28	2	
Moisture	142.36	3.8023	0.054	0.11956	28	2	
LOI	180.37	4.9985	0.035	0.15148	28	2	*
Mud	218.32	6.2864	0.016	0.18335	28	2	*
Sand	64.637	1.6072	0.22	0.054284	28	2	*
Chla	0.34397	0.008091	0.969	0.000289	28	2	
East	26.524	0.63792	0.457	0.022275	28	2	
North	392.51	13.769	0.002	0.32964	28	2	*

South Canterbury Bight, epifaunal prey productivity

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df	Final model
Fish	0.9613	0.019354	0.938	0.000691	28	2	
Depth	468.52	14.211	0.002	0.33666	28	2	*
M2	317.55	8.2778	0.007	0.22818	28	2	*
mxcurr	302.24	7.7679	0.012	0.21718	28	2	*
median	101.43	2.2011	0.174	0.07288	28	2	*
Wmean	347.16	9.306	0.008	0.24945	28	2	*
W99	216.26	5.1516	0.033	0.15539	28	2	*
FW	349.29	9.3824	0.002	0.25098	28	2	
Quake	403.46	11.431	0.004	0.28991	28	2	*
WinT	392.27	10.99	0.007	0.28186	28	2	*
SumT	512.03	16.298	0.001	0.36792	28	2	*
vel	459.48	13.801	0.001	0.33016	28	2	*
Moisture	189.67	4.4183	0.047	0.13629	28	2	*
LOI	272.06	6.8037	0.019	0.19549	28	2	*
Mud	294.2	7.506	0.009	0.2114	28	2	*
Sand	95.291	2.0581	0.144	0.068472	28	2	*
Chla	0.4953	0.009969	0.964	0.000356	28	2	*
East	5.439	0.10986	0.755	0.003908	28	2	*
North	445.13	13.167	0.001	0.31985	28	2	*

South Canterbury Bight, infaunal community composition

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df	Final model
Fish	7073.8	2.8363	0.002	0.091979	28	2	*
FG6	1668	0.62073	0.917	0.021688	28	2	*
Depth	13693	6.0649	0.001	0.17804	28	2	*
M2	10397	4.3771	0.001	0.13519	28	2	*
mxcurr	6566	2.6136	0.004	0.085375	28	2	*
median	8121.8	3.3061	0.001	0.10561	28	2	*
Wmean	11730	5.0393	0.001	0.15253	28	2	*
W99	10028	4.1984	0.001	0.13039	28	2	*
FW	13875	6.1635	0.001	0.18041	28	2	*
Quake	7212	2.8974	0.001	0.093775	28	2	*
WinT	10886	4.6167	0.001	0.14154	28	2	
SumT	13402	5.909	0.001	0.17426	28	2	*
vel	13988	6.2249	0.001	0.18188	28	2	*
Moisture	9100	3.7577	0.001	0.11832	28	2	
LOI	12547	5.4585	0.001	0.16314	28	2	
Mud	12689	5.5325	0.001	0.16499	28	2	*
Sand	10899	4.6231	0.001	0.14171	28	2	*
Chla	6381.9	2.5337	0.003	0.082982	28	2	*
East	4381.9	1.6917	0.038	0.056977	28	2	*
North	9318.3	3.8603	0.001	0.12116	28	2	*

South Canterbury Bight, infaunal species richness

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df	Final model
Fish	28.555	0.093497	0.904	0.003328	28	2	
FG6	387.81	1.3255	0.254	0.0452	28	2	*
Depth	383.37	1.3096	0.274	0.044682	28	2	*
M2	311.4	1.0545	0.333	0.036294	28	2	
mxcurr	24.435	0.07997	0.898	0.002848	28	2	
median	3639.4	20.626	0.003	0.42418	28	2	
Wmean	226.85	0.76041	0.493	0.026439	28	2	
W99	148.58	0.49343	0.594	0.017317	28	2	
FW	518.77	1.8019	0.174	0.060463	28	2	
Quake	281.08	0.94834	0.432	0.03276	28	2	
WinT	742.95	2.6544	0.061	0.086592	28	2	*
SumT	702.65	2.4976	0.064	0.081895	28	2	*
vel	234.37	0.78632	0.397	0.027316	28	2	*
Moisture	1739.2	7.119	0.008	0.20271	28	2	
LOI	323.44	1.0969	0.321	0.037697	28	2	*
Mud	384.15	1.3124	0.257	0.044773	28	2	*
Sand	944.1	3.462	0.053	0.11004	28	2	*
Chla	466.86	1.6112	0.159	0.054413	28	2	
East	32.253	0.10565	0.931	0.003759	28	2	*
North	270.31	0.91082	0.408	0.031504	28	2	*

South Canterbury Bight, infaunal diversity (Shannon-Weiner)

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df	Final model
Fish	7.9355	0.12516	0.785	0.00445	28	2	*
FG6	12.744	0.20154	0.727	0.007147	28	2	*
North	47.911	0.77308	0.419	0.026868	28	2	*
M2	76.309	1.2518	0.278	0.042793	28	2	
mxcurr	4.0198	0.063262	0.872	0.002254	28	2	
median	674.86	17.049	0.002	0.37845	28	2	*
Wmean	26.992	0.43035	0.548	0.015137	28	2	*
W99	15.152	0.23996	0.621	0.008497	28	2	*
FW	69.559	1.1366	0.321	0.039008	28	2	*
Quake	57.995	0.94127	0.351	0.032523	28	2	*
WinT	171.3	2.9756	0.103	0.096064	28	2	*
SumT	129.99	2.2016	0.166	0.072896	28	2	*
vel	25.593	0.40772	0.552	0.014352	28	2	
Moisture	378.63	7.5481	0.012	0.21233	28	2	
LOI	68.794	1.1236	0.32	0.038579	28	2	*
Mud	118.49	1.9929	0.178	0.066446	28	2	*
Sand	88.392	1.4603	0.242	0.04957	28	2	*
Chla	173.44	3.0167	0.085	0.097261	28	2	*
Depth	43.033	0.69243	0.433	0.024133	28	2	*
East	1.5982	0.025117	0.943	0.000896	28	2	*

South Canterbury Bight, infaunal evenness

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df	Final model
Fish	14.267	0.52234	0.474	0.018313	28	2	*
FG6	43.553	1.658	0.202	0.055905	28	2	*
Depth	8.7182	0.31689	0.629	0.011191	28	2	
M2	1.9849	0.071523	0.807	0.002548	28	2	
mxcurr	0.78912	0.028391	0.916	0.001013	28	2	*
median	6.8877	0.24976	0.554	0.008841	28	2	*
Wmean	4.7248	0.17085	0.724	0.006065	28	2	*
W99	3.9515	0.14275	0.751	0.005072	28	2	*
FW	7.139	0.25896	0.647	0.009164	28	2	*
Quake	2.7444	0.098987	0.801	0.003523	28	2	*
WinT	2.8223	0.10181	0.78	0.003623	28	2	
SumT	3.0086	0.10855	0.794	0.003862	28	2	*
vel	5.8218	0.21082	0.679	0.007473	28	2	
Moisture	2.9418	0.10613	0.764	0.003776	28	2	
LOI	2.4059	0.08674	0.821	0.003088	28	2	*
Mud	5.9418	0.2152	0.678	0.007627	28	2	*
Sand	31.266	1.1707	0.309	0.040134	28	2	*
Chla	15.424	0.56557	0.457	0.019799	28	2	*
East	0.3045	0.010948	0.97	0.000391	28	2	
North	2.5784	0.092977	0.81	0.00331	28	2	*

South Canterbury Bight, infaunal biomass

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df	Final model
Fish	591.16	1.7614	0.194	0.059185	28	2	*
FG6	601.3	1.7935	0.177	0.060199	28	2	
Depth	1706.4	5.769	0.016	0.17084	28	2	*
M2	2405.2	8.8806	0.008	0.24079	28	2	*
mxcurr	106.75	0.30247	0.647	0.010687	28	2	*
median	320.73	0.92891	0.36	0.03211	28	2	*
Wmean	2576.7	9.7343	0.003	0.25797	28	2	
W99	2842.3	11.136	0.002	0.28456	28	2	
FW	1576.2	5.2464	0.023	0.1578	28	2	*
Quake	203.37	0.58193	0.495	0.02036	28	2	*
WinT	630.16	1.8854	0.181	0.063089	28	2	
SumT	777.95	2.365	0.124	0.077885	28	2	*
vel	1637	5.4885	0.014	0.16389	28	2	*
Moisture	303.98	0.87886	0.384	0.030433	28	2	
LOI	220.84	0.63305	0.476	0.022109	28	2	*
Mud	158.82	0.4524	0.554	0.0159	28	2	
Sand	664.13	1.9943	0.171	0.066489	28	2	*
Chla	252.69	0.72674	0.419	0.025298	28	2	
East	1757.1	5.9772	0.019	0.17592	28	2	
North	158.57	0.45168	0.539	0.015875	28	2	*

South Canterbury Bight, infaunal prey

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df	Final model
Fish	401.98	1.2846	0.279	0.043867	28	2	*
FG6	693.56	2.2928	0.124	0.075686	28	2	
Depth	1034.6	3.5635	0.063	0.1129	28	2	*
M2	1545.9	5.682	0.022	0.16869	28	2	*
mxcurr	228.87	0.71725	0.409	0.024976	28	2	*
median	208.77	0.65277	0.48	0.022782	28	2	*
Wmean	1386.3	4.991	0.031	0.15128	28	2	
W99	1382	4.9728	0.031	0.15081	28	2	*
FW	925.07	3.144	0.068	0.10095	28	2	*
Quake	142.04	0.44084	0.547	0.0155	28	2	*
WinT	254.39	0.79951	0.357	0.027761	28	2	
SumT	521.29	1.6889	0.211	0.056887	28	2	*
vel	1254.2	4.4399	0.049	0.13686	28	2	
Moisture	283.3	0.89324	0.366	0.030915	28	2	*
LOI	201.85	0.63064	0.449	0.022027	28	2	*
Mud	185.4	0.57819	0.479	0.020232	28	2	*
Sand	705.9	2.3369	0.12	0.077032	28	2	*
Chla	176.99	0.55147	0.472	0.019315	28	2	
East	633.8	2.0805	0.152	0.069165	28	2	*
North	159.46	0.49587	0.497	0.017401	28	2	*

South Canterbury Bight, infaunal productivity

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df	Final model
Depth	779.9	6.8713	0.02	0.19705	28	2	*
East	557.87	4.5942	0.044	0.14095	28	2	*
North	34.367	0.24526	0.637	0.008683	28	2	*
M2	1088.8	10.626	0.005	0.27509	28	2	*
mxcurr	98.857	0.71727	0.442	0.024977	28	2	*
median	12.478	0.088555	0.82	0.003153	28	2	*
Wmean	1070.9	10.386	0.005	0.27057	28	2	
W99	1119.8	11.048	0.001	0.28293	28	2	
FW	834.41	7.4799	0.013	0.21082	28	2	*
Quake	20.172	0.14343	0.748	0.005097	28	2	*
WinT	300.46	2.3002	0.122	0.075914	28	2	*
SumT	425.98	3.377	0.071	0.10763	28	2	*
vel	839.56	7.5385	0.014	0.21212	28	2	
Fish	385.01	3.0173	0.101	0.097277	28	2	*
Moisture	13.685	0.097153	0.81	0.003458	28	2	*
LOI	151.13	1.1116	0.296	0.038184	28	2	*
Mud	93.686	0.67884	0.439	0.023671	28	2	*
Sand	239.63	1.8045	0.171	0.060545	28	2	*
Chla	30.806	0.21965	0.681	0.007783	28	2	*
FG6	150.98	1.1104	0.303	0.038145	28	2	

South Canterbury Bight, infaunal prey productivity

Group	SS(trace)	Pseudo-F	P	Prop.	res.df	regr.df	Final model
Fish	292.45	2.2235	0.12	0.07357	28	2	*
FG6	200.44	1.4869	0.236	0.050425	28	2	
Depth	523.13	4.2432	0.051	0.1316	28	2	*
M2	783.92	6.8782	0.02	0.19721	28	2	*
mxcurr	133.22	0.97094	0.343	0.033514	28	2	*
median	41.123	0.29269	0.643	0.010345	28	2	*
Wmean	671.47	5.691	0.027	0.16892	28	2	
W99	656.99	5.544	0.028	0.16528	28	2	
FW	560.76	4.5985	0.038	0.14107	28	2	*
Quake	24.344	0.17253	0.707	0.006124	28	2	*
WinT	145.52	1.064	0.341	0.036608	28	2	
SumT	309.25	2.3621	0.142	0.077797	28	2	*
vel	651.84	5.492	0.028	0.16398	28	2	
Moisture	15.422	0.10906	0.779	0.00388	28	2	*
LOI	126.74	0.92214	0.346	0.031884	28	2	*
Mud	109.86	0.79583	0.38	0.027637	28	2	*
Sand	221.89	1.6554	0.194	0.055821	28	2	
Chla	17.815	0.12605	0.738	0.004482	28	2	*
East	261.93	1.9751	0.18	0.065892	28	2	*
North	40.282	0.28664	0.629	0.010134	28	2	*