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Designing a programme to monitor trends in deep-water benthic communities

New Zealand Aquatic Environment and Biodiversity Report No. 143

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TABLE OF CONTENTS

EXECUTIVE SUMMARY

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New Zealand has national and international obligations to prevent adverse effects to marine ecosystems caused by bottom-contact fishing. To assess the effectiveness of measures implemented to manage the effects of fishing, it is necessary to monitor temporal trends in the status of benthic habitats and communities. Any monitoring programme that uses consistent methods to collect data at regular intervals over an extended period is likely to generate valuable insights into ecosystem behaviour. To address specific questions, however, such as how benthic habitats are affected by disturbance from fishing, a monitoring programme would need to be carefully specified. In this report, we consider how such a benthic monitoring programme might be designed, what resources would be required, and whether there are opportunities for gathering useful data by augmenting existing fisheries research surveys. Specifically, we consider: (1) what data are currently available on benthic habitats and fauna; (2) what these data tell us about characteristic spatial scales of variability; (3) which attributes of benthic habitats and communities should be monitored; (4) design options for a programme to monitor temporal trends, and (5) the potential costs of such a programme.

The most complete data on deep-water benthic distributions in New Zealand are for Chatham Rise and, to a lesser extent, other regions which have been surveyed under Ocean Survey 20/20 (OS 20/20). Scales of variability in benthic communities are not consistent between regions of the New Zealand EEZ, however. Thus, baseline information specific to each region of interest for monitoring would need to be developed. We suggest that Chatham Rise is the best candidate region for initial development of a benthic monitoring programme, because of its wealth of benthic data and importance to fisheries. Given recent international assessments and the data and sampling methods available in New Zealand at present, there are three main attributes of benthic habitats and communities that we consider to be both important and practical to monitor: biodiversity; ecosystem function, and biological traits, with the concept of ecological integrity providing an overarching framework. Deciding which attributes to concentrate on in practice will be less important than ensuring that the survey itself is designed with a statistically robust structure such that the data collected enable any or all of them to be derived.

Benthic organisms are either sessile or have limited mobility in their adult phase and thus differ fundamentally from demersal and pelagic fish, which are mobile over large areas. Because of the likelihood of high sample variance that would result from sampling the large-scale strata of conventional fisheries stratified-random surveys, such a survey design would be less appropriate for monitoring change in benthic communities than would a ‘repeated measures’ or ‘sentinel sites’ design in which sites are initially assigned randomly within selected strata but then revisited at each subsequent time-point. Sampling strata in a repeated measures survey design could be defined on the basis of: gradients of fishing disturbance; known distributions of benthic communities or habitats, and the locations of existing or planned benthic protection areas. Detection of temporal trends related to fisheries disturbance is complicated by the lack of control areas in which there is no anthropogenic disturbance. A wider goal in this context would be to integrate a benthic monitoring programme with the gathering of data from existing and future protected areas and, ideally, to establish no-fishing control areas within the current fisheries footprint.

Methods used for benthic monitoring should be non- or minimally- destructive, otherwise the monitoring becomes part of the main factor for which effects are being investigated. Based on analyses of OS 20/20 data, the most cost-effective and relevant benthic faunal component to monitor for detecting ecological effects at the scale of deep-sea fisheries is likely to be mega-epifauna. These two considerations suggest photographic transects as the single most appropriate method for monitoring deep-water benthic communities. At present, this demands use of a research vessel for

camera deployment. Attaching benthic sampling gear to trawl gear used in existing fishery surveys or commercial trawling during their core operations would not be acceptable because of the potential effects on catchability. Thus, existing surveys would have to be extended to allow additional dedicated benthic sampling. Data from gear types that can readily be attached to fisheries trawls to sample the benthos at present suggest that they cannot be relied upon to provide representative samples of the benthic community.

The cost of a benthic monitoring programme would be dependent on high-level decisions associated with its location, purpose, extent, and the magnitude of change to be detected. A programme that would have the statistical power to detect trends, even for a relatively small part of the EEZ, would require a substantial commitment of funds over many years. In principle, it is better to start a well-designed programme on a smaller scale and maintain it consistently over the long term than to run a more ambitious programme that, because of its high costs, might not be funded beyond the initial years.

1 INTRODUCTION

The deep seafloor (deeper than 200 m) of the New Zealand Exclusive Economic Zone (EEZ) covers an area of 3.86 million km^2 (more than 93% of the EEZ), and encompasses a diverse range of topographic and physico-chemical habitats and associated benthic communities (Thompson 1991, Gordon et al. 2010). The composition and structure of deep-water communities can change over time as a result of natural forces and anthropogenic activities. Natural drivers of change in deep-water benthic communities include variation in temperature, oxygen, salinity, pressure, light, and current flow (Thistle 2003). Bottom trawling is considered to be one of the most significant current threats to the integrity of deep-water benthic habitats and communities, worldwide, as well as in the New Zealand region (Jones 1992, Thrush & Dayton 2002, Glover & Smith 2003, Ramirez-Llodra et al. 2011).

In response to threats from fishing, and obligations under the international Convention on Biological Diversity, the New Zealand Ministry for Primary Industries (MPI, previously Ministry of Fisheries) has legislation including the Fisheries Act 1994 and strategies (e.g., Ministry for the Environment 2000, Ministry of Fisheries 2005) in place to "… avoid or minimise adverse effects on biological diversity" and "to manage effects from the impact of deep-water fishing activity on the benthic habitat". In order to assess the effectiveness of any measures implemented to manage the effects of fishing, and to adapt management strategies, it is necessary to monitor temporal trends in the ecological status of benthic habitats and communities.

Typically, monitoring for the effects of fishing is undertaken via examination of the status of fish stocks and bycatch species (mainly fish but also benthic and bentho-pelagic invertebrates) in comparison to levels of fishing and potential environmental drivers (Hanchet et al. 2008b). Such monitoring programmes can be cost-effective, because data can be collected by the fishery or related stock assessment research, and in some instances can successfully detect the impact of a fishery on benthic communities (Thrush et al. 1998, Cryer et al. 2002) and declining trends in the populations of some fish and invertebrate species (Clark et al. 2000, Anderson & Clark 2003). However, these programmes are not particularly robust for assessing change in habitat.

In order to detect trends in benthic habitats and communities potentially resulting from fishing activity and other disturbances, some countries have implemented additional monitoring programmes. Typically, these rely upon recovering data on benthic community composition from a set of sampling stations using small beam trawls, grabs or corers (Rees et al. 1999, Ellis et al. 2000, Blake et al. 2009). With the advent of cost-effective imaging technologies, cameras – either attached to trawls or towed independently – are also being used to collect biological and habitat information in monitoring surveys (Dyer et al. 1982, Cranmer et al. 1984, Rumohr 1995, Solan et al. 2003).

To detect change in benthic habitat and communities reliably, it is necessary first to design a robust monitoring programme (Anderson 2002). It is important that such a programme should evaluate, via a preliminary "desktop analysis", the type and distribution of both biological and environmental data (including ecological or environmental indicators) that would be required to monitor change and to reveal the likely cause(s) of such change. This initial evaluation should aim to identify, for example, appropriate habitat and community components to measure (e.g. biogenic habitat structures, epifauna, infauna), the required sampling strategy (e.g. targeted, random or fixed grid survey design) and sample density (in space and time) that would allow for a statistically robust analysis of change (Alden et al. 1997, Van der Meer 1997, Armonies 2000, Rogers et al. 2008). It is also important to incorporate specific or summarized knowledge of the spatio-temporal patterns of natural and anthropogenic drivers, including fishing pressure, into the monitoring programme design (Stelzenmueller et al. 2008).

An ideal monitoring programme is likely to be prohibitively expensive, and so sometimes monitoring is conducted by obtaining additional data from surveys planned for other purposes. However, such adapted (or "tagged on") surveys run the risk of acquiring data that will not allow for the statistically robust detection of change, as well as presenting a risk to the long-term continuity of data collection if they are not principal objectives of the main survey. Thus, in order to ensure effective monitoring of the ecological status of benthic habitats and communities it is necessary to determine the feasibility and utility of adapting planned surveys to deliver appropriate and reliable time-series data. In this study, we examine how such a benthic monitoring programme appropriate for the New Zealand region might be designed, what resources would be required, and whether or not there are opportunities for gathering useful data by augmenting existing fisheries research surveys.

Overall objective: To design a programme to monitor trends in deep-water benthic communities.

Specific objectives:

1. To design and provide indicative costs for a programme to monitor trends in deep-water benthic communities.

2. To explore the feasibility of using existing trawl and acoustic surveys to capture data relevant to monitoring trends in deep-water benthic communities.

2 METHODS

2.1 Designing a monitoring programme

Five main tasks were defined under this specific objective: (1) examining the data on benthic habitats and fauna currently available for the New Zealand EEZ, (2) assessing what these data can tell us about characteristic spatial scales of variability in benthic faunal communities; (3) determining which attributes of benthic habitats and communities to monitor; (4) examining survey design options for a programme to monitor trends in benthic habitats and communities, and (5) exploring the potential costs of such a monitoring programme.

2.1.1 Description of data currently available on benthic habitats and communities in the New Zealand EEZ

We investigated the availability of deep-water benthic invertebrate data for a range of faunal size classes (meiofauna, macrofauna, megafauna) and living positions (epifauna, infauna, and demersal) and collected by different sampling methods (trawls, sleds, grabs, corers, cameras). Data sources included:

- commercial fishing returns
- information from the Scientific Observer programme
- fishery and biodiversity research surveys
- other research work undertaken by New Zealand Institute of Water and Atmospheric Research (NIWA – previously the New Zealand Oceanographic Institute, NZOI), through the Foundation for Research Science and Technology (FRST), the Ministry of Business, Innovation and Employment (MBIE), and Ocean Survey 20/20 (OS 20/20) programmes.

These data were accessed through databases maintained by MPI and NIWA. Effort was concentrated on research data from the *Specify*, *Bio-ds*, and *trawl* databases because these sources are the most reliable in terms of completeness of catch and accuracy of species identifications. Specialist scientists at NIWA provided information about other datasets and databases relevant to deep-water benthic habitats, including layers collated for development of environmental classifications, and consistent metadata summaries were created for all datasets.

2.1.2 Assessing scales of variability

Available data on benthic community composition from OS 20/20 surveys of Chatham Rise, Challenger Plateau (Bowden 2011, Floerl et al. 2012), and the Northland shelf (http://www.os2020.org.nz/) were used to generate the attributes and metrics identified in Section 2.1.3 as being useful for a monitoring programme. Data were from towed video transects and epibenthic sled samples, with each transect or tow being treated as a point sample, as described by Bowden (2011). For each dataset, the variability (standard deviation) of these metrics in relation to the spatial separation between sampling sites was calculated and plotted as line graphs. In combination with earlier analyses (Bowden & Hewitt 2012), these graphs enabled estimation of characteristic spatial scales of community change.

2.1.3 Determining which attributes of the benthic habitat and communities to monitor

Many ecological and environmental attributes have been identified to characterise the state of natural systems. These may include information on the occurrence, abundance, or size, of particular ‘key’ or readily observed taxa (including habitat-forming species), or the species composition of particular communities. However, to be useful for monitoring purposes they need to be readily and reliably measured. Useful attributes also have to be suitable for detecting change as a result of both natural variability and anthropogenic activities, particularly fishing in the present context. For this project, a suite of appropriate attributes and indices (“response variables”) was developed and agreed by consultation through meetings with MPI science and fisheries management staff, and scientists with particular expertise in the use of such indices (within NIWA and overseas institutions). A number of other MPI projects have involved identification of community metrics and attributes suitable for use as ecological/environmental indicators, and the findings of these projects were considered in the selection process here (e.g. ENV2006-04 *Ecosystem Indicators for New Zealand Fisheries*, ZBD2007-01A *Chatham-Challenger Post-Voyage Analyses*, and DEE2010-05 *Development of a suite of ecosystem and environmental indicators for deep-water fisheries*).

2.1.4 Survey design options for a monitoring programme

Gear type

Deciding on the type of gear to be used for a benthic monitoring programme is a key decision, which is closely related to selection of the most appropriate benthic community attributes. Not only must the methods used yield appropriate spatial and taxonomic resolution for the faunal components of interest but, because of the need for robust comparisons between time points that is inherent in any monitoring study, they must also be reliably consistent in these attributes over months, years, and potentially decades. In this section, we assess, briefly, the pros and cons of the gear options available. Under Specific Objective 2, below, we use data from published studies and from a recent survey in New Zealand (R/V *Tangaroa*, voyage TAN1208) during which multiple gear types were used at the same sites, to compare how effectively each gear samples benthic habitats and fauna.

Identifying appropriate schemes for survey stratification

Appropriate survey stratification demands some prior knowledge of the nature, extent, and variability of benthic habitats across the survey area, and is particularly important for any long-term monitoring programme in which the aim is to detect community change over time. This requirement for prior knowledge suggests that any area selected for such monitoring should have credible baseline data on benthic habitats. For deep-water regions within the New Zealand EEZ, Chatham Rise is the only area for which such information exists: it has been extensively studied for fisheries and benthic research over many years, was the subject of the first OS 20/20 benthic project, and is the most biologically productive deep-sea region in New Zealand waters. There are several potential and existing schemes that might be used for spatial stratification of a benthic survey programme on Chatham Rise. The most relevant of these in the present context are: (1) the Benthic Optimised Marine Environment Classification (BOMEC, Leathwick et al. 2012), (2) the Biotic Habitats generated from OS 20/20 sample data (Hewitt et al. 2011a, Floerl et al. 2012), and (3) the Chatham Rise hoki and middle-depths research trawl strata (O'Driscoll et al. 2011). We consider the relative merits of each of these stratification schemes in terms of their relevance to benthic habitats and communities, the scope of data used in their formulation, and their practicality for use as the basis of allocating survey sites.

Determining the number and type of samples

Using the original Chatham Rise OS 20/20 data and the Biotic Habitats developed by Floerl et al (2012) as strata, Bowden & Hewitt (2012) developed a power analysis (Cohen 1988, Fairweather 1991) to indicate the number of samples per stratum that would be needed to detect differences between the Biotic Habitats across a number of levels of contrast ('effect size') between habitats. We apply the same procedure here, using the same biological data from the OS 20/20 surveys but using the alternative stratification scheme selected from the possibilities assessed above as being most appropriate for a monitoring programme.

2.1.5 Costing an optimal monitoring programme

The indicative cost of implementing the optimal benthic monitoring programme for Chatham Rise developed through the stages above was estimated based on resources (vessel, gear, personnel) and time used for comparable benthic sampling programmes over recent years, with a breakdown of costs into sample collection, sample processing, sample curation and archiving, data analysis, and reporting. Costs are presented in terms of resources required (vessel-days, person-hours) rather than dollar values, which will certainly vary considerably through the course of a monitoring programme as a consequence of changes in, for instance, the economic environment and technological developments.

2.2 Exploring the feasibility of using existing surveys to monitor benthic habitats

Three main tasks were defined under this specific objective: (1) determining the location, extent, timing and frequency of planned fisheries research trawl and acoustic surveys in the New Zealand EEZ; (2) determining the feasibility of a "tagged-on" benthic monitoring programme, and (3) evaluating the potential cost of a "tagged on" monitoring programme.

2.2.1 Determining the location, extent, timing and frequency of planned trawl and acoustic surveys

We identified proposed and planned surveys within the New Zealand EEZ, and also considered how the planned coverage of these surveys might tie in with areas, such as Chatham Rise, where there are existing benthic data that might serve as reference data or extend the time-series backwards (e.g. OS 20/20 surveys, Fisheries research trawl surveys, Seamounts Programme).

2.2.2 Determining the feasibility of a "tagged on" monitoring programme

The extent to which it would be possible to incorporate aspects of the design for an optimal monitoring programme developed in Specific Objective 1 into planned trawl and acoustic surveys was evaluated in terms of both the survey design and the types of gear deployed. A major part of the second aspect of this evaluation involved comparing the catch composition and catch rates of different types of benthic sampling gear that might typically be used in a "tagged on" survey, using data from a research voyage (RV *Tangaroa* voyage TAN1208) during June 2012 carried out under an MPI-funded project DEE2011-05; "*Further development of acoustic methodologies and assessment of New Zealand's deep-water fisheries and habitats*".

Data from TAN1208

Benthic invertebrate data were collected by five methods:

- rat-catcher trawl
- net bags attached beneath the ground-rope of the rat-catcher trawl
- beam trawl
- video camera mounted on the headline of the rat-catcher trawl
- still image camera mounted on the headline of the rat-catcher trawl

The survey methods and gear used are detailed in O'Driscoll (2012) and outlined briefly here. Samples were collected in four depth strata: 300–400 m (S); 500–600 m (M); 700–800 m (D), and 950–1050 m (X). In each depth stratum, the beam trawl and the rat-catcher trawl (with video and still cameras attached) were each towed at three sites, resulting in a total of 4 strata × 3 sites = 12 tows for each of the gear types (Table 1).

The rat-catcher trawl had upper and lower wings, with a headline height of about 3.3 m, 150 mm mesh in the wings, 40 mm mesh cod end, and low (200 mm bobbins) ground-gear. A net bag about 5 m wide was fitted, covering the centre portion of the rat-catcher ground-gear (about 10% of the full width of the net). The rat-catcher trawl with net bag was towed for about 1.5 nautical miles at 3 knots. The beam trawl had a 3 m mouth width and a 10 mm mesh cod end. Each beam trawl was towed for about 0.3 nautical miles at 1.0–1.5 knots. The rat-catcher trawl and net bag catches were processed separately and were treated as separate gear types in analyses.

Table 1: Benthic sampling stations from RV *Tangaroa* voyage TAN1208 from which data were used to compare catch characteristics of a beam trawl (BT), a 'rat-catcher' trawl (RT), and a ground rope net bag (NB) attached to the rat-catcher trawl.

Stratum	Area	Station	Latitude S	Longitude E	Depth (m)	Gear	Distance (nautical miles)
D	1	35	42.547	179.5771	758	BT	0.3
D	2	34	42.521	179.4472	789	BT	0.41
D	3	33	42.518	179.3458	765	BT	0.34
D	1	32	42.545	179.5716	759	RT, NB	1.49
D	2	49	42.522	179.4438	772	RT, NB	1.42
D	3	53	42.515	179.3344	772	RT, NB	1.5
M	1	36	43.121	179.5047	518	BT	0.38
M	2	37	43.092	179.4199	524	BT	0.29
M	3	38	43.099	179.2809	522	BT	0.31
M	1	50	43.117	179.5153	518	RT, NB	1.52
M	2	42	43.089	179.3972	516	RT, NB	1.5
M	3	51	43.096	179.2629	527	RT, NB	1.5
S	1	67	43.487	179.1774	344	BT	0.26
S	2	70	43.452	179.0462	397	BT	0.29
S	3	69	43.359	179.1771	404	BT	0.28
S	1	66	43.488	179.1859	338	RT, NB	1.45
S	2	71	43.452	179.0467	396	RT, NB	1.49
S	3	64	43.351	179.1942	400	RT, NB	1.71
X	1	57	42.487	179.5901	975	BT	0.34
X	2	56	42.483	179.5328	956	BT	0.32
X	3	58	42.486	179.4959	1005	BT	0.37
X	1	55	42.486	179.5919	989	RT, NB	1.53
X	2	63	42.483	179.528	957	RT, NB	1.42
X	3	59	42.486	179.5003	1010	RT, NB	1.49

Invertebrate taxa were identified to the finest taxonomic level possible and all specimens of each taxon were weighed separately to the nearest 0.1 kg (wet weight). Identification and weighing was conducted at sea. Pelagic taxa (squid, salps, and jellyfish) were not included in analyses. An unusually large biomass of the swimming sea cucumber *Enypniastes eximia* was obtained at station 59 (stratum X, rat-catcher trawl), which was two orders of magnitude greater than the total invertebrate biomass at all other stations. Because this taxon was absent at all but one other station (station 49, 0.1kg), it was omitted from analyses of community structure.

To obtain more detailed information on the identity of the fauna present at the sampling sites for the present project, benthic invertebrate specimens from all beam trawl stations (456 specimen lots) and most of the rat-catcher trawl stations (30 lots) were identified to finer taxonomic level (i.e., species, genus, or family) by specialist taxonomists and parataxonomists.

Video and still imagery were recorded by a centrally-mounted, downward-looking camera system attached to the headline of the rat-catcher trawl, viewing the central portion of the ground-rope. The video camera was a GoPro Hero 3 and the stills camera was a Nikon E5000, each in a custom pressure housing. One station in each of the four depth strata was selected for intensive video and stills analysis, the station with the best overall image quality being selected. Image quality was assessed on a scale of 1–4, with only scores of 1 and 2 being used in analyses:

1- Images good - Seabed and benthos in focus, well-lit and of good resolution, can observe all benthic fauna larger than about 5 cm.
2- Images ok - Seabed and benthos roughly in focus, marginally lit and of average resolution, can observe most benthic fauna larger than about 5 cm.
3- Images poor - Seabed and benthos roughly in focus, poorly lit and of poor resolution, can observe few large benthos.
4- Images unusable - Seabed and benthos very poorly lit and of poor resolution, cannot observe benthos.

Analysis of the video footage was conducted using Ocean Floor Observation Protocol (OFOP, www.ofop-by-sams.eu), the software that NIWA generally uses for image analysis of benthic video transects (Bowden & Jones In press). All identifiable benthic organisms observed over the full length of each video transect, and in all useable still images, were counted.

Data analyses

For the comparison between rat-catcher trawl, beam trawl, and net bag, analyses of total biomass, taxon richness, and community structure were conducted using the PERMANOVA routine in the multivariate software package PRIMER v6 (Clarke & Gorley 2006). PERMANOVA is a semi-parametric, permutation-based routine for analysis of variance based on any similarity measure (e.g., Bray-Curtis). Analyses were run using a repeated measure design to take into account the lack of independence between tows within each area, and between the rat-catcher trawl and net bag, which originated from the same tows (Quinn & Keough 2002). The following factors were used: Gear (fixed, three levels: Beam trawl, Rat-catcher trawl, and Net bag), Stratum (fixed, four levels: S, M, D, and X) and Area (random, nested within Stratum but not Gear). Pairwise comparisons were conducted using Monte-Carlo sampling due to the low number of replicates (Anderson et al. 2008b). P-values for individual predictor variables were obtained using 999 permutations. Because PERMANOVA can be sensitive to differences in multivariate dispersion among groups, the PERMDISP routine in PRIMER was used to test for homogeneity of dispersion when significant factor effects were found (Anderson et al. 2008b).

Analyses of multivariate community structure were based on similarity matrices using Bray-Curtis similarity (Clarke et al. 2006) of log-transformed biomass (wet weight) data standardised for tow distance. The log transformation was used to decrease the influence of abundant taxa on community patterns (Clarke & Warwick 2001), and standardisation was used to account for the difference in tow length between beam (about 0.3 nautical mile) and rat-catcher trawl (about 1.5 nautical miles). The RELATE routine was used to test for correlations between similarity matrices derived from data from each of the gear types. The Similarity Percentages routine (SIMPER) was used to identify the taxa contributing to dissimilarities between gear types. Analyses of univariate measures (i.e., biomass and taxon richness) were based on similarity matrices built using Euclidean distance of untransformed data. Data from video and still imagery were compared simply by listing the taxa identified in each, and graphically comparing numbers and proportions of individuals recorded by each method.

2.2.3 Costing a “tagged on” monitoring programme

The indicative cost of implementing a “tagged on” monitoring programme was estimated based on how a given benthic sampling approach could be implemented alongside the existing objectives of other surveys, or from ships of opportunity. Estimated costs were assessed for four components: sample collection; sample and data archiving; data analysis, and report writing and presentation.

3 RESULTS AND DISCUSSION

3.1 Designing a monitoring programme

3.1.1 Description of data currently available on benthic habitats and communities in the New Zealand EEZ

A summary of the principal datasets and databases that hold information about deep-water benthic fauna distributions within the New Zealand EEZ is given in Appendix 1 (Table A 1). These records represent data in four broad categories: datasets compiled for use in the Benthic Optimised Marine Environment Classification (BOMEC); data from OS 20/20 surveys; benthic invertebrate biodiversity databases, and fisheries databases. Analysis for some of these records has yet to be completed, whereas others represent groomed data that have already been used in community or habitat analyses. In addition to these faunal data, several physical oceanographic parameters of direct relevance to benthos distributions have been developed into data layers with consistent coverage across the entire EEZ. Data on seabed sediment characteristics have also been collected by geological surveys over many decades. Because benthic distributions are strongly associated with substratum type, these data have the potential for use in stratifying benthic surveys. However, until work currently in progress to collate and standardise data from disparate sources (initiated at NIWA under project CO1X0702 – *Consequences of Earth-Ocean Change* – and continuing under project CO1X1229 – *Vulnerable Marine Ecosystems* – in collaboration with the University of Colorado) is complete, these data remain of limited practical use for guiding benthic sampling effort across large spatial scales.

While there has been widespread benthic sampling across the New Zealand EEZ over many decades, it is only with the instigation of the research trawl surveys in the early 1990s (e.g. O'Driscoll et al. 2011) and the OS 20/20 programme in 2006 (e.g. Bowden 2011) that standardised methods and reporting have been established for capturing benthic data over large spatial scales. The OS 20/20 surveys (Chatham Rise - TAN0705, TAN1306; Challenger Plateau - TAN0707; the Northland shelf and Bay of Islands - TAN0906, KAH0905, and the Ross Sea - TAN0802), in particular, have generated detailed, spatially extensive data on benthic diversity and distributions using a standard range of sampling methods including cameras, epibenthic trawls and sledges, and sediment corers. Data from these voyages thus provide a potentially useful basis against which to evaluate future change; the use of OS 20/20 data is discussed in subsequent sections here (Sections 3.1.2, 3.1.4). For specific deep-sea habitats, including seamounts, canyons, methane seeps, and hydrothermal vents, sampling conducted under the FRST/MBIE-funded Seamounts and Vulnerable Deep-Sea Communities programmes has generated detailed and consistent data on benthic distributions within the New Zealand EEZ. Because these data are spatially restricted, however, they do not provide a suitable baseline for benthic monitoring over broader scales.

Fisheries bycatch records, particularly those from research trawl survey programmes, are a potential source of consistent time-series data on the status of benthic communities. O’Driscoll et al. (2011) reviewed all catch data from a time-series of 19 hoki and middle-depth research trawl surveys on Chatham Rise from 1992 to 2010 and noted that, of 558 biological groups recorded in the surveys, “A large number (were) invertebrates with very low catch weights or frequency of occurrence”. Although high proportions of ‘rare’ taxa are a characteristic of many ecosystems (Gray et al. 2005), this pattern in the research trawl data is likely, in part, to be a consequence of the low catchability of most benthic taxa by demersal fish trawls.

The review of the trawl survey data is useful in the present context, however, because it is the longest consistent time-series in New Zealand deep water fisheries and presents data from a rigorously designed, stratified random survey programme using consistent sampling gear and methods throughout. These data enabled O’Driscoll et al. (2011) to calculate statistically valid estimates of within-stratum biomass variability (as coefficient of variation, CV) for 49 taxa but only nine of these were benthic invertebrate groupings: Anemones; Corals; Crabs; Urchins; Gastropods; Holothuroids; Octopods; Sponges, and Asteroids (Table 2). Although each of these groupings was comprised of at least three and up to sixteen separate taxa, the taxonomic resolution of identifications varied between taxa and between years, and because codes for some taxa changed during the course of the time series, some single taxa are represented twice (e.g. ECN and ECH for 'urchin'). Plots of the numbers of individual taxa recorded in higher taxonomic groupings across all years of the time series illustrate that reported taxon richness of benthic invertebrates in the data-set has increased with time, particularly from the early 2000s when benthic invertebrate guides became available (Figure 1).

In addition to being identified and recorded at sea, benthic invertebrate bycatch specimens caught during the hoki and middle-depth research trawl surveys on Chatham Rise are returned to the NIWA Invertebrate Collection (NIC) and identified by parataxonomists and taxonomists at NIWA and other research institutions. Data resulting from these laboratory identifications are recorded in the *Specify* database of the NIC. Records in *Specify* span 15 of the 19 hoki and middle-depths fisheries surveys, from 1996–2010 (Figure 2), and include 293 named taxa. The total number of individual specimens recorded, however, is only 1676 over the entire period and the number of specimens recorded per annual survey ranges from only 2 (TAN9601) up to 487 (TAN0601).

Table 2: Benthic invertebrate groupings derived from 19 years of Chatham Rise research trawl survey records (1992–2010), showing the number of taxa resolved within each grouping. Note, taxonomic levels of codes vary and some codes have changed over the duration of the time-series, resulting in more than one code for a single taxon (e.g. ECN and ECH for echinoids). Adapted from table 4 in O'Driscoll et al. (2011).

Taxon group code	Taxon group name	Identification codes for taxa included in group
ANT	Anemones	ACS, ANT, HMT
COU	Corals	COU, GDU, SIA, SPN
CRB	Crabs	ATC, CRB, DAP, PAG, PZE, SSC, LMU, NEB, OVM, PHS
ECN	Urchins	ARA, DHO, ECH, ECN, GPA, GRM, PBU, PMU, SPT, TAM
GAS	Gastropods	FMA, GAS
HTH	Holothuroids	BAM, HTH, LAG, PMO, SCC
OCP	Octopods	DWO, EZE, OCP, OCT, OPI
ONG	Sponges	ANZ, GLS, HYA, ONG, SUA
SFI	Asteroids	ASR, CDY, CJA, DMG, GOR, HTR, MSL, OPH, PKN, PT, PRU, PSI, SFI, SMO, SOT, ZOR

Thus, while bycatch data from *Specify* are more taxonomically consistent than those from ship-board records, and therefore might be expected to be more useful for monitoring trends, the data in *Specify* are incomplete (Figure 2). This is partly because not all samples from all years have yet been identified, and partly because not all specimens from a given trawl will have been preserved and returned to shore at the time of capture. It is also likely that there will have been some inconsistency between years in the thoroughness with which benthic invertebrates were recorded and retained at sea. These findings suggest that data derived from the research trawl surveys in the New Zealand EEZ are presently unreliable for monitoring long-term change in benthic invertebrate communities.

Existing information on benthic communities across the New Zealand EEZ have been used to tune algorithms underlying the Benthic-Optimised Marine Environments Classification (BOMEC) (BOMEC, Leathwick et al. 2012). This classification is the latest incarnation of on-going initiatives to build a comprehensive classification of New Zealand's marine environments (Snelder et al. 2007, Leathwick et al. 2012) and will be discussed later during consideration of survey stratification schemes (Section 3.1.4).

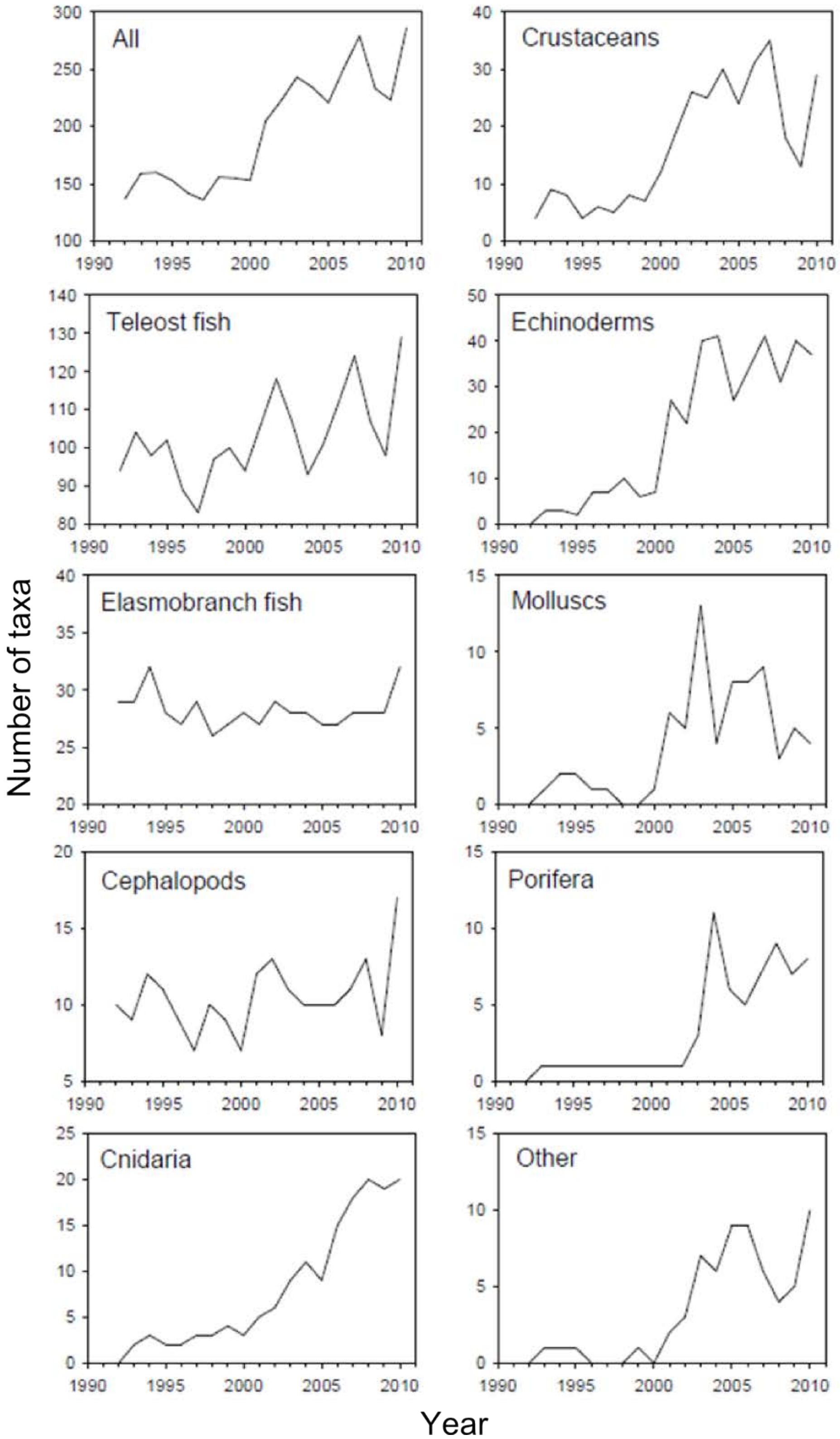

Figure 1: Numbers of individual taxa (species or other) identified in each of ten broad groupings in trawl surveys on Chatham Rise from 1992 to 2010. Standardised identification guides for benthic invertebrate taxa (Crustaceans, Echinoderms, Molluscs, Porifera, Cnidaria) were phased in from 2000. Note: Scale on the y axis differs between plots.

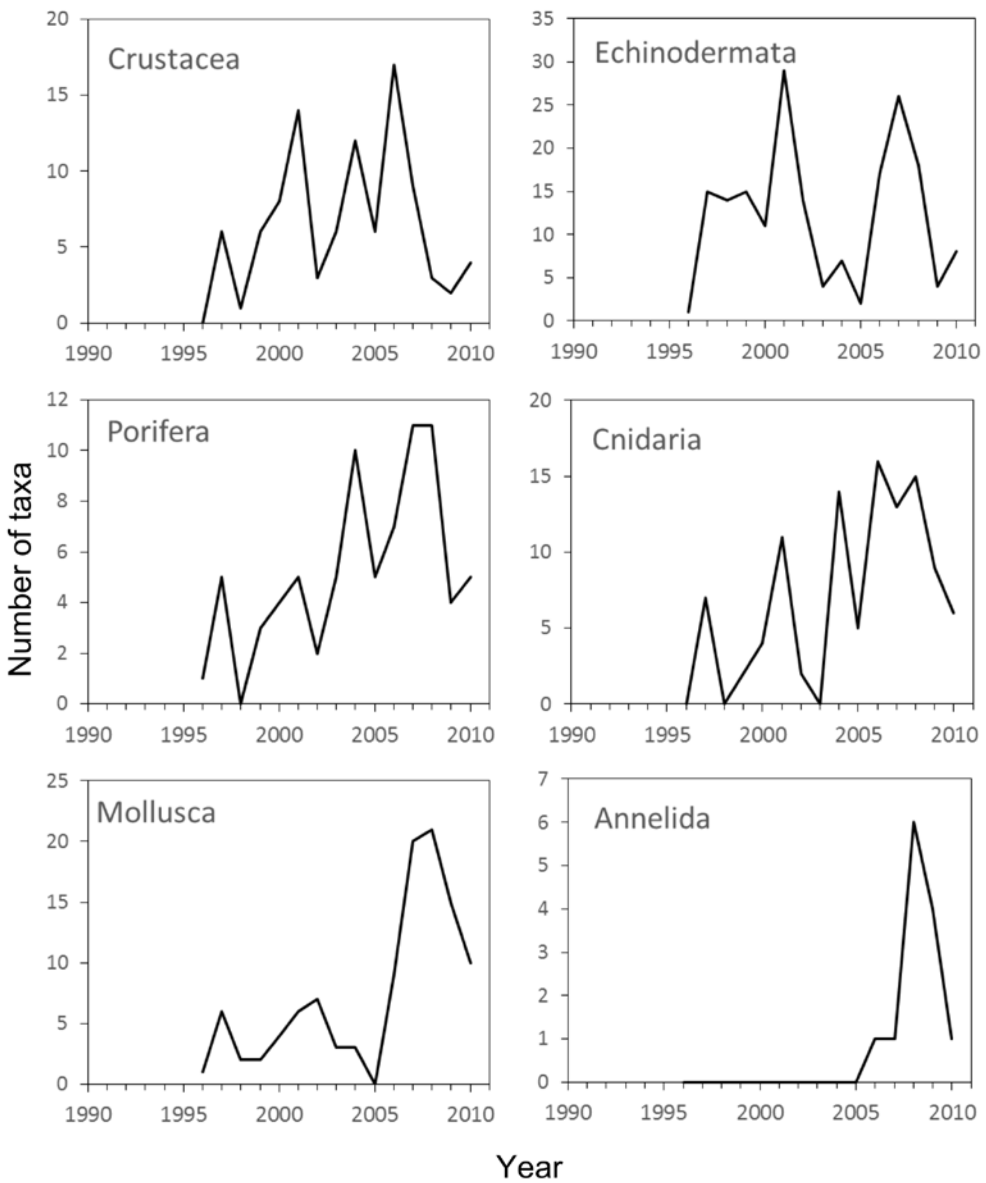

Figure 2: Numbers of individual taxa (species or other) identified in the *Specify* database in each of six broad groupings in trawl surveys on Chatham Rise from 1992 to 2010 (Crustaceans, Echinoderms, Porifera, Cnidaria, Molluscs, and Annelids). The trawl samples are the same as those used for Figure 1 but in *Specify*, only specimens that were retained and preserved at sea are represented. Note: Scale on the y axis differs between plots.

3.1.2 Assessing scales of variability

Plots of the variability (as standard deviation) of a suite of benthic community metrics (see Section 3.1.3 for discussion of metrics) in relation to the spatial distance between samples, based on data from video transects using NIWA's Deep Towed Imaging System (DTIS), showed that there were no consistent general patterns between three representative areas of the EEZ surveyed under OS 20/20: Chatham Rise; Challenger Plateau, and the Northland shelf, including the Bay of Islands (Figure 3, Figure 4, and Figure 5, respectively). For both the Northland shelf and for Chatham Rise, there was least variability at the smallest spatial scale (less than 1 km and less than 5 km, respectively), increasing with increasing spatial separation. For Challenger Plateau, by contrast, variability was more constant across spatial scales.

These results demonstrate that scales of variability in benthic communities are not consistent between regions of the New Zealand EEZ, and thus that an effective benthic monitoring strategy would need to be developed on the basis of detailed baseline information for the particular region in question.

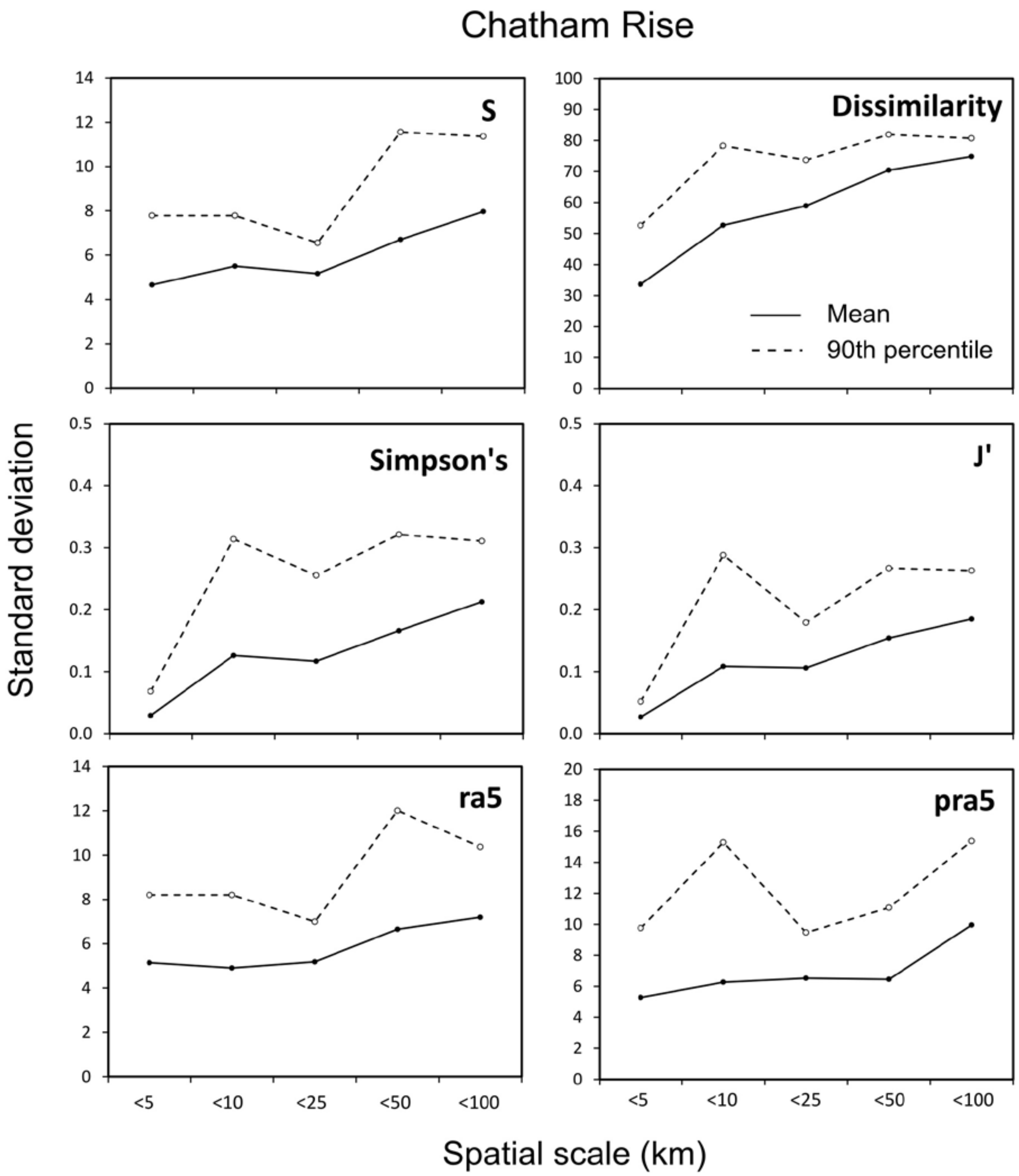

Figure 3: Variability (standard deviation) of benthic community metrics in relation to the spatial distance between samples on Chatham Rise. Based on benthic invertebrate data from video transects using NIWA's DTIS towed camera system during Chatham-Challenger OS 20/20 voyage TAN0705. Metrics: number of taxa (S); Bray-Curtis dissimilarity (Dissimilarity); Simpson's diversity (Simpson's); Pielou's evenness (J'); number of taxa rare in abundance (ra5); proportion of taxa rare in abundance (pra5).

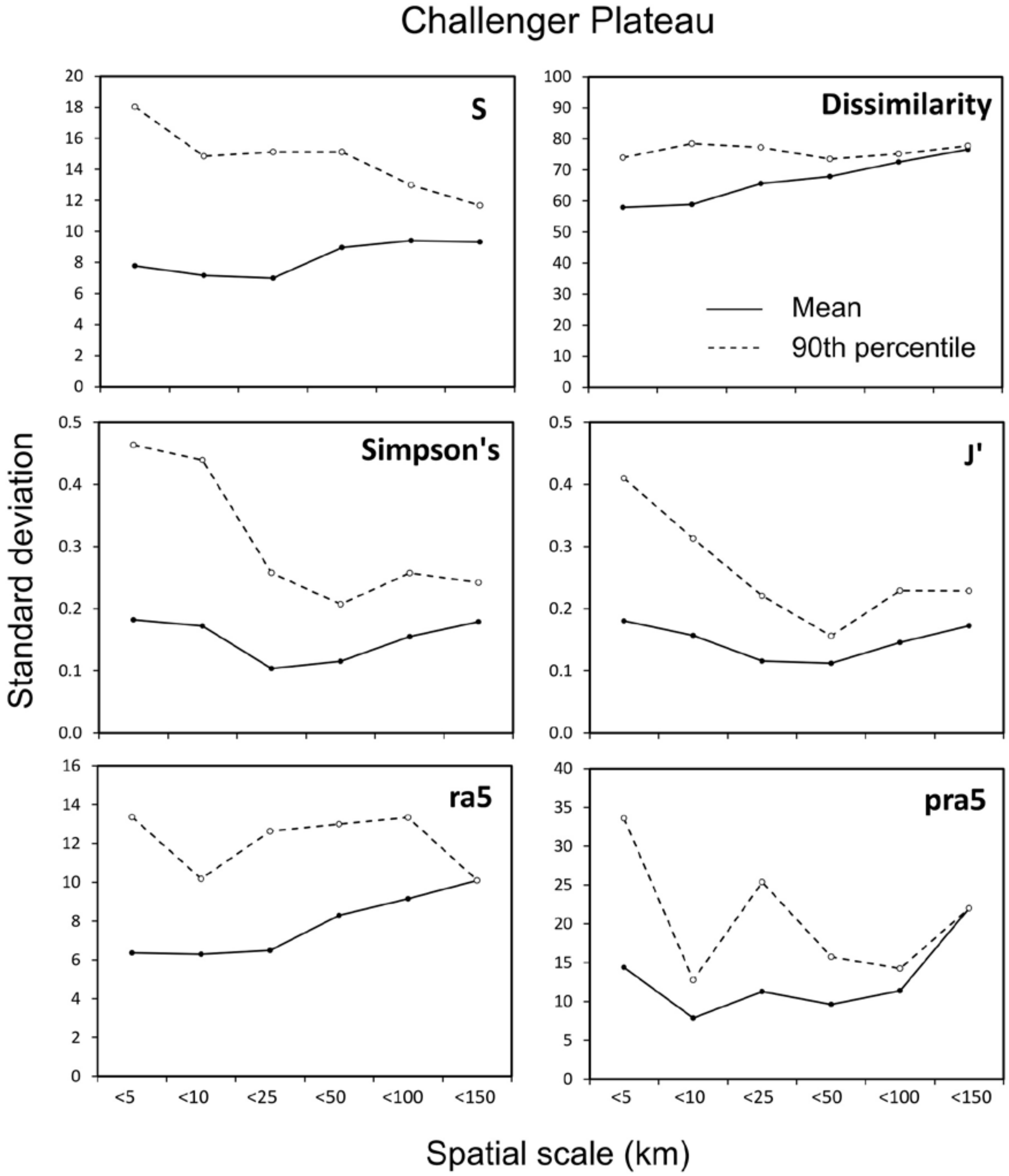

Figure 4: Variability (standard deviation) of benthic community metrics in relation to the spatial distance between samples on Challenger Plateau. Based on benthic invertebrate data from video transects using NIWA's DTIS towed camera system during Chatham-Challenger OS 20/20 voyage TAN0705. Details as for previous figure.

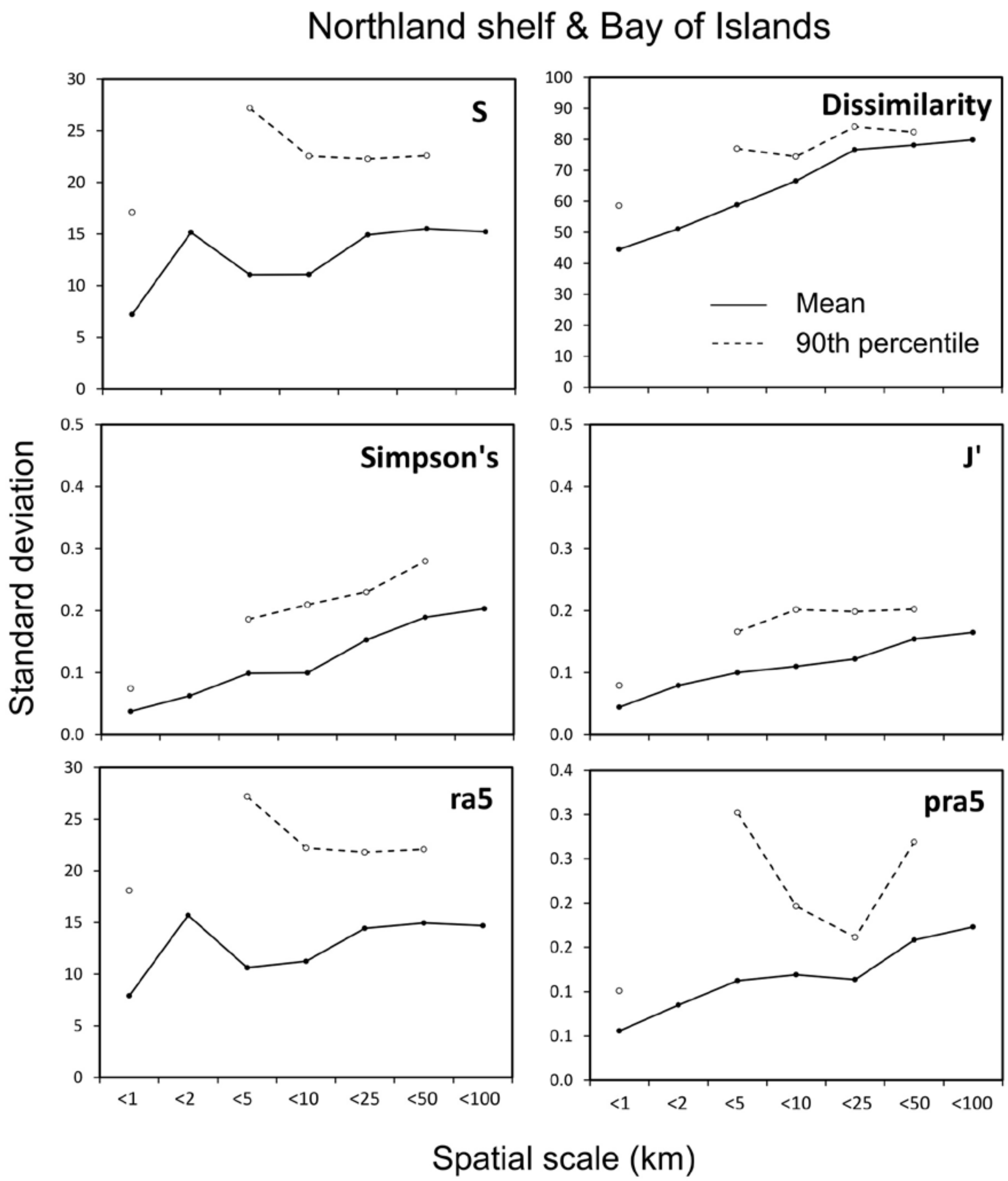

Figure 5: Variability (standard deviation) of benthic community metrics in relation to the spatial distance between samples from the Northland shelf, including the Bay of Islands. Based on benthic invertebrate data from video transects using NIWA's DTIS towed camera system during Bay of Islands OS 20/20 voyages TAN0906 and KAH0907. Details as for previous figure.

3.1.3 Determining which attributes of the benthic habitat and communities to monitor

The recently developed European Marine Strategy Framework Directive (MSFD, http://ec.europa.eu/environment/water/marine/ges.htm) uses as an overarching principle the concept of "good environmental status". In summary, this covers maintenance of ecologically diverse, clean, healthy and productive systems, and sustainable use that preserves the structure, functions, processes, and resilience of the systems. Eleven descriptors were defined under Article 9 (3) of the MSFD:

- Descriptor 1: Biological diversity.
- Descriptor 2: Non-indigenous species.
- Descriptor 3: Population of commercial fish / shell fish.
- Descriptor 4: Elements of marine food webs.
- Descriptor 5: Eutrophication.
- Descriptor 6: Sea floor integrity.
- Descriptor 7: Alteration of hydrographical conditions.
- Descriptor 8: Contaminants.
- Descriptor 9: Contaminants in fish and seafood for human consumption.
- Descriptor 10: Marine litter.
- Descriptor 11: Introduction of energy, including underwater noise.

An international group of experts considered the MSFD with regard to monitoring seafloor habitats and concluded that there are eight categories of attributes that should be monitored (Rice et al. 2012):

- Substratum
- Bioengineers
- Oxygen concentration
- Contaminants and hazardous substances
- Species composition
- Size distribution
- Trophodynamics
- Energy flow and life history traits.

A plethora of metrics and indices can be generated from such attributes for monitoring the state of the environment and its responses to anthropogenic activities. For coastal marine indices alone, Martinez-Crego et al. (2010) investigated 90 published biotic indices, the majority of which are also applicable to deep-sea environments. Diaz et al. (2004) evaluated 64 separate indices of seafloor habitat quality and concluded that the number of indices reflected a lack of consensus among environmental managers and scientists as to which specific metrics are most useful. There is greater consensus, however, on the general properties required for such metrics to be useful as effective ecological indicators. They must:

- be easily and cost-effectively assessed with accepted (standard) methods;
- be used nationally and internationally;
- be sensitive to stressors on the system in a predictable fashion, and preferably sensitive to change in advance of the whole ecosystem so that management intervention can be effective;
- have wide geographic coverage and be able to be utilised across key environmental gradients;
- have low, or predictable, natural temporal variability;
- relate to management goals;
- be scientifically defensible.

Evaluations across indices suggest that most fail as indicators of ecological status because of narrow applicability and their reliance on the definition of reference conditions (Diaz et al. 2004, Borja et al. 2009, Martinez-Crego et al. 2010). To some extent, this problem can be overcome by combining different but complementary characteristics of individual indices, and by integrating indices across ecosystem elements (Borja et al. 2009).

It is important to identify ecosystem components and pressures of greatest importance for a particular area (Rice et al. 2012). In the context of the present project, this requires a focus on indicators related to fishing. Recently, a number of indicators have been developed to assess the impact of fishing. Many focus on traditional fisheries management, others relate to data that can be routinely collected by fishers (Tuck et al. 2009), or that relate to ecosystem function or seafloor habitat integrity. In particular, the application of biological traits analysis has both facilitated comparisons across communities (Hewitt et al. 2008) and made important links between changes in community structure and function (Bremner et al. 2006b, Bremner et al. 2006a). For example, filter-feeding, attached epifaunal organisms and large organisms, in general, tend to show negative correlations with trawling intensity, whereas small infauna and scavengers tend to become more abundant (Tillin et al. 2006). Over the long term, a common pattern emerges of the loss of epifauna and large and long-lived organisms such as burrowing urchins, large bivalves, sea pens, and reef-building sabellid polychaetes (Robinson & Frid 2008). De Juan et al. (2007, 2009) developed a multivariate approach to assessing trends in benthic communities based on the abundance and density of large epifaunal organisms. This approach utilised a combination of biological traits that relate to size, age, rarity and vulnerability to trawling disturbance, with trials indicating that the approach was effective in both sandy and muddy sediments. The biological traits approach was also utilised in MPI project ZBD200701 Objective 10, to identify habitats that may be vulnerable to the disturbance created by trawling on the Chatham Rise and Challenger Plateau (Hewitt et al. 2011a).

Modelling approaches have also been advocated as a way of developing indicators and interpreting their broader significance. One approach is to develop indicators based on data already being gathered for traditional fisheries management. However, this generally results in a minimal suite of indicators (Link et al. 2002). Link et al. (2002) emphasised the need for a diverse array of indicators to characterise ecosystem status and called for the development of mechanistic or analytical models of key ecosystem processes.

In New Zealand, the Department of Conservation is exploring monitoring within the framework of "ecological integrity" for within and outside Marine Reserves. Ecological integrity (variously described as ecosystem or biological integrity) is a holistic term that is based on the necessity to safeguard the self-organising capacity of ecosystems (Burkhard et al. 2011). Lee et al. (2005) defined biological integrity as: "The capacity to support and maintain a balanced, integrated, adaptive biological system having the full range of elements (genes, species, and assemblages) and processes (mutation, demography, biotic interactions, nutrient and energy dynamics, and metapopulation processes) expected in the natural habitat of a region (Karr & Thomas 1996)." Thrush et al (2011) have operationalised this concept for marine coastal areas in a recent report, which DOC is presently testing in monitoring in and around a number of marine reserves (Hewitt et al. 2014). Specifically this report deals with indices that represent processes, such as connectivity, through landscape-scale information on patch structure or the degree of bare space (Garrabou et al. 1998, Bartel 2000, Bostrom et al. 2006) and resilience, through species richness and the degree of variability in community composition (beta-diversity) (Hewitt et al. 2005, Thrush et al. 2006, Thrush et al. 2008, Thrush et al. 2010).

Here we assess the potential of using indicators based on four high-level characteristics of benthic habitats and communities that are commonly referred to in ecological literature. Three of these are direct measureable attributes, albeit with considerable overlap between them, while the fourth is the overarching concept of 'ecosystem integrity':

- Biodiversity
- Biological traits
- Ecosystem function
- Ecological integrity

Assessment of each of these attributes is likely to be both practical and important to monitor given the available data, sampling methods, and seabed resource use in New Zealand.

Biodiversity

Biodiversity has over 42 potential measures (Magurran 2004) and can be defined at many spatial scales in three forms: point (alpha) diversity; overall (gamma) diversity, and the difference in assemblage composition from one sample point to another or between point and overall diversity (beta diversity). Most indices for alpha and beta diversity fall into four main categories:

1. Richness. Based on the number of taxa observed or predicted.
2. Evenness and Dominance. Based on the relative abundances of the taxa present (ranging from 0, where each taxon constitutes a different proportion of the sample, to 1 where all taxa are present in equal proportions) or the proportion of abundance accounted for by the most numerically-dominant taxa.
3. Combined indices. Incorporating elements of both the number of taxa present and how abundance is distributed amongst them.
4. Rarity. Rarity indices are either an estimate of the number of taxa that are low in abundance or the number of taxa that occur infrequently.

Hewitt et al. (2011b) analysed benthic invertebrate community data from the OS 20/20 surveys of Chatham Rise and Challenger Plateau and investigated fourteen indices associated with diversity. They recommended a sub-set of six of these as being potentially useful for assessing variations in benthic communities:

1. Number of taxa (species richness, S)
2. Pielou's evenness (J')
3. Simpson's diversity (λ).
4 Number of taxa rare in abundance (SRA), calculated as the number of taxa representing 5% or less of the abundance
5. Number of infrequently occurring taxa (SRF), calculated as the number of taxa occurring at less than three sites
6. Beta diversity calculated from abundance data (e.g. Bray-Curtis similarity)

McIntyre's dominance was also noted as a potentially useful indicator of stress.

Tuck et al. (2009) discuss the use of routinely collected fisheries data (e.g., commercial fisheries returns and the MPI fisheries observer programme) for fish diversity information. With respect to biodiversity of the benthic systems, there are a number of problems with using this information, including the varying taxonomic resolution and accuracy achieved by the observers (see Section 3.1.1), and the varying amount of benthos collected depending on gear type and the way the gear is deployed (see Section 3.2.2). Trawl survey data would be less problematic if gear type and use were standardised across all fisheries but regardless of this only a sub-set of the available benthic species will be collected. This issue will impact on all aspects of biodiversity measurement but most importantly on the metrics SRA and SRF (the number of taxa rare in abundance and the number of infrequently occurring taxa, respectively). That is, rare and infrequently occurring taxa are less likely to be sampled by fisheries trawls because this gear is not designed to sample the benthos. Conversely, core, sled and video data can all be analysed for biodiversity indices, although differences will be observed between data collected by the three methods due to differences in the size of organism collected, and, for the video, whether they are epifauna or infauna.

Biological traits

Aspects of ecosystem functioning and resilience can be represented by considering a range of organism attributes, each of which encompasses a number of 'biological traits' (Bremner et al. 2003, Hewitt et al. 2008, de Juan & Demestre 2012). In their review of potential responses of benthos to fishing, Thrush & Dayton (2002) include a number of biological traits that appear to be sensitive to physical disturbance. Hewitt et al. (2011a) in their analysis of data from the 2007 OS 20/20 Chatham-Challenger project (MPI project ZBD200701A) summarised these traits and their responses to physical disturbance under a list of higher-level attributes (Table 3).

These traits are an abbreviated list developed specifically for the OS 20/20 analyses, and do not include some functionally important attributes such as size, age, rarity and population density. These were not considered practicable for the OS 20/20 analyses, nor for use in a deep-water monitoring programme, because: (a) information on these attributes for the New Zealand region is limited; (b) some of the information would need to be spatially explicit (e.g., population density) and thus would not fit into a general framework; or (c) they are more related to recoverability than sensitivity (e.g., rarity). The majority of the information required for the categories in Table 3 can be assembled from seabed video or epibenthic sledge sample data, whereas, based on results from the OS 20/20 projects (Bowden & Hewitt 2012), benthic data from fisheries trawl samples are less useful in this context.

Table 3: Biological traits of benthic taxa and their potential responses to disturbance (from Hewitt et al. 2011a).

Attribute	Trait	Response to disturbance (rationale)
Feeding	Scavengers and predators	Positive (provision of additional food source)
	Suspension, deposit, grazers	Neutral (this is a conservative interpretation as variability in the magnitude of positive or negative effects is likely to be dependent on location, disturbance regime and individual traits)
Habit	Erect	Negative (liable to breakage)
	All others	Neutral (other habits are encompassed in the analysis by attributes related to living position)
Mobility	Sedentary	Strongly negative (unable to move away from approaching disturbance)
	Limited	Negative (may be able to move away)
	High	Neutral (able to move away from, or bury below, approaching disturbance)
Living position	Sediment surface	Strongly negative (will be disturbed)
	In top 2 cm of sediment	Negative or neutral (dependent on depth of disturbance)
	Deeper than 2 cm in sediment	Negative or neutral (dependent on depth of disturbance)
Fragility	Very fragile	Strongly negative (will be damaged/killed if disturbed)
	Fragile	Negative (will be damaged if disturbed)
	Robust or not known	Neutral

Ecosystem function

The aggregate response to disturbance of taxa in a system, across all functional attributes, can cause modification of ecosystem functioning (de Juan et al. 2009, Puig et al. 2012). Disturbance through bottom fishing activities, such as dredging and trawling, directly impacts commercially targeted species but also has direct and indirect effects on benthic habitats, the resident biota, and, thus, key ecosystem functions (Thrush & Dayton 2002). These effects include the modification of sediment substratum characteristics through sediment removal and turnover (e.g., changes in grain size, chemistry, and organic content), and damage or destruction of large habitat-forming epibenthic species. Here, we focus on a condensed set of taxon attributes that affect nutrient recycling (within the sediment and between the sediment and water column), dispersal, longevity, sediment stabilisation or destabilisation, trophic levels, feeding characteristics, and size. This set of attributes was initially compiled for New Zealand marine taxa for use with disturbance models being developed under MPI project ZBD200925, *Predicting impacts of increasing rates of disturbance on functional diversity in marine benthic ecosystems* (Lundquist et al. 2013). These disturbance models are based around the differential susceptibility and recovery rates of benthic taxa categorised into eight functional groups on the basis of their functional traits:

1. Opportunistic early colonists – limited substrate disturbance
2. Opportunistic early colonists – considerable substrate disturbance
3. Substrate stabilisers
4. Substrate destabilisers
5. Shell hash-creating species
6. Late colonisers – emergent epifauna
7. Late colonisers – burrowers
8. Predators/scavengers

The project (ZBD200925) analysed benthic community data collected during the 2007 OS 20/20 surveys of Chatham Rise and Challenger Plateau and the results demonstrated that groups 4–8 above can be adequately characterised by beam trawl and towed video sampling, but that groups 1–3 (based on smaller animals living in the sediment) need finer-resolution sampling using, for instance, sediment corers or high-resolution still camera imagery.

Ecological integrity

Ecological Integrity measures are generally separated into those representing categories of nativeness, pristineness, diversity, and resilience (Schallenberg et al. 2011, Thrush et al. 2011). Table 4 here lists the measures suggested by Thrush et al. (2011) as indicators that can be measured for the marine environment, and that might be applicable to the deep sea.

In the Nativeness category (Table 4), Items 6 and 7 would be assessable from general information and Items 1–4 could be addressed through the MPI fisheries observer programme. Item 5 is problematic with any of the data normally collected.

In the Pristineness category, Items 1–3 are general information used to set context and allow comparison between different geographic areas. Items 4 – 6 are associated more with species in the water column or air and thus not directly relevant to this project. Item 7 can be determined with data from beam trawls, epibenthic sleds, or video (Bowden & Hewitt 2012) and, potentially, multibeam acoustics. Item 8 would require benthic sampling but might potentially be covered by the MPI fisheries observer programme. Items 9 and 10 are not readily assessable by any of the present sampling regimes, although possibly Item 10 might be determined in some cases by video transect analyses (i.e., observations of suspension-feeding organisms affected by sedimentation) or, potentially, backscatter from multibeam or other acoustic surveys.

Items in the Diversity category were devised specifically to be measured using video, although for Items 8, 9, and 10, there is potential to use fisheries observer or trawl data (Thrush et al. 2011). Item 8 refers to ‘key’ species, such as habitat-forming corals, that can be easily monitored indicators of higher overall community diversity. Item 9 relates to the expectation that community diversity will be lowered in habitats in which one or more species (e.g. scampi) is commercially exploited.

In the Resilience category, Items 1–4 can be determined by most of the collection techniques, although Item 2 would best be undertaken using methods that collect both infauna and epibenthos. Items 5 and 6 can be determined by a mix of video and biological trait analysis. Items 7 – 10 cannot be assessed at present, but it might be possible to address Items 9 and 10 as consistent time-series are built up over the course of a longer monitoring programme (‘Flickering’ in time-series data refers to increased short-term variability which can be an indication of impending ecosystem instability (Anderson et al. 2008a)).

Table 4: Ecological integrity measures for the marine environment (adapted from Thrush et al. (2011)). Table cells are colour-coded to indicate whether they are: achievable and useful in a deep-water monitoring programme (no shading); potentially useful but problematic to measure at present (light shading), or probably not practicable (dark shading). Numeric labels in the 'Item' column serve to differentiate between table rows and do not imply priority ranking. See text for details on each item.

Item	Category			
	Nativeness	**Pristineness**	**Diversity**	**Resilience**
1	Invasive species outbreaks	Broad scale oceanographic features	Diversity of visible organisms and traces	Production to biomass (or size) ratios
2	Invasive species recognised as major threats	Mixed layer depth	Functional trait diversity of visible organisms and traces	Food-chain length and trophic diversity
3	Nuisance species outbreaks	Eutrophication status	Compositional variability of visible organisms and traces within habitats	Presence of large and old organisms
4	Presence or spread of invasive habitat-forming species or invasive bioengineers	Assemblages of marine mammals; sea and shore birds; large predatory fish and invertebrates	Compositional variability of visible organisms and traces across habitats	Redundancy within functional groups
5	Unusual events not included above (e.g., harmful algal blooms, disease outbreaks, die offs)	Number of species listed as threatened or at-risk under the New Zealand Threat Classification System	Species richness	Resistance to disturbance
6	Change to the natural disturbance regime (i.e. change from natural state)	Status of threatened or at-risk species in region	Biological traits	Variability in spatial structure of community composition
7	Intensive marine activity (e.g., fishing, mining, presence of unnatural underwater noise)	Number, areal extent, and diversity of habitat types	Compositional variability in community structure	Transport vs recycling of energy and matter
8		Quantities of marine litter (plastics)	Key species	Maintenance of feedback processes
9		Resident organism contaminant levels	Exploited communities	Recovery rates
10		Sediment draping of organisms and surfaces	Fish species diversity	Flickering in time-series

Thus, many metrics, indices, and criteria have been, and are being, developed for evaluating ecosystem status, any or all of which might be appropriate for application in a benthic monitoring programme (see e.g., de Juan et al. In press). Deciding which subset of these to use in practice will be important once the decision to proceed with a monitoring programme has been made. However, the more fundamental task to be addressed in the preliminary stages is to ensure that the survey itself is designed in such a way that the raw data collected will be of an appropriate type, and embedded in an appropriate, statistically valid, structure, to enable any or all of these approaches to be implemented.

3.1.4 Survey design options

High-level choices around survey design for a monitoring programme centre on deciding: (a) which component(s) of the benthic community will be monitored and, therefore, what sampling gear will be used; (b) the region(s) where the survey(s) will take place; (c) the sampling design and spatial extent of the survey(s) within the region(s) and, in particular, what ecologically relevant strata exist within it, and (d) what level of replication will be needed to be able to detect changes of a given magnitude between sampling events. These choices, in turn, depend upon the higher level management objectives that drive the need for monitoring benthic invertebrate communities. Regardless of the actual design employed, the type of sampling gear used and the number of gear deployments at each time-step of the survey are the principal factors that will determine the overall cost, and thus the extent and feasibility of any proposed monitoring survey.

Sampling method

Available methods for sampling marine benthos in the deep sea can be assigned to three broad categories: sleds, trawls, and dredges for sampling surface-dwelling organisms (epifauna); corers and grabs, primarily for sampling organisms within sediments (infauna), and cameras, for sampling epifauna (Eleftheriou & McIntyre 2005). While most of these methods have been employed in research since the earliest days of benthic investigations, photographic sampling is being adopted increasingly across a range of exploration, survey, and monitoring roles (e.g. Bax et al. 1999, Collie et al. 2000, Sheehan et al. 2010, Kipson et al. 2011, Compton et al. 2012, Bowden et al. 2013b) as imaging technology becomes more versatile and economical to use (Rhoads et al. 2001, Solan et al. 2003).

Based on their analyses of samples and data from the first OS 20/20 surveys to Chatham Rise and Challenger Plateau, Bowden & Hewitt (2012) concluded that sampling of mega-epifauna (sensu Grassle et al. 1975; surface-dwelling fauna identifiable in camera imagery) by means of a towed camera system provided more useful information for broad-scale mapping of benthic community structure than did data from the beam trawl and epibenthic sleds that were used at the same sites (Bowden 2011). In the published literature there are also valuable examples of comparisons between benthic sampling methods in terms of both resource costs, ecological relevance, and effectiveness for use in monitoring applications. For instance, in the UK, Rogers et al. (2008) conducted a resource analysis of six sampling methods used in parallel in an extensive survey of offshore soft-sediment benthic assemblages, and assessed trade-offs between the time required for sample collection and processing and the numbers of replicate samples required to detect a given change in a range of community metrics. Their results, summarised in Table 5 here, indicate that sampling macro-infauna had the highest overall costs and that, whereas both meiofauna and mega-epifauna had lower costs for collection and processing, mega-epifauna was likely to yield higher statistical power for a given sampling effort.

Table 5: Summary of comparisons between three benthic faunal components (meiofauna, macro-infauna, and mega-epifauna) in terms of collection and processing time, the number of samples required to distinguish between sites using a range of univariate and multivariate metrics, and the resulting statistical power to detect change of a given magnitude. Comparisons are summarised as either low (L) or high (H). Adapted from table 5 in Rogers et al. (2008).

Faunal component	Sampling method	Collection and processing time	Number of samples required	Statistical power
Meiofauna	Multicorer	L	H	L
Macro-infauna	Multicorer, Van Veen and Day grabs	H	H	H
Mega-epifauna	Beam and Agassiz trawls	L	L	H

For an ‘optimal’ survey designed to detect changes over time, we assume use of non-destructive sampling using camera systems, potentially in combination with minimally destructive methods such as coring if macrofauna, meiofauna, or sediments are deemed to be useful indicators of change. The principal arguments for this are that: (1) destructive sampling is inappropriate for monitoring of sessile and sedentary fauna because the sampling method would introduce disturbance effects similar to those the study is designed to measure; (2) photographic sampling enables measurements of small-scale spatial structure and patch characteristics that are not possible using available physical sampling methods, and (3) more reliably quantitative data are generated than is the case with sledges and trawls that sample epifauna at similar scales (Bowden & Hewitt 2012).

For deep-water surveys in the New Zealand region at present, this effectively restricts the choice of primary sampling method to a towed camera system, of which NIWA’s Deep Towed Imaging System (DTIS, Hill 2009) is the best locally available, providing both high resolution colour video and still images along seabed transects of about 1.4 km. While it is certain that more sophisticated platforms and imaging systems will become widely available in future, it is important for any monitoring programme that the methods used should be comparable across all years of the survey and, thus, that practical, repeatable, standards should be established at the outset. For photographic methods, this demands that camera orientation (angle of the optical axis in relation to the plane of the seabed) and imaged seabed area should be consistent, and that image resolution should be at the highest level practicable using current technology (Bowden & Jones In press). The latter point is a particularly difficult issue to plan for because improvements in the optical resolution of imaging systems can result in ‘new’ taxa appearing in the time-series data simply because they were not detectable in earlier surveys using older technology. However, this effect can be tested for by using paired analyses, one of which is based on the taxonomic and image resolution available at the start of the monitoring.

Survey location

As an example location on which to base our designs for a hypothetical long term monitoring programme, we use Chatham Rise. This is because this feature has the most spatially and temporally complete existing information about benthic habitats (Probert et al. 1997, Carney 2005, Bowden 2011, Compton et al. 2012, Leathwick et al. 2012, Nodder et al. 2012), is the site of some of New Zealand’s major commercially important fisheries (Clark 1999, O'Driscoll et al. 2011), notably for hoki (*Macruronus novaezelandiae*) and orange roughy (*Hoplostethus atlanticus*) and, arguably, encompasses some of the most vulnerable benthic habitats within the EEZ (Clark et al. 2000, Clark & Rowden 2009, Williams et al. 2010). Chatham Rise is a large bathymetric feature, measuring more than 500 nautical miles (NM) west to east and more than 120 NM north to south, rising from depths deeper than 3000 m to shallower than 200 m. It would, therefore, be impractical to attempt to monitor all benthic habitats present. Commercial trawl fisheries currently operate in depths down to about 1200 m (Black & Wood 2014), thus this depth could be used to define an appropriate maximum extent for the survey design. However, deeper fisheries might be developed in the future, and there is potential for trawling effects to occur outside of the main fishing footprint, including impacts on fauna at depths deeper than the main fishing horizon (Puig et al. 2012). Therefore it would be necessary to develop an approach for selecting which habitats and sites are to be sampled that are relevant to both ecosystem and fisheries approaches to environmental management.

Survey design and extent

Van de Meer (1997) evaluated three approaches to designing monitoring programmes for marine benthos in terms of their power to detect change with time: (1) random station placement in each year of the programme; (2) random placement of stations in the first year, which are then revisited in subsequent years, and (3) fixed non-randomly selected stations that are revisited in all subsequent years. Based on analyses of statistical power (Cohen 1988, Quinn & Keough 2002), primarily using Analysis of Variance (ANOVA) on univariate metrics, he concluded that: "… *a random selection of stations in the first year, which are revisited in succeeding years, seems to be the most appropriate design for a monitoring programme for soft-bottom marine benthos, where the primary objective is detection of change.*" This (i.e. Van de Meer's option 2) can be thought of as a form of 'repeated measures' or 'sentinel site' design, in that the same stations are re-sampled at each time point. That this is a different approach to the stratified-random designs used in conventional fisheries research trawl surveys reflects fundamental differences in the distributional characteristics of benthic and pelagic fauna. Benthic communities consist of organisms that are either sessile or have limited mobility in their adult phase, whereas demersal and pelagic fish populations are mobile over large areas throughout their life histories. Thus, an effective survey design for fish may not be appropriate for benthos because distributing benthic sampling randomly within spatially-extensive strata is likely to introduce a large 'exploration' component in each year of the survey, resulting in high variance in data within and between years.

These arguments notwithstanding, because a major incentive for a monitoring programme would be to assess long-term interactions between benthos and fisheries, and because considerable research effort has gone into developing and assessing broad-scale 'bioregionalisations' of the New Zealand EEZ in general and Chatham Rise in particular (Sharp et al. 2007, Snelder et al. 2007, Bowden et al. 2011, Compton et al. 2012, Leathwick et al. 2012), we explore two possible strategies for a sampling design to monitor change in benthic habitats and communities over time:

(1) a stratified-random design along the lines of a conventional research trawl survey, in which a broad-scale stratification scheme is used as the basis, with benthic sampling sites distributed randomly within each stratum at each time-point of the monitoring programme.

(2) a 'repeated-measures' design, comparable to Van der Meer's (1997) design 2, in which a number of relatively small-scale seabed areas are defined, *a priori*, either by random placement within broad-scale strata or distributed along environmental gradients, and revisited for benthic sampling at each time-point of the monitoring programme.

Considerations for a stratified-random design

The actual density of sampling required within each stratum of a stratified-random survey will be dependent on the extent of the survey area, the number and size of individual strata, and the variability of benthic habitats within those strata. To some extent, assessing the density of sampling required to detect change demands prior knowledge of the distribution of habitats, which we do not have for most of the EEZ. For Chatham Rise, however, power analyses based on the TAN0705 OS 20/20 data give an indication of the trade-offs involved in such decisions (Bowden & Hewitt 2012). Thus, at the scale of Chatham Rise, some estimation of the sample density required to detect spatial differences between benthic habitats is possible. Before the density of sampling per stratum can be determined, however, it is first necessary to decide on the spatial extent of the survey area and the number of strata. We considered four existing classification/stratification schemes for Chatham Rise:

- Fisheries Management Areas (FMAs);

- BOMEC classes (Leathwick et al. 2012);
- Biotic Habitats (Floerl et al. 2012);
- Hoki and middle-depths research trawl survey strata (O'Driscoll et al. 2011).

FMAs were discounted because they are at a larger spatial scale than is likely to be useful for characterising benthic habitats and communities or detecting temporal change in them. The BOMEC (Figure 6) and the Biotic Habitats (Figure 7) classifications were initially thought likely to be the most appropriate schemes on which to base a survey of benthic communities because both are derived from sampled distributions of benthic fauna; the BOMEC incorporating data collected by various methods over many decades (Leathwick et al. 2012) and the Biotic Habitats being based entirely on OS 20/20 samples from Chatham Rise (Floerl et al. 2012). However, consideration of these schemes in relation to historical data not included in their formulation (Kudrass & von Rad 1984) and more recent sampling in the region (notably an environmental survey conducted in relation to prospective mining of phosphorite nodules on the central crest of the rise, and the June 2013 OS 20/20 Chatham Benthos survey, voyage TAN1306) suggests that neither the BOMEC classes nor the Biotic Habitats adequately represent some significant scales of variability on the rise. For example, at approximately 180° E on the central crest of the rise there are relatively extensive hard substrate habitats colonised by the stony coral *Goniocorella dumosa* (Bowden et al. 2013a). In the Biotic Habitats and at the published 15 class-level of BOMEC (Figure 6, upper panel), which has been recommended for broad-scale classification (Leathwick et al. 2012), the crest of Chatham Rise appears as a single habitat class. At the full resolution of the BOMEC (Figure 6, lower panel), the central crest is more finely differentiated but this level still does not represent known distributions particularly well and presents a number of very small area classes, which might be impractical for use as survey strata.

Perhaps unexpectedly, the trawl survey strata (Figure 8) are quite consistent with what we currently know about the distributions of benthic habitats on Chatham Rise, and show some congruence with both the full resolution BOMEC (Figure 6) and the Biotic Habitats (Figure 7) classifications. This probably should not come as a surprise, however, because these trawl strata have been developed on the basis of many years of practical experience of sampling the seabed, both by fisheries researchers and by commercial fishers. Depth is widely recognised as a key factor in determining the relative abundance of demersal fish (Francis et al. 2002), and hence has always been an important factor in the stratification of fisheries trawl surveys. The trawl survey strata might, therefore, serve as a potentially useful basis upon which to design a benthic monitoring programme aimed at detecting trends over time related to fishing.

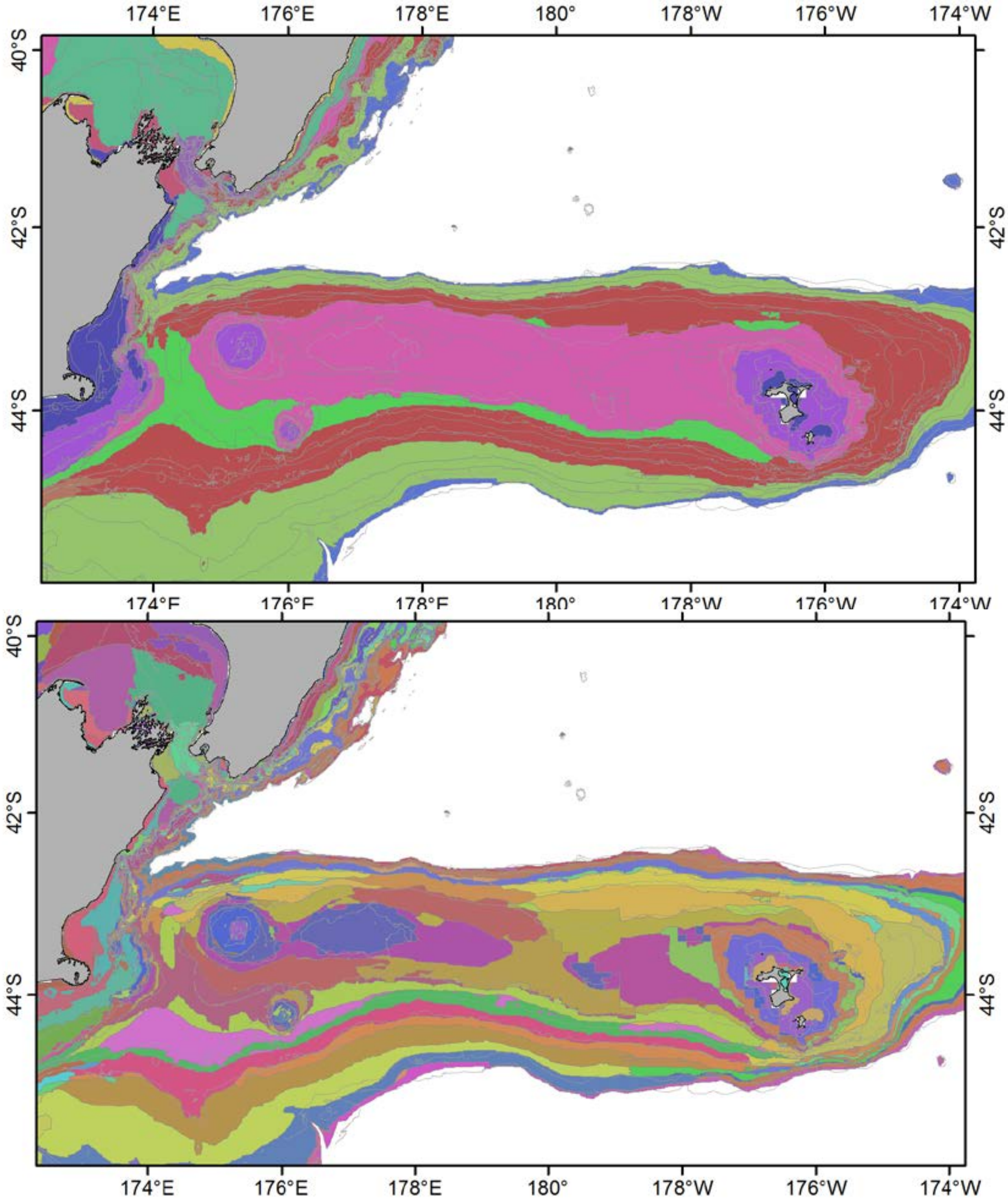

Figure 6: Chatham Rise, showing the Benthic Optimised Marine Environment Classification (BOMEC, Leathwick et al. 2012) at 15 class level (above) and full resolution (below).

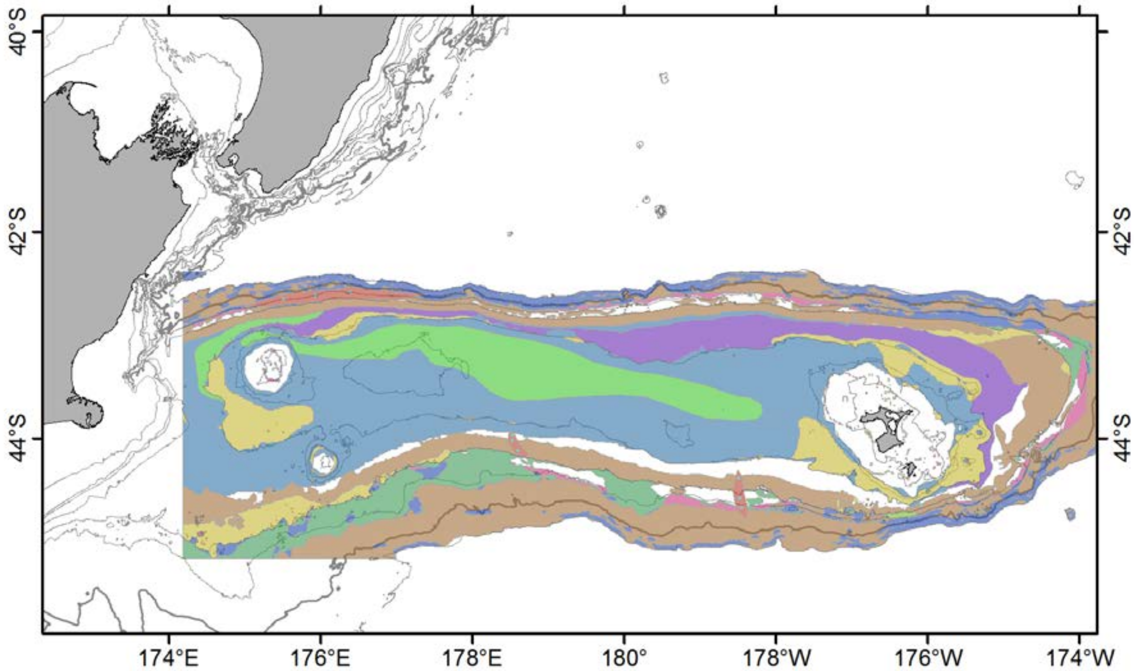

Figure 7: Biotic Habitats (Floerl et al. 2012) classification for Chatham Rise benthic habitats and fauna, based on towed video and epibenthic sled sample data from the Chatham-Challenger OS 20/20 project (voyage TAN0705).

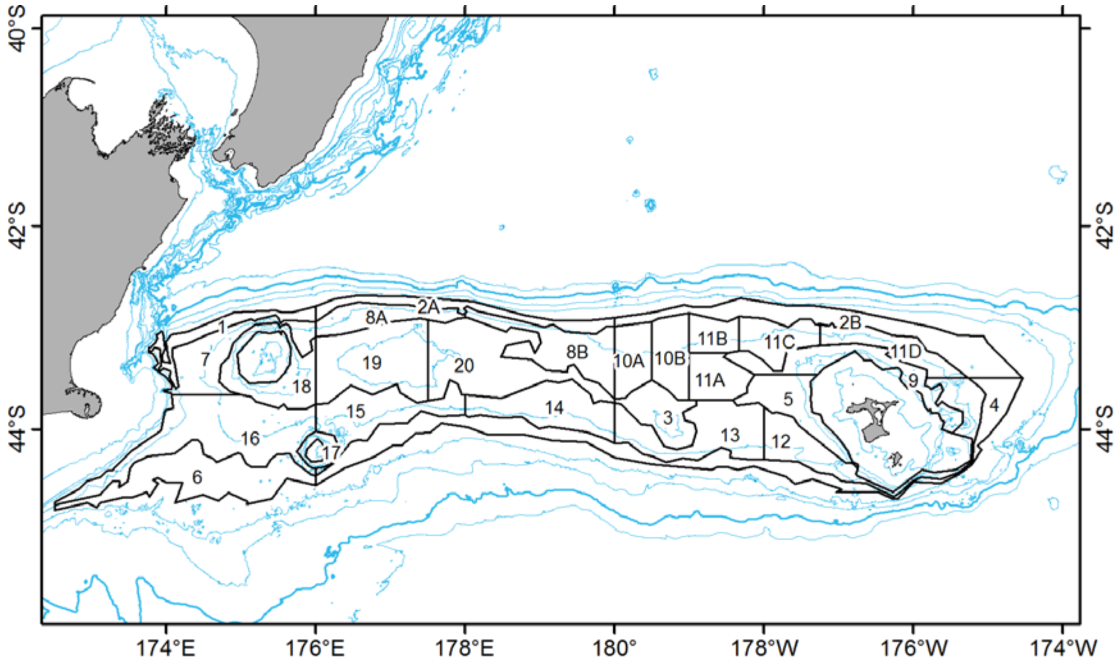

Figure 8: Chatham Rise hoki and middle-depths research trawl survey strata (year 2006, numbered black polygons).

Determining the number and type of samples

Both precision and cost increase with increasing sampling effort (Andrew & Mapstone 1987). Thus, any sampling design will involve a trade-off between the number of samples per stratum required to detect changes of a given magnitude over time, and the material and human resources available for the survey. In practical terms, this means that decisions must be made regarding: (a) the magnitude of potential change that is relevant to the objectives and thus must be detectable; (b) the budget that can be allocated to the survey, and (c) as a product of these two considerations, the spatial area over which the survey will operate. That is, given a finite budget and a required number of samples per stratum to detect a given change, what is the maximum number of strata that can be included in the survey?

Power analyses developed from the 2007 OS 20/20 survey of Chatham Rise (Bowden & Hewitt 2012), provide our best estimate of the level of sampling required to detect changes between spatial strata in the study region. They suggest that ideally eight or more samples are needed per stratum when using the "Biotic Habitats" scheme of Floerl et al. (2012) for stratification (power analysis, Figure 9). However, we can also use the 2007 OS 20/20 survey data to generate an estimate of the number of samples required to gain a given level of precision for the 2006 trawl survey strata. Plots of Standard Error (SE, as a measure of precision) for each of six benthic community metrics against the number of samples per trawl stratum again, unsurprisingly, show increasing precision (decreasing SE) with increasing sampling effort (Figure 10). Calculation of effect size, in terms of the dissimilarity between benthic communities within each stratum based on the OS 20/20 data, for this combination of samples and strata yielded a range from 0.53–0.74. This result is similar to that obtained by Bowden & Hewitt (2012) using the original OS 20/20 survey strata (effect size range 0.54–0.86) but lower than that using the Biotic Habitats as strata (Floerl et al. 2012) (effect size 0.78–0.83). Given this similarity in overall range of effect size, the power curves developed by Bowden & Hewitt (2012) based on these earlier stratification schemes (Figure 9) are also an appropriate guide for planning the intensity of sampling when using the trawl survey strata.

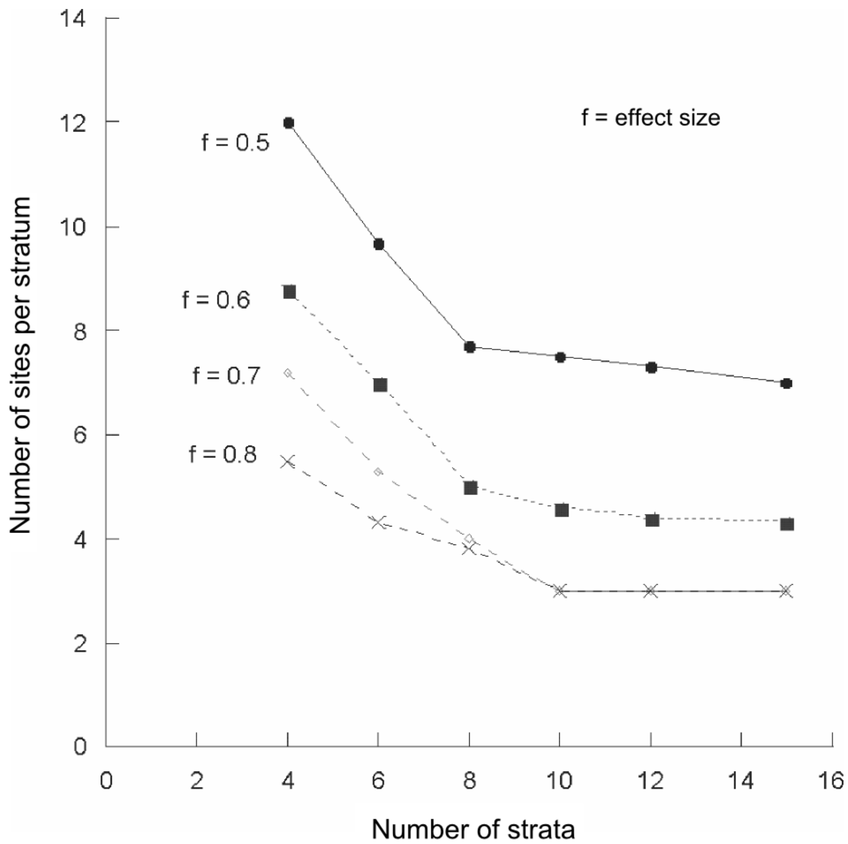

Figure 9: Power analysis from the Chatham Rise OS 20/20 survey (TAN0705): the number of sampled sites required per survey stratum to achieve an 80% probability of detecting differences between strata. Values are shown for four effect sizes (f = 0.5 to f = 0.8) as proportional differences between the mean values of a range of benthic community metrics (e.g. diversity, evenness) between habitats. Thus, for example, if there were a total of 10 strata in the survey area, each of which was, on average, 50% different from the others for a given assemblage metric (i.e. effect size, f=0.5), more than 7 sites would need to be sampled within each habitat to achieve an 80% probability of differentiating between them. (Figure adapted from Bowden & Hewitt 2012).

For the purpose of the present study, however, it is important to note that these analyses provide guidance for the level of sampling effort required to enable discrimination between strata (habitats) at a single time point (i.e. within a single survey). The goal here, by contrast, is to determine the intensity of sampling that would be required reliably to detect benthic community change in a given stratum over successive years. Such changes are likely to be of smaller magnitude than those between strata (in part, because the strata/habitats we use have been defined on the basis of observed differences in survey data in the first place and, therefore, only represent the resolution of the input data) and thus their detection is a more challenging problem. Given this situation, it is probable that greater sampling intensity will be required for the detection of community change over time.

Taking the smallest effect size in the analyses of Bowden & Hewitt (2012), i.e. a 50% difference between strata, as the minimum required detection power for monitoring, the power curves developed using the OS 20/20 video data and the Biotic Habitats classification as strata (Figure 9) indicate that at least eight samples per survey stratum would be required. When the trawl survey strata are used with the same sample data, however, for some strata at least, there can be considerable within-stratum variability even at this level of sampling (Figure 10). Given that an effect size of 50% is likely to represent a relatively major change in community composition in the context of an annual or biennial monitoring programme (Rogers et al. 2008), this suggests that an optimum benthic monitoring survey design should be planned on the basis of at least ten video transects per survey stratum.

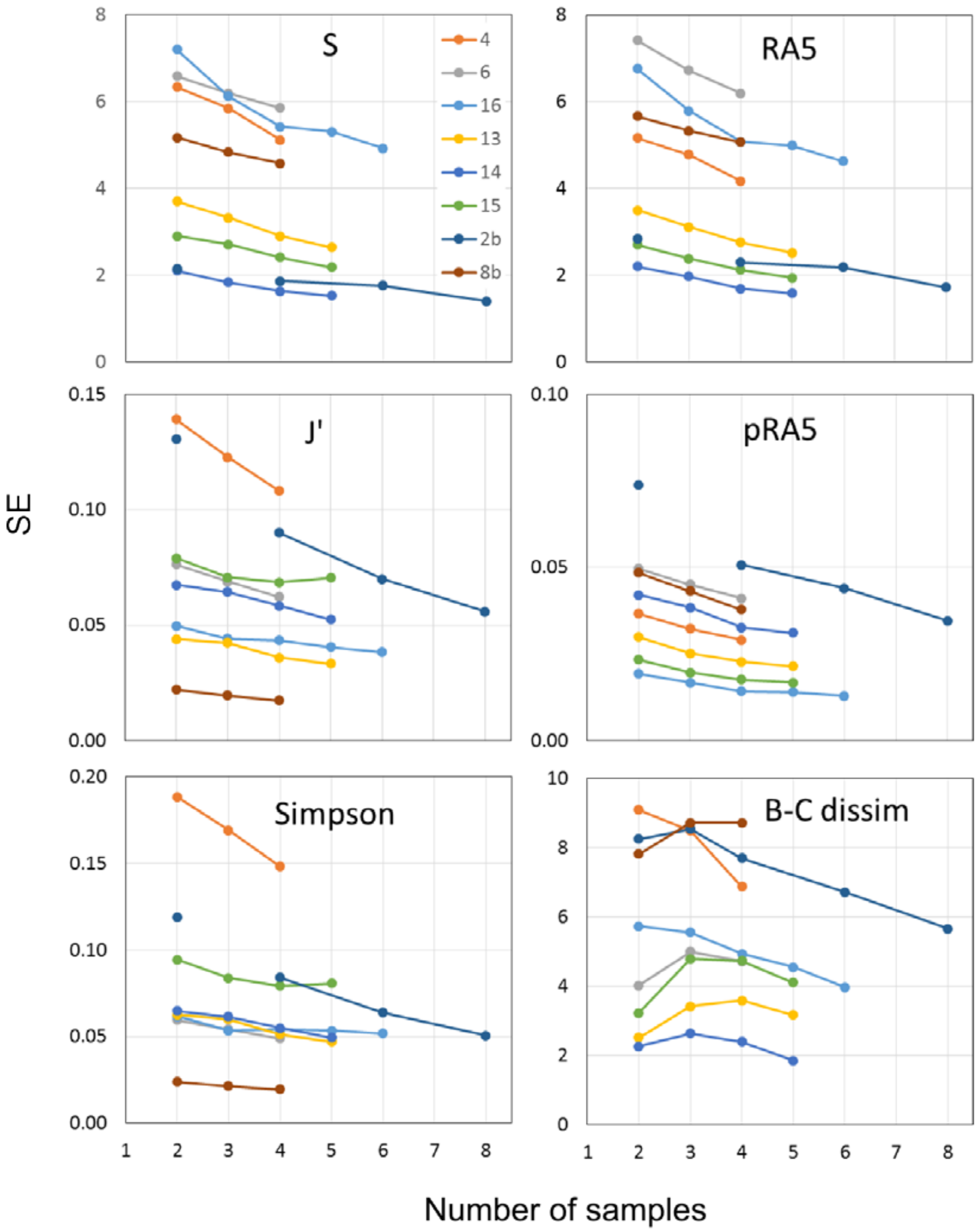

Figure 10: Chatham Rise: variability (as Standard Error, SE) of benthic community metrics in relation to number of samples collected within a survey stratum. Community data are from DTIS video transects collected during OS 20/20 survey TAN0705 and strata are those used for the 2006 hoki and middle-depths research trawl survey (legend in top left panel shows stratum numbers; see previous map figure for their locations and extents). S; number of taxa, RA5; number of taxa rare in abundance, J'; Pielou's evenness, pRA5; proportion of taxa rare in abundance, Simpson; Simpson's Diversity (1-λ), B-C dissim; Bray-Curtis dissimilarity. Compare with figure 7 in Bowden & Hewitt (2012).

Considerations for a repeated measures design

The principal potential advantage of a repeated measures design for a monitoring programme, as opposed to a stratified-random one, is that the spatial distribution of sampling is likely to be more appropriate to the characteristics of benthic communities. In contrast with pelagic and demersal fauna, such as fish, which are characterised by high mobility across scales of kilometres to hundreds or thousands of kilometres, benthic fauna have limited or no mobility in their adult form. Most benthic taxa either move slowly across the seabed (vagile taxa, e.g., echinoderms, crustaceans, gastropods), are stationary in or on sediments (sedentary taxa, e.g., polychaetes, infaunal bivalves, burrowing anemones), or are permanently attached to hard substrata (sessile taxa, e.g., corals, sponges, bryozoans). Their distributions can also be strongly influenced by variations in substrate type across a range of scales, and historical disturbances can have long-lasting influences on community composition at local scales. Because these factors can result in patchy distributions that change only relatively slowly (up to thousands of years for some sessile, habitat-forming taxa such as corals), re-sampling the same patches over successive time-steps is likely to be a more appropriate way to monitor change (Van der Meer 1997) than is random allocation of sampling distributed across larger spatial strata.

Staying with Chatham Rise as the target area, we explored the potential for basing a repeated measures-style design on the survey sites mapped and sampled by the 2013 OS 20/20 Chatham Benthos survey (TAN1306, MPI project ZBD2012-03). The TAN1306 survey was designed to enable detection of variations in benthic community structure across gradients of cumulative trawl fishing intensity (Bowden et al. 2013a). The sampling units of the study were 10 × 10 km seabed boxes that were defined *a priori*, aligned along gradients of trawl fishing intensity (Figure 11). Each box was first mapped by multibeam echosounder (MBES), yielding detailed bathymetric and acoustic backscatter data at 25 × 25 m grid resolution, and then sampled by three DTIS camera transects, and three multicorer deployments targeted on the DTIS lines. The aim was to gather data on benthic habitats and communities across a broad range of spatial scales and faunal sizes, from kilometres (MBES and DTIS video) to millimetres (sediments and meiofauna in multicore samples) and thus provide the best chance of detecting differences between boxes. Because DTIS transects were about 1.5 km long and the survey boxes were 10 × 10 km in size, the three transects within each box represented a considerably higher density of sampling than could be achieved in a stratified-random survey design for the same sampling effort.

For a monitoring programme, the OS 20/20 Chatham Benthos survey boxes (Figure 11) in the Mernoo and South areas (i.e. areas where trawl fishing for hoki takes place) would be revisited at appropriate intervals (annual, biennial, or longer) and re-sampled using either the same methods as in the original survey (video and multicorer), or potentially just the three video transects per box. The appropriate gear or combinations of gears to be used would be determined on the basis of results from the OS 20/20 Chatham Benthos project itself. Thus, if the best discrimination between sites were to come from, say, meiofauna community data, the multicorer would be most appropriate for monitoring, whereas if epifauna data provided the best discrimination, video or still image transects would be most appropriate.

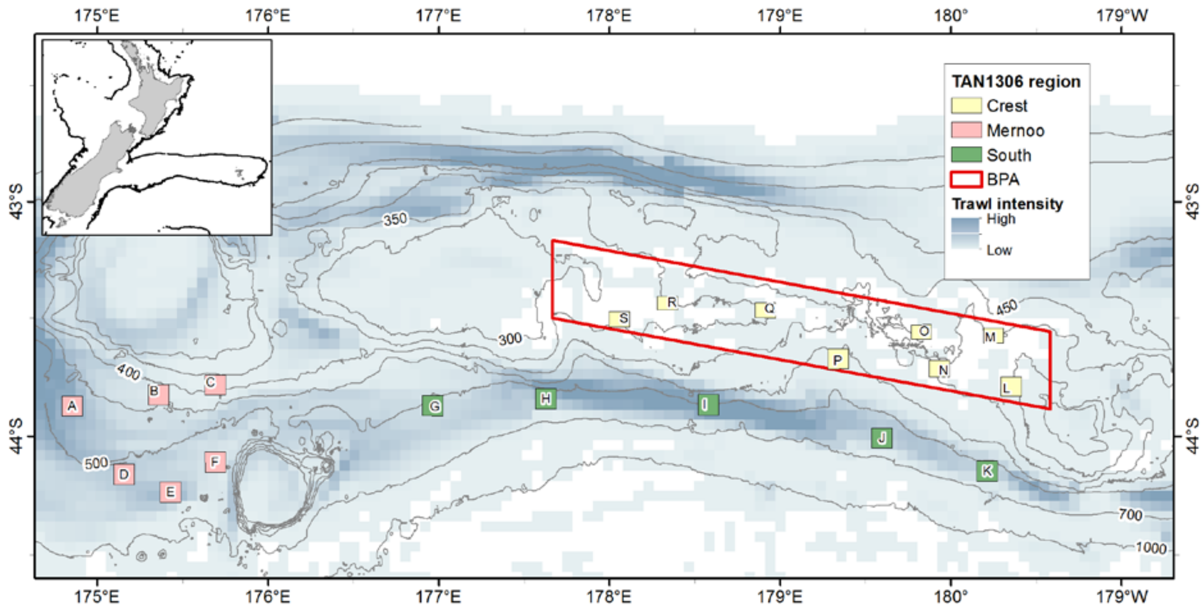

Figure 11: Benthic survey design for the Ocean Survey 20/20 Chatham Benthos project (ZBD2012-03). Survey units consist of approximately 10 × 10 km boxes (coloured polygons, labelled A–S) distributed across gradients of trawling intensity within each of two areas ("Mernoo" and "South"), and within the Central Chatham Rise Benthic Protection Area (BPA) on the crest of the rise ("Crest"). Boxes were mapped using a Kongsberg EM302 multibeam echosounder and within each box, three camera transects and three multicorer samples were collected.

3.1.5 Costing an optimal monitoring programme

The spatial extent of either of the survey designs outlined above would be most appropriately decided by consultation between fisheries managers and research scientists working within the constraints of the budget available. The cost of such a monitoring programme would, thus, be dependent on high-level decisions associated with its location, purpose, extent, and the magnitude of change to be detected. While these considerations make it impractical to provide a comprehensive costing here, indicative, relative costs for collecting and processing benthic samples using a range of methods can be gathered from the literature (see e.g., Rogers et al. 2008, and Table 5) and from recent practical experience in New Zealand waters, particularly with OS 20/20 surveys but also other biodiversity-orientated sampling initiatives (Table 6).

Table 6: Indicative cost (as person-hours, exclusive of vessel time) per unit sample for four sampling methods of potential use in a benthic monitoring programme: DTIS, Deep Towed Imaging System; TB, beam trawl; CM, multicorer; RT+NB, rat-catcher trawl with ground-rope net bag attached. Costs for each method are divided across the five distinct phases of work: at-sea collection; post-voyage processing of samples for taxon identification and counts; curation and archiving of identified specimens; analysis of data to identify spatial patterns and temporal trends, and reporting of results.

	Sampling method			
	DTIS	**TB**	**CM** [c]	**RT+NB**
Collection (all at-sea operations)[a]	9	9	9	9
Processing (identifications and counts)	25[b]	5	35	3
Curation & archiving (storage and databasing)	0.5	15	15	7.5
Data analysis (for spatial pattern and trends) [d]	1	1	1	1
Reporting (outputs to managers and peers)	0.5	0.5	0.5	0.5
Total (h)	36.0	30.5	60.5	21.0

Notes: [a] at-sea operations based on requirement for three science staff for all deployments; [b] DTIS processing based on 10 h (1.5 days) for video and 15 h (2 days) for stills (60 images @ 15min per image); [c] assumes analysis for macrofauna only, and two cores per deployment, the top 10 cm of each processed as two 5 cm depth intervals, sieved at 300µm; [d] for all methods, assumes analysis is on full dataset thus unit cost per sample is low.

Returning to the stratified-random example outlined above, if all twenty-six strata in the research trawl survey scheme were to be monitored with ten camera transects in each stratum, it would require 260 video transects per survey. The first OS 20/20 survey to Chatham Rise (voyage TAN0705) collected a total of 431 benthic samples over four weeks, of which 112 were DTIS transects (Nodder 2007). The second OS 20/20 survey to Chatham Rise (voyage TAN1306) collected a total of 103 benthic samples over three weeks, of which 57 were DTIS transects (Bowden et al. 2013a). In current voyage planning, NIWA bases its survey estimates on completing an average of eight DTIS transects per 24 h working day, assuming no other gear deployments and suitable sea-state. Thus, to survey all twenty-six strata with ten camera transects in each would require commitment of a suitable vessel and support staff for about 32 days at each time-step of the monitoring. This would be a significant commitment of resources. A more likely scenario is that a sub-set of the trawl survey strata would be selected for benthic monitoring.

With the repeated measures example above, if all nineteen survey boxes surveyed during the TAN1306 OS 20/20 Chatham were to be re-sampled using DTIS alone, each time step of a monitoring programme (e.g. annual re-sampling) would require about 7 full days (57 transects at 8 transects per 24 h day) of workable sea time at each time-step. Whichever design were to be used for a benthic monitoring survey, the spatial extent, the number of strata, and therefore the total number of individual samples, would need to be managed to keep costs conservatively within bounds that would be supportable on an annual or biennial basis for an indefinite period into the future.

3.2 Exploring the feasibility of using existing surveys to monitor benthic habitats

3.2.1 Determining the location, extent, timing and frequency of planned trawl and acoustic surveys

In 2010, the then Ministry of Fisheries published a National Fisheries Plan for deep-water and middle depth fisheries (Ministry of Fisheries 2010). The intention of this was to map out a 10 year research programme to deliver more consistent time-series data to support stock assessments. Under this plan, a number of surveys were proposed:

(a) Chatham Rise trawl survey at depths 200 – 800 m and 800 – 1300 m. This survey, takes place in January, almost every year (8 of 10 years), and is predicted to have a long-term future. If it is conducted using RV *Tangaroa* in a similar manner to recent surveys, it will fully utilise the vessel time available, and so days for any benthic monitoring programme would need to be added. The stratification is broadly consistent with what is known of benthic distributions (Section 3.1.4) and thus may be appropriate as the basis for a benthic monitoring design. The survey area encompasses locations previously sampled by the NIWA Seamounts Programme and OS 20/20 Chatham Rise surveys. Sampling inside and outside Benthic Protection Areas and across a range of trawling intensities is feasible.

(b) Sub-Antarctic trawl survey at depths 300 – 1000 m. This survey takes place over the November-December period on Campbell Plateau, with sampling confined to daylight hours (to minimise bias through vertical fish migrations). Thus, there is an opportunity to utilise some time during the night in areas where transits between trawl stations are short (e.g. Puysegur and Snares Shelf). The survey takes place in every second year, with the next in late 2014.

(c) West Coast South Island trawl and acoustic survey at depths 250 – 650 m. This is a new survey, so it is uncertain whether there could be any available time. It would seem likely, given the area to be covered by the combined survey, that time added on for monitoring would be needed. The survey will be in July of every second year if successful, although the initial planned survey in 2013 was delayed and it is uncertain when it will take place.

(d) Campbell Plateau southern blue whiting survey. This survey occurs to depths of 600 m in September, every second year (last survey was August 2013). Rough conditions are often experienced in this region at this time of the year and could be an issue for camera deployments (i.e. if too rough for acoustics, it will certainly be too rough for seabed camera systems). The survey location depends upon the distribution of fish aggregations, and this is variable between years. Hence it could be difficult to get robust comparable data between surveys.

(e) Southern Chatham Rise Oreo trawl survey at depths down to 1200 – 1300 m. These surveys in Quota Management Areas BOE 3A and SSO 4 are scheduled to be less frequent than the above. They take place in October-November corresponding to oreo spawning times. It is proposed that surveys of Quota Management Areas SSO 4 and BOE 3A could be carried out by two vessels, one doing acoustics (e.g., RV *Tangaroa*), and a catcher vessel conducting trawling. Some time on the acoustics vessel could be available for ancillary sampling.

(f) Jack mackerel off the coast of Taranaki (Quota Management Area 7). There was a pilot acoustics/trawl survey in 2011–12 conducted by RV *Tangaroa*. Results were mixed, as there were issues with the quantity of acoustic marks, and trawl adequacy for species identification. There is no certainty of it becoming a consistent longer time-series. Hence it may be low priority for any monitoring programme.

(g) Scampi trawl and photographic surveys. These surveys already use camera gear, and so it could be both feasible and useful to add some stations outside the fishing footprint. The surveys are scheduled to cover four fishing grounds (east and northeast coast of the North Island (February-March), Auckland Islands (March), and Chatham Rise (October).

(h) Northwest Hills orange roughy. These small seamount features on the northern flank of the Chatham Rise have been the target of benthic habitat surveys in 2001, 2006, and

2009. They have also been the sites of experimental research on orange roughy, in particular, the use of acoustic and camera equipment to study fish behaviour. The benthic habitat survey design has incorporated a combination of fished, unfished, and fished-and-now-closed hills to enable monitoring of changes over time in the benthic invertebrate community (see Clark & Rowden 2009, Clark et al. 2010). Any work in this region could continue this monitoring programme on Graveyard, Morgue, Gothic, and Diabolical seamounts.

Opportunities for using other surveys are less certain. Other possibilities under the MPI 10 year deep-water fisheries plan include surveys for black cardinal fish off the east and northeast coasts of the North Island, Puysegur orange roughy, and Cook Canyon orange roughy.

3.2.2 Determining the feasibility of a "tagged on" monitoring programme

General considerations

Because modifications to fishing gear during research trawl surveys that are part of an established time-series are not permissible, any 'tagged-on' benthic monitoring component would demand additional dedicated gear deployments. Minimum extra costs might be entailed for the 'tagged-on' component if benthic sampling were to be conducted using either the primary fishing gear, the primary gear with additional benthic 'net-bags' fitted to the ground rope, or a beam trawl or epibenthic sled that could be deployed on a single trawl warp. An alternative to these methods would be to use non-destructive photographic systems but these would also demand dedicated deployments and would entail greater complexity, and therefore cost, in terms of equipment, staffing, and vessel operations.

As discussed in Section 3.1, non-destructive (photographic) or minimally destructive (e.g. coring) sampling methods would be more appropriate than bottom-contact fishing gear or dredges in any benthic monitoring programme. Deep-sea camera technology is developing rapidly but at present, in New Zealand, the optimum camera system is NIWA's DTIS towed camera, which requires relatively sophisticated technical support, including either RV *Tangaroa* or RV *Kaharoa*. This factor alone considerably restricts the potential for using commercial vessels, which typically do not have winches and conducting cables suitable for the work. Moreover, modification of fish trawl gear by addition of extra benthic sampling components (e.g. ground rope net bags or headline cameras) would not be allowable for either research trawl survey or commercial operations because of its potential effects on target species' catchability. It would be conceivable, however, to use planned down-time during a fishing voyage (e.g. night or day if fishing for the target species of the main survey is restricted to one or the other) to deploy benthic sampling gear.

In cases where surveys have science staff on board as part of the routine fish survey team, they might also be used to carry out a benthic monitoring programme, but additional staff with specialist benthic identification and handling skills would be advantageous. It is probable that the likely utility of any "tagged on" monitoring will vary between the planned surveys, and thus the feasibility of different vessel, gear, area, and timing options would need to be assessed carefully. Given the need in any long-term monitoring programme for rigorous consistency of methods between time-steps, and for the spatial distribution of individual deployments to be matched to the characteristics of the fauna and habitats being studied, reliance on a "tagged-on" approach would only be viable if the requirements of the benthic survey were not compromised. Such compromises are likely to occur in relation to the type of sampling gear used and the sampling design. For instance, if the primary purpose of the voyage were a trawl survey, arguments might be made for saving time and staffing costs by use of a trawl-based sampling method rather than one better suited to sampling benthos. Similarly, if the movements of the vessel during the survey are primarily dictated by the demands of the trawl survey, it might be considered too costly in terms of time to superimpose on this track a set of benthic sampling sites that follow a different pattern.

Gear comparisons

A dedicated biodiversity survey will typically use a research vessel that is designed and set up for deploying sophisticated equipment that itself is especially designed for sampling a range of benthic animals. In recent years in New Zealand, biodiversity surveys have used a variety of sampling equipment with different target components of the fauna:

- Towed cameras (primarily NIWA's Deep-towed imaging system (DTIS) which can be deployed close to the seafloor, and be towed at slow speeds to record epibenthic macro- and mega-fauna.
- Multicorer/Box corer/grabs to sample the upper layers of the sediment, and capture infaunal macrofauna, meiofauna, and bacteria.
- Sleds and dredges that can sample hard or rocky seafloor and retain sessile, sedentary or slow-moving macro- and mega-faunal specimens.
- Trawls, ranging from small Agassiz trawls through various beam trawl sizes, to large commercial trawl net designs. These will typically retain larger macrofauna and megafauna, and with larger-meshed trawls also fast-swimming animals that will not be sampled by other gear types.

The selectivity of different sampling gear is well known (e.g. Eleftheriou & McIntyre 2005) and NIWA surveys will typically use combinations of towed camera, epibenthic sledge or beam trawl, and corer/grab that enable species of all sizes to be collected (e.g. Clark et al. 2010, Rowden & Clark 2010, Bowden 2011). However, the towed camera gear requires considerable technical expertise, and this combination of gear is generally not used in conjunction with large trawl gear or during fisheries surveys except where there are fish biodiversity objectives as well (e.g. Clark & Roberts 2008, Hanchet et al. 2008a, Rowden et al. 2012). In addition, the relative selectivity of each gear type for measuring different fauna has generally been accepted, but not analysed in detail to evaluate their relative effectiveness for tracking patterns in benthic community structure.

Trawl and net bag catch comparisons from voyage TAN1208

Data from *RV Tangaroa* voyage TAN1208 (see section 2.2.2) provide an opportunity to analyse the catch characteristics of three benthic sampling methods; beam trawl, rat-catcher trawl, and net bag, that might be practical for use on a fisheries-related survey.

The total biomass of benthic invertebrates collected using the three gear types ranged from 0.8 to 60.8 kg tow^{-1}. Sample biomass standardised for tow area (kg 0.1 km^{-2}) did not differ significantly between strata, gear types or their interaction (PERMANOVA, $P > 0.05$; Table 7). A total of 87 benthic taxa were identified across all gear types. The mean number of taxa per sample was 14 and ranged from 3 to 24. There was no significant main effect of gear on taxon richness, but strata and the interaction of gear and strata both had significant effects (PERMANOVA, $P < 0.05$; Table 7; Figure 12). Pairwise comparisons showed no significant differences between gear types within each of the stratum for taxon richness ($P > 0.05$). Variability in taxon richness did not vary significantly between gear types (PERMDISP, $P > 0.05$). Overall, taxon richness tended to increase from the shallow stations (S) to the deepest stations (D and X).

Table 7: Results of PERMANOVA analysis testing for the effects of gear (beam trawl, rat-catcher trawl, net bag), strata (S, M, D, X), and area on estimates of taxon richness and community structure of benthic invertebrates. [df = degrees of freedom, SS = Sum of Squares, Pseudo-F = Pseudo-F Statistic, P = probability, Perms. = number of unique permutations].

4 Source	df	SS	Pseudo-F	P	Perms.
Biomass (kg 100 m^{-2})					
Gear	2	2.39	1.72	0.222	999
Strata	3	2.39	1.22	0.165	961
Area (Strata)	8	5.20	0.94	0.603	997
Gear × Strata	6	3.83	0.92	0.543	999
Residual	16	11.105			
Total	35	24.915			
Taxon richness					
Gear	2	52.7	2.6	0.106	999
Strata	3	376.2	24.4	0.002	478
Area (Strata)	8	41.1	0.5	0.830	998
Gear x Strata	6	221.1	3.6	0.020	998
Residual	16	162.9			
Total	35	854			
Community structure					
Gear	2	41 470	9.54	0.001	999
Strata	3	24 412	3.41	0.001	968
Area (Strata)	8	14 796	1.38	0.009	998
Gear x Strata	6	30 392	2.33	0.001	997
Residual	16	34 791			
Total	35	150 110			

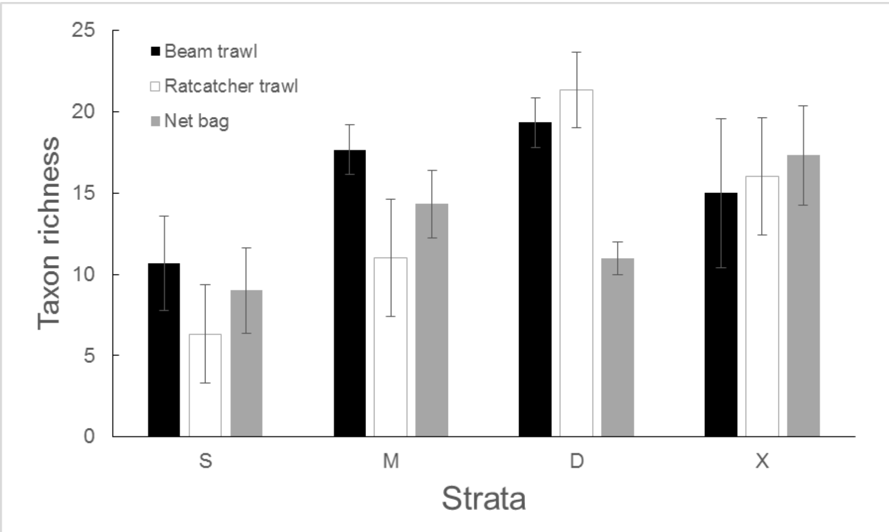

Figure 12: Mean taxon richness of benthic invertebrate samples collected using three gear types: beam trawl, rat-catcher trawl, and net bag attached to the ground rope of the rat-catcher trawl. Samples were collected in four depth strata (S about 350 m; M about 500 m; D about 700 m, and X about 1000m) on the central northern region of Chatham Rise. Error bars are standard deviation from mean (N = 3).

Estimates of community structure obtained from beam trawl samples were significantly correlated with estimates based on both rat-catcher trawl and net bag samples (RELATE, $P < 0.05$). The correlations were, however, relatively weak ($R^2 = 0.31$ and 0.07, respectively). There were significant effects of gear and strata and their interaction term, on estimates of benthic community structure (PERMANOVA, $P = 0.001$; Table 7, Figure 13). Gear had the strongest effect on community structure (44.2%), followed by stratum (27.8%). Pairwise comparisons showed all gear types to be significantly different from each other (P < 0.01); similarity was highest between rat-catcher trawl and net bag (14.7%) and lowest between beam trawl and both other gear types (2–4%). Multivariate dispersion of beam trawl data (mean deviation from centroid = 25.8) was significantly smaller than for rat-catcher or net bag data (mean deviation from centroid = 57.5 and 52.8, respectively) (PERMDISP with pairwise comparisons, P = 0.001, and evident in Figure 13).

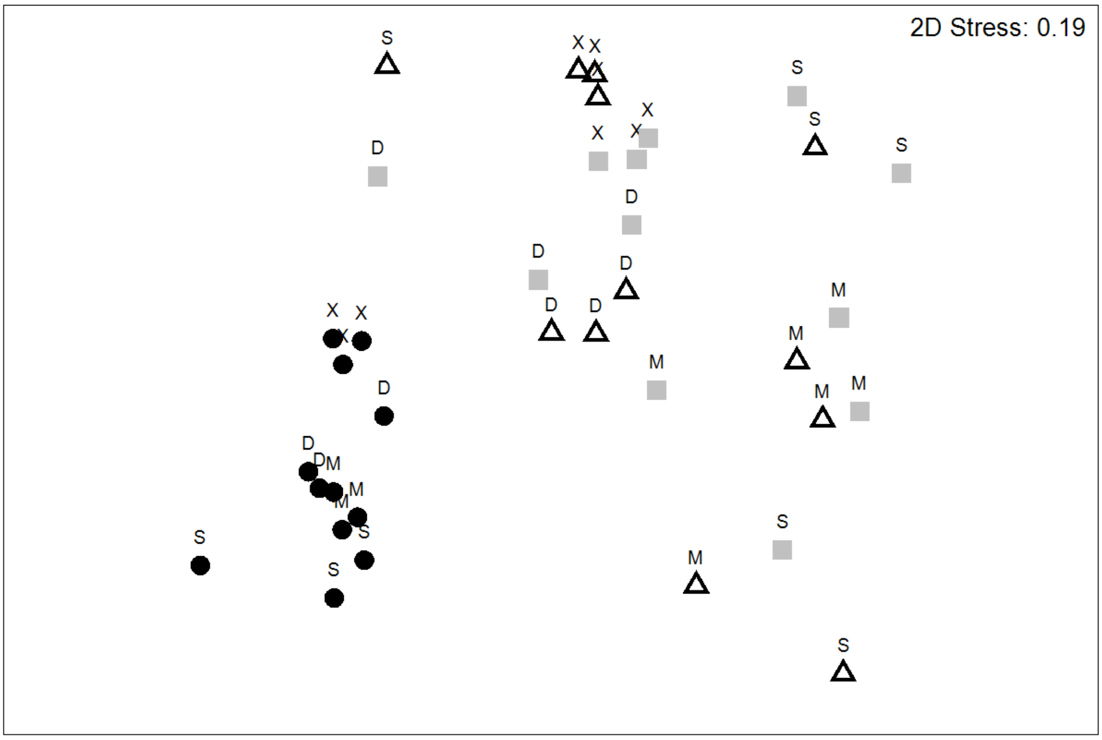

Figure 13: MDS ordination illustrating the degree of similarity between benthic invertebrate community samples collected using three gear types: Beam trawl (black circles); Rat-catcher trawl (open triangles); and Net bag (grey squares). Samples were collected across three depth strata: 300–400 m (S), 500–600 m (M), 700–800 m (D), and 950–1050 m (X).

SIMPER results show that similar taxa were responsible for the difference observed between beam trawl and rat-catcher trawl, and between beam trawl and net bags (Table 8). Beam trawl samples were characterised by greater biomass (catch per swept area) of almost all taxa. Echinoida (sea urchins), Gastropoda (snails), Holothuroidea (sea cucumbers), Decapoda (shrimp), and Asteroida (sea stars) contributed most to the differences between gear types. The only taxon more abundant in the rat-catcher trawl and net bags was the group Echinothuriidae and Phormosomatidae (Tam O’Shanters). Relatively small and/or fragile taxa, such as Polychaeta (bristle worms), Ophiuroidea (brittle stars), Anthozoa (sea anemones and corals), Hydrozoa (hydroids), Sipuncula (peanut worms), Isopoda (sea slaters), Galatheida (squat lobsters), Brachyura (crabs), and Scleractinia (stony corals) were well represented in the beam trawl samples but were either absent or rare in the rat-catcher trawl and net bag samples. Some of the larger taxa, such as Porifera (sponges), *Brisingida* (a genus of starfish), and *Stichopus mollis* (a sea cucumber) were relatively well represented in rat-catcher trawls and net bags but were more abundant in beam trawls.

Sampling methods used for monitoring benthic communities should provide samples that are: (1) representative (i.e., give an accurate description of what taxa are present and their relative abundance/biomass), and (2) consistent (i.e., provide estimates that are not so variable as to preclude any ability to detect ecologically meaningful change). Although the biomass and number of taxa caught by the different gear types did not vary significantly, the identity of these taxa did. The results of the analysis show that large and sturdy taxa that occur in low densities are likely to be sampled more effectively by rat-catcher trawl and net bag gear with wide openings towed over long distances, than with a beam trawl with narrower opening towed over a short distance. Some taxa were not found in beam trawl samples but were common in net bag samples. This difference may be due to the location of the net bags on the ground gear of the rat-catcher trawl, which may dig into the sediments and capture relatively large buried or semi-buried fauna. Compared with the beam trawl, the absence or near absence of many small but common benthic invertebrate taxa in the rat-catcher trawl and net bag samples was striking, but not surprising given the larger mesh size of the rat-catcher trawl cod end and net bag relative to the beam trawl.

In addition to the size selectivity issue associated with the rat-catcher trawl and net bag, greater variability in estimates of community structure derived from these gear types relative to the beam trawl was observed. This difference may be related to the haphazard way in which benthic invertebrates are collected by the rat-catcher trawl and net bag as a result of the relatively high tow speed. The nature of the invertebrate catch may also be affected by the abundance of fish caught; large numbers of fish may crush the more fragile invertebrate taxa and/or exclude them from the net. Regardless of the cause, high variability in community structure estimates poses serious issues for the use of such data in a monitoring programme. Because of the problems of selectivity and variability associated with the use of rat-catcher and net bag gear, the beam trawl represents a more accurate (i.e., more representative) and precise (i.e., less variable estimates of community structure) tool for the monitoring of deep-sea benthic communities if used in “tagged on” surveys.

Table 8: Results of SIMPER analyses showing taxa accounting for <u><</u>80% of dissimilarity in community structure estimates between beam trawl and rat-catcher trawl, and between beam trawl and net bag. [Av. Bio = Average biomass (kg 0.1 km^{-2}), Av. Diss = Average dissimilarity, Diss/SD = dissimilarity/Standard Deviation, Contrib% = % contribution to overall dissimilarity, Cum% = % cumulative dissimilarity.]

Beam trawl vs rat-catcher trawl	Average No. Beam trawl	Rat-catcher	Av.Diss	Diss/SD	Contrib%	Cum.%
Gastropoda	61.8	0.0	7.3	3.8	7.5	7.5
Decapoda	36.2	0.0	6.6	2.9	6.8	14.2
Echinoidea	129.2	<0.1	6.6	2.0	6.7	20.9
Holothuroidea	75.7	0.2	6.2	1.7	6.3	27.3
Asteroidea	32.0	<0.1	4.9	1.9	5.0	32.3
Ophiuroidea	17.2	<0.1	4.9	2.4	5.0	37.3
Polychaeta	23.0	<0.1	4.6	1.5	4.7	42.0
Sipuncula	15.4	0.0	4.3	1.9	4.4	46.5
Galatheidae	17.3	<0.1	4.1	1.1	4.2	50.6
Hydrozoa	13.9	0.0	4.0	1.5	4.1	54.7
Porifera	20.2	3.6	3.9	1.5	4.0	58.7
Paguroidea	14.3	<0.1	3.8	1.6	3.9	62.6
Isopoda	13.0	0.0	3.8	1.5	3.9	66.5
Brisingida	97.2	5.5	3.5	0.8	3.6	70.1
Pleocyemata	16.0	<0.1	3.4	1.4	3.5	73.6
Bivalvia	11.3	0.0	3.3	1.3	3.4	77.0
Anthozoa	17.6	<0.1	3.3	1.1	3.4	80.4

Beam trawl vs net bags	Av. Bio Beam trawl	Av. Bio Net bags	Av.Diss	Diss/SD	Contrib%	Cum.%
Gastropoda	61.8	<0.1	6.3	4.2	6.5	6.5
Decapoda	36.2	<0.1	5.7	3.2	5.9	12.5
Echinoidea	129.2	<0.1	5.7	2.0	5.9	18.4
Holothuroidea	75.7	0.6	5.3	1.8	5.5	23.8
Asteroidea	32.0	0.0	4.3	2.0	4.5	28.3
Ophiuroidea	17.2	0.1	4.2	2.6	4.4	32.7
Polychaeta	23.0	0.0	4.0	1.5	4.1	36.9
Sipuncula	15.4	0.0	3.8	2.0	4.0	40.8
Galatheidae	17.3	0.0	3.5	1.2	3.7	44.5
Hydrozoa	13.9	<0.1	3.5	1.5	3.6	48.1
Porifera	20.2	0.7	3.4	1.7	3.5	51.6
Isopoda	13.0	0.0	3.3	1.6	3.5	55.1
Paguroidea	14.3	0.3	3.3	1.7	3.4	58.4
Brisingida	97.2	3.9	3.1	0.8	3.2	61.6
Pleocyemata	16.0	0.0	3.0	1.4	3.1	64.8
Anthozoa	17.6	<0.1	2.9	1.1	3.0	67.8
Bivalvia	11.3	0.0	2.9	1.3	3.0	70.8
Scleratinia	12.7	0.0	2.7	1.0	2.8	73.6
Pseudostichopus mollis	10.7	1.9	1.8	0.9	1.9	75.5
Echinothuriidae and Phormosomatidae	0.0	4.6	1.8	1.1	1.8	77.3
Pennatulacea	7.7	0.1	1.8	0.9	1.8	79.1
Nudibranchia	6.8	<0.1	1.4	0.6	1.5	80.6

Camera data comparisons

In total, thirteen sites across the four depth strata were surveyed using the rat-catcher trawl with attached headline cameras. A number of the video records of a tow were hazy, or the water clarity was poor, and hence, four stations were selected where the image quality of both video and still cameras was deemed to be adequate for comparison. These four stations included one from each stratum. These were stations 66 (344 m depth), 50 (518 m depth), 32 (758 m depth) and 63 (956 m depth). The taxa identified in the images are listed in Table 9.

Table 9: Summary of taxa recorded in trawl-mounted video and still image camera images, Y = present.

Taxon	Common name	Video	Still
Actiniaria	Anemones	Y	
Zoantharia	Anemones		Y
Epizoanthus	Zoanthid anemone	Y	Y
Ceriantheria	Tube anemone		Y
Hydrozoa	Hydroid/octocoral		Y
Flabellum	Cup coral		Y
Radicipes	Golden coral	Y	Y
Pennatulacea	Sea pen	Y	
Pennatulacea sp. A	Sea pen		Y
Pennatulacea sp. B	Sea pen		Y
Scyphozoa	Medusae		Y
Decapoda Natantia	Shrimps	Y	Y
Campylonotus rathbunae	Sabre prawn		Y
Paguridae	Hermit crab		Y
Majidae	Masking crab		Y
Gastropoda	Sea snails		Y
Asteroidea	Seastars	Y	Y
Brisingidae	Armless seastar	Y	
Crossaster	Sunstar		Y
Echinoidea	Sea urchin	Y	
Phormosomatidae	Tam O'Shanters	Y	
Cidaridae	Cidaroid urchin		Y
Spatangidae	Heart urchin		Y
Laetmogonidae	Sea cucumber		Y

The smaller number of taxa observed in the video relative to still imagery is obvious, despite the video footage covering a greater width, and the whole length of the tow. The video also only enabled very large animals, or taxa with a distinctive shape, to be readily identified. Table 10 summarises the taxonomic level of the groups that were recorded. Although the still camera data had two taxa at the Class level, these were Gastropoda and Hydrozoa, which are typically small-bodied, and could not be recognised at all in the video footage.

Table 10: Summary of number of taxa (proportion of total) identified at different taxonomic levels in data from trawl-mounted video and still image cameras.

Taxonomic level	Video	Still
Class		2 (0.10)
Order	5 (0.55)	3 (0.16)
Family	2 (0.23)	7 (0.37)
Genus	1 (0.11)	3 (0.16)
Species	1 (0.11)	4 (0.21)
Total	9	19

The majority of video identifications were at the Order level, whereas most megafaunal taxa in still images could be identified at the Family level, with a relatively high proportion to species level.
Counts were made of all taxa observed in the images or footage. Figure 14 again highlights that the number of taxa observed was greater in still images but also that the relative proportions of taxa were very different.

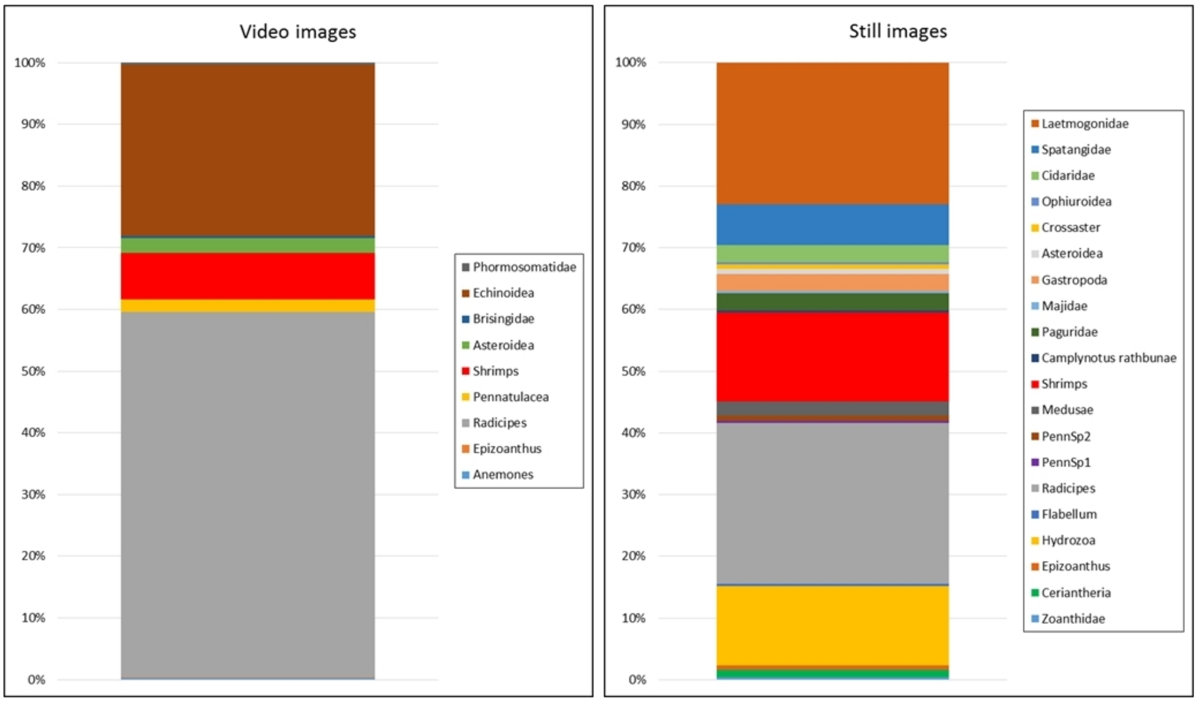

Figure 14: Comparison of the benthic invertebrate taxa observed in video (left) and still (right) imagery from four trawl-mounted camera tows selected from TAN1208. Data were recorded as counts of individuals per tow and are shown as percentages of the total number of individuals recorded summed over the four tows.

The majority of benthic invertebrate animals observed in the video imagery were *Radicipes* sp., a distinctive spiral whip-like coral that is frequently observed on soft sediment seafloor, and echinoids (Figure 14). In the still image data, *Radicipes* sp. and echinoids (discriminated as Spatangidae and Cidaridae) were, again, the most frequently recorded taxa. However, smaller taxa were either less well-represented (e.g., shrimps) or completely absent (e.g., Laetmogonidae sea cucumbers, gastropods, and hydrozoans) in the video data despite being common in the still image data.

Results are presented for the combined tows, but the general pattern was consistent among the different depths, with more, and smaller-sized, taxa identified in still images compared to video. However, the relative abundances of various taxa also changed with depth and location (Figure 15). The video records at the two deeper sites (mainly shrimps) were relatively sparse in comparison with the large numbers of *Radicipes* seen at the 500 m site. Similarly, the 340 m station was strongly dominated by one taxon, the echinoids (urchins), which were readily identifiable by their shape. The relative abundances of taxa were also reflected in the still images. Although fewer animals were observed in the still images (which is expected as there is no scaling up of these data to represent swept area) the composition of the most abundant fauna was similar to that recorded in the video. The 340 m and 500 m stations recorded mainly the urchin family Spatangidae, and *Radicipes*. Hence, while the size and distinctive spiral form of *Radicipes* enabled consistent identification even in the video images, the higher resolution still images were able to refine the identification of the urchins from order to family. Overall, the results of the comparison of imagery data indicate that it is preferable to identify taxa using still images, rather than video, if headline cameras are to be used for monitoring benthic communities during fishery surveys.

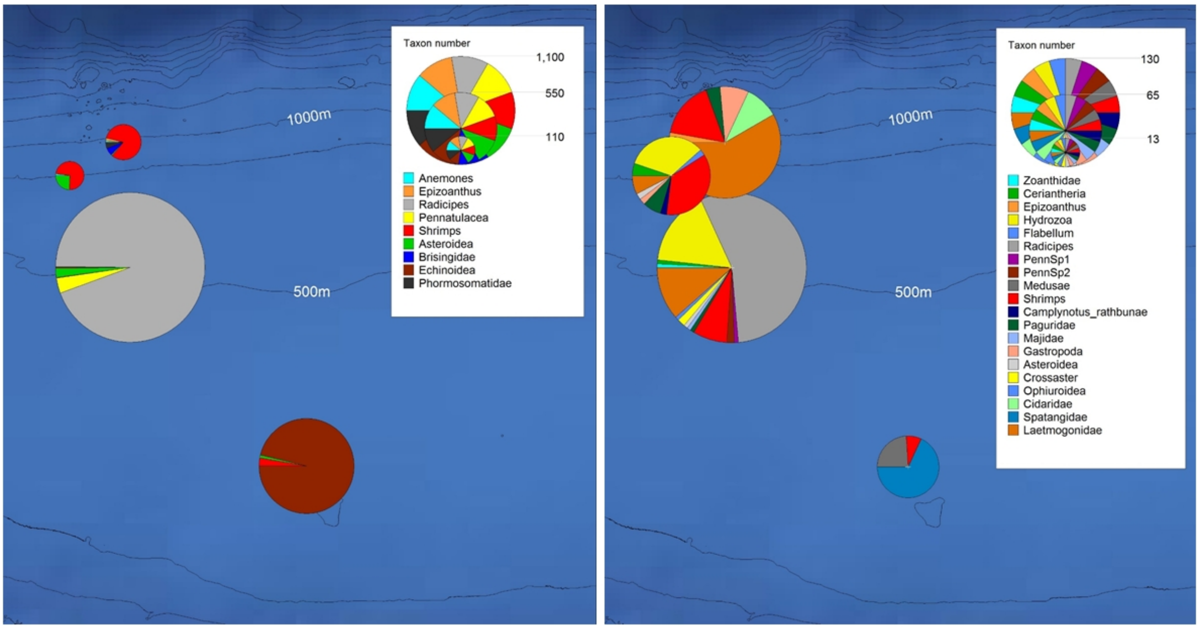

Figure 15: Spatial distribution of numbers of individuals per taxon recorded in tows with the video camera (left panel) and still camera (right panel). Note that absolute numbers are not comparable between camera types, but are comparable within camera type.

4.1.1 Costing a "tagged on" monitoring programme

Our findings in previous sections of this study indicate that dedicated gear deployments would be needed for any effective benthic monitoring programme, and that a minimum set of trained staff for these deployments would also be required. Thus, even if a fisheries trawl could carry cameras or other modifications to enable collection of benthic samples, any sampling using the modified gear would have to be in addition to sampling for the planned core purpose of the survey, with consequent extension of the overall duration of the voyage. In addition, it is probable that extra science and technical staff with benthic invertebrate identification and handling skills would be needed to collect and process samples for the benthic survey.

We conclude that the unit cost of benthic sampling in a "tagged-on" programme, in terms of person-hours and ship time at sea, would be the same as outlined for a dedicated 'optimal' monitoring programme (see Table 6). Economies might be made in some areas, however, including: voyage mobilisation costs; transit times to and from the survey area, and potentially staffing costs in cases where individual scientists or technicians have skills in both fisheries and benthic invertebrate sampling methods. Potential savings in these areas would have to be weighed carefully against the extra costs of keeping any dedicated fisheries science staff, who are not involved with the benthic sampling, at sea for longer periods.

5 SUMMARY AND CONCLUSIONS

Any monitoring programme that uses consistent, repeatable, methods to collect data on marine ecosystems at regular intervals over an extended period is likely to generate valuable insights into ecosystem dynamics. To address specific questions, however, such as how benthic habitats are affected by disturbance from fishing over time, a monitoring programme would need to be very carefully specified. Below we summarise the findings and conclusions of the present study to design a programme to monitor trends in deep-water benthic communities in the New Zealand EEZ.

5.1 Designing a monitoring programme

5.1.1 Available data

Data about deep-sea benthic habitats and communities around New Zealand are patchy and incomplete. Thus, there are few deep-sea areas within the EEZ for which we have sufficient data, collected in a consistent fashion, to consider embarking with confidence on a programme to monitor trends over time. In the best-sampled areas of the EEZ, notably Chatham Rise, Challenger Plateau, and the Northland continental shelf – i.e. areas surveyed under Ocean Survey 20/20 – we can make some estimation of the faunal composition and spatial extent of habitats, and the scales over which they vary. Such information provides essential context against which to assess any monitoring of trends in the status of benthic habitats and communities. Thus, prior to any long-term monitoring programme starting in regions that have yet to be the subject of an OS 20/20-style survey, a prerequisite is that a comprehensive study of benthic habitats and communities should be completed. This might start with a desk-top study to ascertain the quantity and quality of existing data available for the area in question, and progress to a full base-line sampling survey if these data did not provide sufficient information on which to design a long-term monitoring programme.

5.1.2 Scales of variability

Even the OS 20/20 surveys provide only a limited, single time-point, characterisation of an area and it is clear from developing knowledge of benthic distributions on Chatham Rise in particular (Section 3.1.4), that entire habitats and communities can be missed in such surveys. Given the spatial variability in benthic community structure seen within such OS 20/20 surveys, there is also uncertainty in any estimates of the statistical power of subsequent monitoring to detect ecologically significant change. Analyses of data from existing OS 20/20 surveys (Chatham Rise, Challenger Plateau, Northland shelf and Bay of Islands) indicate that characteristic scales of variability for conventional biodiversity metrics cannot be assumed to be comparable between regions of the EEZ. This result also suggests that dedicated benthic surveys of similar intensity to the OS 20/20 surveys would be required before sensible benthic monitoring strategies for new regions could be developed.

5.1.3 Habitat and community attributes to monitor

In light of recent international assessments and the data and sampling methods available in New Zealand at present, there are three main attributes of benthic habitats and communities that we consider to be both important and practical to monitor: biodiversity; biological traits, and ecosystem function, with the concept of ecological integrity providing an overarching framework.

Six indices of biodiversity have been identified as being potentially useful for assessing variations in benthic communities:

1. number of taxa (species richness, S);
2. Pielou's evenness (J');
3. Simpson's diversity (1-λ);

4. number of taxa rare in abundance (SRA);
5. number of infrequently occurring taxa (SRF); and
6. beta-diversity calculated from abundance data.

Analysis of the biological traits of benthic organisms is useful for detecting the effects of fishing activities, such as dredging and trawling, because some specific biological traits are more sensitive to physical disturbance. A practical subset of these biological traits has been identified (i.e., scavenging, sedentary etc.) and matched to attributes (i.e., feeding, mobility etc.) and their potential response to disturbance (i.e., positive, strongly negative etc.). Most of the information required to undertake a biological traits analysis can be assembled from seabed video or, to a lesser extent, epibenthic sled and beam trawl data, in conjunction with reference to the literature, but fisheries trawl bycatch data may be less useful in this context because the organisms cannot be observed in their un-damaged living state.

For assessing ecosystem function, eight functional groups can be defined which have different susceptibility and recovery characteristics. Five of these functional groups can be sampled adequately by beam trawl and towed video sampling, but the other three demand finer-resolution sampling by box- or multi-corers, or by still camera imagery.

Measures of ecological integrity are generally separated into those representing categories of nativeness, pristineness, diversity and resilience, and 7–10 indicators can be measured in the deep sea for each of these categories.

5.1.4 Optimal survey design

Methods used for a benthic monitoring programme should ideally be non-destructive or minimally destructive, otherwise the monitoring becomes part of one of the factors for which effects are being investigated (i.e., physical disturbance). The most useful target faunal component for cost-effectiveness and relevance to ecological effects at the scale of deep-sea fisheries is likely to be mega-epifauna. Together, these considerations suggest camera transects as the single most appropriate method for monitoring deep-water benthic communities. If analyses currently in progress (MPI project ZBD2012-03, OS 20/20 Chatham Benthos) indicate that other components of the benthic system, such as macro- or meio-infauna, or sediment properties are also useful indicators of fisheries-related change, sediment corers (multicorer) could be deployed in combination with camera transects.

Benthic communities consist of organisms that are either sessile or have limited mobility in their adult phase. These life habits are fundamentally different to those of demersal and pelagic fish, which are mobile over large areas. Because of the likelihood of high sample variance that would result from sampling the large-scale strata of a conventional stratified-random survey, as used in fisheries stock assessment research, such an approach would be less appropriate for monitoring change in benthic communities than would a ‘repeated measures’ design in which sites are initially assigned randomly within selected strata but then revisited at each subsequent time-point. Sampling strata in a repeated measures survey design would be selected on the basis of: (a) known gradients of fishing disturbance; (b) known distributions of benthic communities or habitats, and (c) the locations of existing benthic protection areas (where they exist in a monitoring region). Individual monitoring sites within strata would probably be best defined as patches of seabed of a spatial scale suitable to encompass multiple, non-overlapping, replicate deployments of the sampling gear. For towed camera systems in use at present, this would be no larger than about 10 × 10 km.

5.1.5 Cost of an optimal monitoring programme

The cost of a monitoring programme will be dependent on high-level decisions associated with its location, purpose, extent, and the magnitude of change to be detected. Therefore, until such decisions have been made through consultation between fisheries managers and research scientists, only an indicative costing can be provided (resource and time expenditure, rather than dollars). To undertake a survey with 26 strata and ten camera transects per stratum (e.g., one based on surveying all hoki and middle-depths research trawl strata on Chatham Rise) would require commitment of a towed camera, and a suitable vessel and staff for about 32 days at each time-step of the monitoring survey. Following the survey, sample processing, sample curation and archiving, data analysis, and report writing would require in the region of 120–270 hours of science and technical staff time per stratum (i.e., about 12–27 hours per sample), depending on the level of analysis required (e.g. analysis of video, still images, or both). Stratified-random and repeated measures-type surveys would entail exactly the same resource cost per sample but the repeated-measures approach would yield greater statistical power to detect change. Overall costs for either type of survey could be tailored to the available budget by reducing: (a) the number of strata monitored; (b) the number of survey sites within each stratum; (c) the number of faunal components monitored; and (d) the frequency at which the survey is repeated (e.g., annual versus biennial).

5.2 Feasibility of using existing surveys

5.2.1 Planned surveys

There are a number of fisheries surveys that could provide the opportunity for a “tagged on” benthic sampling component. These surveys include those on the Chatham Rise and Campbell Plateau, off the West Coast South Island, and in the Sub-Antarctic region of the New Zealand EEZ. The planned fishery surveys encompass water depths from 200 m to1300 m. However, there is uncertainty around the frequency or even long-term continuation of some of these surveys. Such uncertainty is not conducive to establishing and maintaining a monitoring programme for benthic communities.

5.2.2 Issues with a “tagged on” monitoring programme

Attaching benthic sampling gear to trawl gear used in existing fishery surveys or commercial trawling during their core operations would not be acceptable because of the potential effects of the added components (e.g., lights, cameras, net bags) on catchability. Thus, in most cases, surveys would have to be extended to allow additional tows explicitly for benthic sampling. In fisheries research trawl surveys, the numbers of replicate tows within strata are planned to reduce variation to be within pre-defined bounds. Benthic community structure does not necessarily co-vary with fisheries strata and thus there are likely to be mismatches between the requirements of fisheries and benthic monitoring in terms of both the spatial extent of appropriate strata and the number of replicate samples required to keep survey variation within required bounds. Thus Fisheries Management Areas and other fisheries-oriented spatial survey schemes may not be appropriate for a benthic monitoring programme, and both the extent of strata and the density of sampling within them should be relevant to known benthic distributions (see also Section 3.1.4 – Survey Design and Extent).

We did not observe any significant differences in estimates of total biomass or number of benthic invertebrate taxa sampled by three different direct sampling gear types that are most likely to be considered for use in a “tagged on” survey. However, although the biomass and number of taxa caught by the different gear types did not vary significantly, the identity of these taxa did. Most taxa were better sampled by the beam trawl than by the rat-catcher trawl and net bags. Neither the rat-catcher trawl nor net bag can be relied upon to provide representative samples of the benthic community, because they provide samples biased towards the largest taxa.

In addition to the size selectivity issue associated with the rat-catcher trawl and net bag, we also observed greater variability in estimates of community structure derived from these gear types relative to the beam trawl. High variability between samples lowers the power of statistical analyses, thereby lowering our ability to detect ecologically meaningful change. Because of the problems of selectivity and variability associated with the use of rat-catcher and net bag gear, the beam trawl represents a more accurate (i.e., more representative) and precise (i.e., less variable estimates of community structure) tool for the monitoring of deep-sea benthic communities in "tagged-on" surveys.

The analysis of video and still image data from a trawl headline camera taken during a fishery survey indicated that, despite the video footage providing continuous data and imaging a greater width of the seafloor, more taxa were observed in the still images. Because of the high tow speed and relatively low image resolution of the video, only very large animals, or taxa with a distinctive shape, could be readily identified. The majority of video identifications were at a high taxonomic level (e.g., Order), whereas most megafaunal taxa in the still images could be identified to lower taxonomic levels (e.g., Family to species). The results of this comparison of data indicate the worth of identifying taxa using still images if headline cameras are to be used for monitoring benthic communities during fishery surveys.

5.2.3 Cost of a "tagged on" monitoring programme

With present-day technology, the potential for cost-savings to be made by "tagging on" a benthic monitoring component to existing fisheries research trawl surveys is limited because the following would still be needed: (a) dedicated benthic sampling gear and deployments, which would be independent of the fishing survey; (b) dedicated benthic sampling staff, and (c) a science research vessel with capability to deploy deep-sea cameras and other benthic gear types. Thus, savings by comparison with a dedicated benthic sampling voyage would be made only with respect to: (i) mobilisation costs; (ii) transit times, and (iii) potentially, a proportion of staffing costs if some science staff have both fisheries and benthic invertebrate sampling skills.

6 MANAGEMENT IMPLICATIONS

To realise the ambitions and obligations of New Zealand to instigate responsible, ecosystem-based management of commercial fisheries, it would be advantageous to instigate monitoring programmes capable of detecting temporal trends in marine ecosystems. This applies to all aspects of the marine system but, because benthic habitats and communities are directly impacted by seabed trawling, it is of primary importance to understand how these systems interact with pelagic and demersal ecosystem components and how they are affected by fishing disturbances.

Instigating a benthic monitoring programme that would have the statistical power reliably to detect trends, even for a relatively small part of the New Zealand EEZ, would require a substantial commitment of funds over many years. Costs per survey are likely to be comparable to those of existing fisheries research trawl surveys, although the actual commitment would vary considerably depending on the scale of the survey. In principle, it is better to start a well-designed programme on a smaller scale and maintain it consistently over the long term than to run a more ambitious programme that, because of its high costs, might not be funded beyond the first one or two years.

Detection of temporal trends in benthic habitats and communities, and particularly of changes related to fisheries disturbance, is complicated by the lack of control areas in which there is no anthropogenic disturbance. A wider goal of environmental management in this context would be to integrate a benthic monitoring programme with the gathering of data from existing protected areas (at present, only the Benthic Protection Areas – of which there are only two in fishing depths on Chatham Rise) and, ideally, to establish control, no-fishing, areas within the current fisheries footprint.

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8 REFERENCES

Alden, R.W.; Weisberg, S.B.; Ranasinghe, J.A.; Dauer, D.M. (1997). Optimizing temporal sampling strategies for benthic environmental monitoring programs. *Marine Pollution Bulletin 34(11)*: 913–922. <http://dx.doi.org/10.1016/s0025-326x(97)00049-0>

Anderson, C.N.K.; Hsieh, C.H.; Sandin, S.A.; Hewitt, R.; Hollowed, A.; Beddington, J.; May, R.M.; Sugihara, G. (2008a). Why fishing magnifies fluctuations in fish abundance. *Nature 452(7189)*: 835–839. <http://dx.doi.org/10.1038/nature06851>

Anderson, M.J. (2002). Structures for establishing a database for marine monitoring. *DOC Science Internal Series. Department of Conservation 58*. 21 p.

Anderson, M.J.; Gorley, R.N.; Clarke, K.R. (2008b). PERMANOVA+ for PRIMER: a guide to Software and Statistical Methods. Plymouth, UK, PRIMER-E. p.

Anderson, O.F.; Clark, M.R. (2003). Analysis of bycatch in the fishery for orange roughy, Hoplostethus atlanticus, on the South Tasman Rise. *Marine and Freshwater Research 54(5)*: 643–652.

Andrew, N.L.; Mapstone, B.D. (1987). Sampling and the description of spatial pattern in marine ecology. *Oceanography and Marine Biology 25*: 39–90.

Armonies, W. (2000). On the spatial scale needed for benthos community monitoring in the coastal North Sea. *Journal of Sea Research 43(2)*: 121–133. <http://dx.doi.org/http://dx.doi.org/10.1016/S1385-1101(00)00008-3>

Bartel, A. (2000). Analysis of landscape pattern: towards a 'top down' indicator for evaluation of landuse. *Ecological Modelling 130(1–3)*: 87–94. <http://dx.doi.org/10.1016/s0304-3800(00)00214-3>

Bax, N.; Kloser, R.; Williams, A.; Gowlett-Holmes, K.; Ryan, T. (1999). Seafloor habitat definition for spatial management in fisheries: a case study on the continental shelf of southeast Australia. *Oceanologica Acta 22(6)*: 705–720. <http://dx.doi.org/10.1016/s0399-1784(00)88961-6>

Black, J.; Wood, R. (2014). Analysis of New Zealand's trawl grounds for key Middle Depths and Deepwater Tier 1 fisheries *New Zealand Aquatic Environment and Biodiversity Report No. 122*. 35 p.

Blake, J.A.; Maciolek, N.J.; Ota, A.Y.; Williams, I.P. (2009). Long-term benthic infaunal monitoring at a deep-ocean dredged material disposal site off Northern California. *Deep-Sea Research Part Ii-Topical Studies in Oceanography 56(19–20)*: 1775–1803. <http://dx.doi.org/10.1016/j.dsr2.2009.05.021>

Borja, A.; German Rodriguez, J.; Black, K.; Bodoy, A.; Emblow, C.; Fernandes, T.F.; Forte, J.; Karakassis, I.; Muxika, I.; Nickell, T.D.; Papageorgiou, N.; Pranovi, F.; Sevastou, K.; Tomassetti, P.; Angel, D. (2009). Assessing the suitability of a range of benthic indices in the evaluation of environmental impact of fin and shellfish aquaculture located in sites across Europe. *Aquaculture 293(3–4)*: 231–240. <http://dx.doi.org/10.1016/j.aquaculture.2009.04.037>

Bostrom, C.; Jackson, E.L.; Simenstad, C.A. (2006). Seagrass landscapes and their effects on associated fauna: A review. *Estuarine Coastal and Shelf Science 68(3–4)*: 383–403. <http://dx.doi.org/10.1016/j.ecss.2006.01.026>

Bowden, D.; Jones, D.O.B. (In press). Towed cameras in deep sea research. *In*: Clark, M.R.; Rowden, A.A.; Consalvey, M. (eds). Biological Sampling in The Deep Sea, pp. Wiley & Sons, 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 No. 65*. 46 p.

Bowden, D.A.; Chin, C.; Davey, N.; Fenwick, M.; George, S.; Gerring, P.; Hart, A.; Leduc, S.; Mills, S. (2013a). Factual voyage report: TAN1306 Ocean Survey 20/20 – Chatham Rise Benthos. Ministry for Primary Industries. 35 p. (Unpublished report held by Ministry for Primary Industries, Wellington.)

Bowden, D.A.; Compton, T.J.; Snelder, T.; Hewitt, J. (2011). Evaluation of the New Zealand Marine Environment Classifications using Ocean Survey 20/20 data from Chatham Rise and Challenger Plateau. *New Zealand Aquatic Environment and Biodiversity Report No. 77*. 27 p.

Bowden, D.A.; Hewitt, J. (2012). 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 No. 91*. 34 p.

Bowden, D.A.; Rowden, A.A.; Thurber, A.R.; Baco, A.R.; Levin, L.A.; Smith, C.R. (2013b). Cold Seep Epifaunal Communities on the Hikurangi Margin, New Zealand: Composition, Succession, and Vulnerability to Human Activities. *PLoS ONE 8(10)*: e76869. <http://dx.doi.org/10.1371/journal.pone.0076869>

Bremner, J.; Rogers, S.I.; Frid, C.L.J. (2003). Assessing functional diversity in marine benthic ecosystems: a comparison of approaches. *Marine Ecology-Progress Series 254*: 11–25.

Bremner, J.; Rogers, S.I.; Frid, C.L.J. (2006a). Matching biological traits to environmental conditions in marine benthic ecosystems. *Journal of Marine Systems 60(3–4)*: 302–316. <http://dx.doi.org/10.1016/j.jmarsys.2006.02.004>

Bremner, J.; Rogers, S.I.; Frid, C.L.J. (2006b). Methods for describing ecological functioning of marine benthic assemblages using biological traits analysis (BTA). *Ecological Indicators 6(3)*: 609–622. <http://dx.doi.org/10.1016/j.ecolind.2005.08.026>

Burkhard, B.; Opitz, S.; Lenhart, H.; Ahrendt, K.; Garthe, S.; Mendel, B.; Windhorst, W. (2011). Ecosystem based modeling and indication of ecological integrity in the German North Sea-Case study offshore wind parks. *Ecological Indicators 11(1)*: 168–174. <http://dx.doi.org/10.1016/j.ecolind.2009.07.004>

Carney, R.S. (2005). Zonation of deep biota on continental margins. *Oceanography and Marine Biology - an Annual Review, Vol.*: 211–278.

Clark, M. (1999). Fisheries for orange roughy (*Hoplostethus atlanticus*) on seamounts in New Zealand. *Oceanologica Acta 22(6)*: 593–602.

Clark, M.R.; Anderson, O.F.; Francis, R.; Tracey, D.M. (2000). The effects of commercial exploitation on orange roughy (*Hoplostethus atlanticus*) from the continental slope of the Chatham Rise, New Zealand, from 1979 to 1997. *Fisheries Research 45(3)*: 217–238.

Clark, M.R.; Roberts, C.D. (2008). Fish and invertebrate biodiversity on the Norfolk Ridge and Lord Howe Rise, Tasman Sea (NORFANZ voyage, 2003). *New Zealand Aquatic Environment and Biodiversity Report No. 28*. 131 p.

Clark, M.R.; Rowden, A.A. (2009). Effect of deepwater trawling on the macro-invertebrate assemblages of seamounts on the Chatham Rise, New Zealand. *Deep-Sea Research Part I-Oceanographic Research Papers 56(9)*: 1540–1554. <http://dx.doi.org/10.1016/j.dsr.2009.04.015>

Clark, M.R.; Rowden, A.A.; Wright, I.; Consalvey, M. (2010). Graveyard Seamounts. *Oceanography 23(1)*: 146–147.

Clarke, K.R.; Gorley, R.N. (2006). PRIMER v6: User manual/tutorial. PRIMER-E Plymouth UK. 192 p.

Clarke, K.R.; Somerfield, P.J.; Chapman, M.G. (2006). On resemblance measures for ecological studies, including taxonomic dissimilarities and a zero-adjusted Bray–Curtis coefficient for denuded assemblages. *Journal of Experimental Marine Biology and Ecology 330(1)*: 55–80. <http://dx.doi.org/http://dx.doi.org/10.1016/j.jembe.2005.12.017>

Clarke, K.R.; Warwick, R.M. (2001). Change in marine communities: an approach to statistical analysis. 2nd. PRIMER-E Ltd., Plymouth, U.K. 172 p.

Cohen, J. (1988). Statistical power analysis for behavioural sciences. Lawrence Erlbaum Associates, Inc., Hillsdale, New Jersey. 567 p.

Collie, J.S.; Escanero, G.A.; Valentine, P.C. (2000). Photographic evaluation of the impacts of bottom fishing on benthic epifauna. *ICES Journal of Marine Science 57(4)*: 987–1001.

Compton, T.J.; Bowden, D.A.; Pitcher, R.C.; Hewitt, J.E.; Ellis, N. (2012). Biophysical patterns in benthic assemblage composition across contrasting continental margins off New Zealand. *Journal of Biogeography 40(1)*: 75–89. <http://dx.doi.org/10.1111/j.1365-2699.2012.02761.x>

Cranmer, G.J.; Dyer, M.F.; Fry, P.D. (1984). Further results from headline camera surveys of the North Sea. *Journal of the Marine Biological Association of the United Kingdom 64*: 335–342.

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.

de Juan, S.; Cartes, J.E.; Demestre, M. (2007). Effects of commercial trawling activities in the diet of the flat fish Citharus linguatula (Osteichthyes : Pleuronectiformes) and the starfish Astropecten irregularis (Echinodermata : Asteroidea). *Journal of Experimental Marine Biology and Ecology 349(1)*: 152–169. <http://dx.doi.org/10.1016/j.jembe.2007.05.003>

de Juan, S.; Demestre, M. (2012). A Trawl Disturbance Indicator to quantify large scale fishing impact on benthic ecosystems. *Ecological Indicators 18*: 183–190. <http://dx.doi.org/10.1016/j.ecolind.2011.11.020>

de Juan, S.; Demestre, M.; Thrush, S. (2009). Defining ecological indicators of trawling disturbance when everywhere that can be fished is fished: A Mediterranean case study. *Marine Policy 33(3)*: 472–478. <http://dx.doi.org/10.1016/j.marpol.2008.11.005>

de Juan, S.; Hewitt, J.E.; Thrush, S.F.; Freeman, D. (In press). Standardizing the assessment of functional integrity in benthic ecosystems. *Journal of Sea Research*.

Diaz, R.J.; Solan, M.; Valente, R.M. (2004). A review of approaches for classifying benthic habitats and evaluating habitat quality. *Journal of Environmental Management 73(3)*: 165–181. <http://dx.doi.org/10.1016/j.jenvman.2004.06.004>

Dyer, M.F.; Fry, W.G.; Fry, P.D.; Cranmer, G.J. (1982). A series of North Sea benthos surveys with trawl and headline camera. *Journal of the Marine Biological Association of the United Kingdom 62:*297–31.

Eleftheriou, A.; McIntyre, A. (eds). (2005). Methods for the study of marine benthos. Blackwell Publishing, Ames IA. 440 p.

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. <http://dx.doi.org/10.1023/a:1004172622751>

Fairweather, P.G. (1991). Statistical power and design requirements for environmental monitoring. *Australian Journal of Marine and Freshwater Research 42*: 555–567.

Floerl, O.; Hewitt, J.; Bowden, D. (2012). Chatham-Challenger Ocean Survey 20/20 Post-Voyage Analyses: Objective 9 - Patterns in species composition. *New Zealand Aquatic Environment and Biodiversity Report No. 97*. 40 p.

Francis, M.P.; Hurst, R.J.; McArdle, B.H.; Bagley, N.W.; Anderson, O.F. (2002). New Zealand demersal fish assemblages. *Environmental Biology of Fishes 65*: 215–239.

Garrabou, J.; Riera, J.; Zabala, M. (1998). Landscape pattern indices applied to Mediterranean subtidal rocky benthic communities. *Landscape Ecology 13(4)*: 225–247.

Glover, A.G.; Smith, C.R. (2003). The deep-sea floor ecosystem: current status and prospects of anthropogenic change by the year 2025. *Environmental Conservation 30(3)*: 219–241.

Gordon, D.P.; Beaumont, J.; MacDiarmid, A.; Robertson, D.A.; Ahyong, S.T. (2010). Marine Biodiversity of Aotearoa New Zealand. *PLoS ONE 5(8)*. <http://dx.doi.org/10.1371/journal.pone.0010905>

Grassle, J.F.; Sanders, H.L.; Hessler, R.R.; Rowe, G.T.; McLellan, T. (1975). Pattern and zonation: a study of the bathyal megafauna using the research submersible *Alvin*. *Deep Sea Research 22*: 475–481.

Gray, J.S.; Bjorgesaeter, A.; Ugland, K.I. (2005). The impact of rare species on natural assemblages. *Journal of Animal Ecology 74(6)*: 1131–1139. <http://dx.doi.org/10.1111/j.1365-2656.2005.01011.x>

Hanchet, S.; Mitchell, J.; Bowden, D.; Clark, M.; Hall, J.; O'Driscoll, R.; Pinkerton, M.; Robertson, D. (2008a). Ocean Survey 20/20 New Zealand IPYCAML Final Voyage Report. NIWA

Client Report. National Institute of Water and Atmospheric Research Ltd. WLG2008-74. 199 p.
Hanchet, S.M.; Stevenson, M.L.; Jones, C.; Marriott, P.M.; McMillan, P.J.; O'Driscoll, R.L.; Stevens, D.; Stewart, A.L.; Wood, B.A. (2008b). Biomass estimates and size distributions of demersal finfish on the Ross Sea shelf and slope from the New Zealand IPY-CAML survey, February-March 2008. *CCAMLR WG-FSA-08/31*. 23 p.
Hewitt, J.; De Juan, S.; Lohrer, A.; Townsend, M.; D'Archino, R. (2014). Functional traits as indicators of ecological integrity. NIWA Client Report No. HAM2014-001/DOC13213.
Hewitt, J.; Julian, K.; Bone, E.K. (2011a). Chatham–Challenger Ocean Survey 20/20 Post-Voyage Analyses: Biotic habitats and their sensitivity to physical disturbance. *New Zealand Aquatic Environment and Biodiversity Report No. 81*. 36 p.
Hewitt, J.; Lundquist, C.J.; Bowden, D.A. (2011b). Chatham-Challenger Ocean Survey 20/20 Post Voyage Analyses: Diversity Metrics. *New Zealand Aquatic Environment and Biodiversity Report No. 83*. 62 p.
Hewitt, J.E.; Anderson, M.J.; Thrush, S.F. (2005). Assessing and monitoring ecological community health in marine systems. *Ecological Applications 15(3)*: 942–953.
Hewitt, J.E.; Thrush, S.F.; Dayton, P.D. (2008). Habitat variation, species diversity and ecological functioning in a marine system. *Journal of Experimental Marine Biology and Ecology 366(1–2)*: 116–122. <http://dx.doi.org/10.1016/j.jembe.2008.07.016>
Hill, P. (2009). Designing a deep-towed camera vehicle using single conductor cable. *Sea Technology 50(12)*: 49–51.
Jones, J.B. (1992). Environmental impact of trawling on the seabed: a review. *New Zealand Journal of Marine and Freshwater Research 26(1)*: 59–67.
Karr, J.R.; Thomas, T. (1996). Economics, ecology, and environmental quality. *Ecological Applications 6(1)*: 31–32. <http://dx.doi.org/10.2307/2269549>
Kipson, S.; Fourt, M.; Teixido, N.; Cebrian, E.; Casas, E.; Ballesteros, E.; Zabala, M.; Garrabou, J. (2011). Rapid Biodiversity Assessment and Monitoring Method for Highly Diverse Benthic Communities: A Case Study of Mediterranean Coralligenous Outcrops. *PLoS ONE 6(11)*. <http://dx.doi.org/10.1371/journal.pone.0027103>
Kudrass, H.-R.; von Rad, U. (1984). Underwater television and photpgraphy observations, side-scan sonar and acoustic reflectivity measurements of phosphorite-rich areas of the Chatham Rise (New Zealand). *Geologisches Jahrbuch D65*: 69–89.
Leathwick, J.; Rowden, A.; Nodder, S.D.; Gorman, A.R.; Bardsley, S.; Pinkerton, M.; Baird, S.J.; Hadfield, M.; Currie, D.J.; Goh, A. (2012). A Benthic-Optimised Marine Environment Classification (BOMEC) for New Zealand waters. *New Zealand Aquatic Environment and Biodiversity Report. 88*. 54 p.
Lee, W.; McGLone, M.; Wright, E. (2005). Biodiversity inventory and monitoring: a review of national and international systems and a proposed framework for future biodiversity monitoring by the Department of Conservation. Landcare Research Contract Report. LC0405/122. 216 p.
Link, J.S.; Brodziak, J.K.T.; Edwards, S.F.; Overholtz, W.J.; Mountain, D.; Jossi, J.W.; Smith, T.D.; Fogarty, M.J. (2002). Marine ecosystem assessment in a fisheries management context. *Canadian Journal of Fisheries and Aquatic Sciences 59(9)*: 1429–1440. <http://dx.doi.org/10.1139/f02-115>
Lundquist, C.J.; Pritchard, M.; Thrush, S.F.; Hewitt, J.E.; Greefield, B.L.; Halliday, J.; Lohrer, A.M. (2013). Bottom disturbance and seafloor community dynamics: develpment of a model of disturbance and recovery dynamics for marine benthic ecosystems. . *New Zealand Aquatic Environment and Biodiversity Report No. 118.* 59 p.
Magurran, A.E. (2004). Measuring biological diversity. Blackwell, Oxford. 256 p.
Martinez-Crego, B.; Alcoverro, T.; Romero, J. (2010). Biotic indices for assessing the status of coastal waters: a review of strengths and weaknesses. *Journal of Environmental Monitoring 12(5)*: 1013–1028. <http://dx.doi.org/10.1039/b920937a>
Ministry for the Environment (2000). The New Zealand Biodiversity Strategy. Ministry for the Environment. Wellington, New Zealand, 144 p.

Ministry of Fisheries (2005). Strategy for Managing the Envronmental Effects of Fisheries. Ministry of Fisheries. Wellington, New Zealand, 20 p.

Ministry of Fisheries. (2010). National Fisheries Plan for Deepwater and Middle-Depth Fisheries. Ministry of Fisheries, Wellington, New Zealand. 59 p.

Nodder, S.D. (2007). Factual Voyage Report TAN0705 OS 20/20 Chatham-Challenger Hydrographic, Biodiversity & Seabed Habitats Project – Voyage 2: Chatham Rise Biodiversity. (Unpublished report held by Ministry for Primary Industries, Wellington.)

Nodder, S.D.; Bowden, D.A.; Pallentin, A.; Mackay, K.A. (2012). Seafloor Habitats and Benthos of a Continental Ridge: Chatham Rise, New Zealand. *In*: Harris, P.T. (ed.). Seafloor Geomorphology as Benthic Habitat, pp. 763–776. Elsevier Inc.,

O'Driscoll, R.L. (2012). Voyage Report TAN1208 - Project DEE2011/05: Further development of acoustic methodologies and assessment of New Zealand's deepwater fisheries and habitats. (Unpublished report held by Ministry for Primary Industries, Wellington.) 31 p.

O'Driscoll, R.L.; MacGibbon, D.; Fu, D.; Lyon, W.; Stevens, D. (2011). A review of hoki and middle-depths trawl surveys of the Chatham Rise, January 1992–2010. *New Zealand Fisheries Assessment Report 2011/47*. 72 p.

Probert, P.K.; McKnight, D.G.; Grove, S.L. (1997). Benthic invertebrate bycatch from a deep-water trawl fishery, Chatham Rise, New Zealand. *Aquatic Conservation-Marine and Freshwater Ecosystems 7*: 27–40.

Puig, P.; Canals, M.; Company, J.B.; Martin, J.; Amblas, D.; Lastras, G.; Palanques, A.; Calafat, A.M. (2012). Ploughing the deep sea floor. *Nature advance online publication*. <http://dx.doi.org/http://www.nature.com/nature/journal/vaop/ncurrent/abs/nature11410.html#supplementary-information>

Quinn, G.P.; Keough, M.J. (2002). Experimental design and data analysis for biologists. Cambridge, Cambridge University Press. 537 p.

Ramirez-Llodra, E.; Tyler, P.A.; Baker, M.C.; Bergstad, O.A.; Clark, M.R.; Escobar, E.; Levin, L.A.; Menot, L.; Rowden, A.A.; Smith, C.R.; Van Dover, C.L. (2011). Man and the Last Great Wilderness: Human Impact on the Deep Sea. *PLoS ONE 6(8)*: e22588. <http://dx.doi.org/e2258810.1371/journal.pone.0022588>

Rees, H.L.; Pendle, M.A.; Waldcock, R.; Limpenny, D.S.; Boyd, S.E. (1999). A comparison of benthic biodiversity in the North Sea, English Channel, and Celtic Seas. *ICES Journal of Marine Science 56(2)*: 228–246. <http://dx.doi.org/10.1006/jmsc.1998.0438>

Rhoads, D.C.; Ward, R.; Aller, J.; Aller, R. (2001). The importance of technology in benthic research and monitoring: Looking back to see ahead. *Organism-Sediment Interactions.* 1–17 p.

Rice, J.; Arvanitidis, C.; Borja, A.; Frid, C.; Hiddink, J.G.; Krause, J.; Lorance, P.; Ragnarsson, S.A.; Skold, M.; Trabucco, B.; Enserink, L.; Norkko, A. (2012). Indicators for Sea-floor Integrity under the European Marine Strategy Framework Directive. *Ecological Indicators 12(1)*: 174–184. <http://dx.doi.org/10.1016/j.ecolind.2011.03.021>

Robinson, L.A.; Frid, C.L.J. (2008). Historical marine ecology: Examining the role of fisheries in changes in North Sea benthos. *Ambio 37(5)*: 362–371. <http://dx.doi.org/10.1579/07-a-300.1>

Rogers, S.I.; Somerfield, P.J.; Schratzberger, M.; Warwick, R.; Maxwell, T.A.D.; Ellis, J.R. (2008). Sampling strategies to evaluate the status of offshore soft sediment assemblages. *Marine Pollution Bulletin 56(5)*: 880–894. <http://dx.doi.org/10.1016/j.marpolbul.2008.01.035>

Rowden, A.; Kroeger, K.; Clark, M. (2012). Biodiversity of macroinvertebrate and fish assemblages of the northwestern Ross Sea shelf. *New Zealand Aquatic Environment and Biodiversity Report No. 101*.

Rowden, A.A.; Clark, M.R. (2010). Benthic biodiversity of seven seamounts at the southern end of the Kermadec volcanic arc, northeast New Zealand. *New Zealand Aquatic Environment and Biodiversity Report No. 62*.

Rumohr, H. (1995). Monitoring the marine environment with imaging methods. *Scientia Marina 59*: 129–138.

Schallenberg, M.; Kelly, D.; Clapcott, J.E.; D'eath, R.; MacNeil, C.; Young, R.G.; Sorrell, B.; Scarsbrook, M. (2011). Approaches to assessing ecological integrity of New Zealand freshwaters. *Science for Conservation 307*.

Sharp, B.; Pinkerton, M.; Leathwick, J. (2007). Marine classification: lessons from the New Zealand experience. CCAMLR paper WS-BSO-07/6. 22 p.

Sheehan, E.V.; Stevens, T.F.; Attrill, M.J. (2010). A Quantitative, Non-Destructive Methodology for Habitat Characterisation and Benthic Monitoring at Offshore Renewable Energy Developments. *PLoS ONE 5(12)*. <http://dx.doi.org/10.1371/journal.pone.0014461>

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. (2007). Development of an ecologic marine classification in the New Zealand region. *Environmental Management 39(1)*: 12–29.

Solan, M.; Germano, J.D.; Rhoads, D.C.; Smith, C.; Michaud, E.; Parry, D.; Wenzhofer, F.; Kennedy, B.; Henriques, C.; Battle, E.; Carey, D.; Iocco, L.; Valente, R.; Watson, J.; Rosenberg, R. (2003). Towards a greater understanding of pattern, scale and process in marine benthic ecosystems: a picture is worth a thousand worms. *Journal of Experimental Marine Biology and Ecology 285–286*: 313–338.

Stelzenmueller, V.; Rogers, S.I.; Mills, C.M. (2008). Spatio-temporal patterns of fishing pressure on UK marine landscapes, and their implications for spatial planning and management. *ICES Journal of Marine Science 65(6)*: 1081–1091. <http://dx.doi.org/10.1093/icesjms/fsn073>

Thistle, D. (2003). The deep-sea floor: An overview. *In*: Tyler, P.A. (ed.). Ecosystems of the Deep Ocean, pp. 5–37. *Ecosystems of The World*. Elsevier Science BV, Amsterdam.

Thompson, R.M. (1991). Gazetteer of seafloor features in the New Zealand region. *Miscellaneous Publication 104*. New Zealand Oceanographic Institute, Wellington, New Zealand.

Thrush, S.; Hewitt, J.; Lundquist, C.J.; Townsend, M.; Lohrer, A. (2011). A strategy to assess trends in the ecological integrity of New Zealand's marine ecosystems. NIWA Client Report HAM2011-140, NIWA, Hamilton, New Zealand. 58 p.

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.; Gray, J.S.; Hewitt, J.E.; Ugland, K.I. (2006). Predicting the effects of habitat homogenization on marine biodiversity. *Ecological Applications 16(5)*: 1636–1642.

Thrush, S.F.; Hewitt, J.E.; Cummings, V.J.; Dayton, P.K.; Cryer, M.; Turner, S.J.; Funnell, G.A.; Budd, R.G.; Milburn, C.J.; Wilkinson, M.R. (1998). Disturbance of the marine benthic habitat by commercial fishing: Impacts at the scale of the fishery. *Ecological Applications 8(3)*: 866–879.

Thrush, S.F.; Hewitt, J.E.; Cummings, V.J.; Norkko, A.; Chiantore, M. (2010). beta-Diversity and Species Accumulation in Antarctic Coastal Benthos: Influence of Habitat, Distance and Productivity on Ecological Connectivity. *PLoS ONE 5(7)*. <http://dx.doi.org/e1189910.1371/journal.pone.0011899>

Thrush, S.F.; Hewitt, J.E.; Hickey, C.W.; Kelly, S. (2008). Multiple stressor effects identified from species abundance distributions: Interactions between urban contaminants and species habitat relationships. *Journal of Experimental Marine Biology and Ecology 366(1–2)*: 160–168. <http://dx.doi.org/10.1016/j.jembe.2008.07.020>

Tillin, H.M.; Hiddink, J.G.; Jennings, S.; Kaiser, M.J. (2006). Chronic bottom trawling alters the functional composition of benthic invertebrate communities on a sea-basin scale. *Marine Ecology Progress Series 318*: 31–45. <http://dx.doi.org/10.3354/meps318031>

Tuck, I.D.; Cole, R.; Devine, J. (2009). Ecosystem indicators for New Zealand fisheries. *New Zealand Aquatic Environment and Biodiversity Report No. 42*. 188 p.

Van der Meer, J. (1997). Sampling design of monitoring programmes for marine benthos: A comparison between the use of fixed versus randomly selected stations. *Journal of Sea Research 37(1–2)*: 167–179. <http://dx.doi.org/10.1016/s1385-1101(97)00007-5>

Williams, A.; Schlacher, T.A.; Rowden, A.A.; Althaus, F.; Clark, M.R.; Bowden, D.A.; Stewart, R.; Bax, N.J.; Consalvey, M.; Kloser, R.J. (2010). Seamount megabenthic assemblages fail to recover from trawling impacts. *Marine Ecology 31*: 183–199. <http://dx.doi.org/10.1111/j.1439-0485.2010.00385.x>

9 APPENDIX 1

Table A 1: Available data sets on benthic invertebrate distributions in New Zealand waters.

Dataset	Contact	Extent	Resolution	Parameters	No. records or stations	Description
BOMEC						
BOMEC classification	NIWA	EEZ	1 × 1 km	Class	N/A	Benthic Optimised Marine Environment Classification: a general classification of marine habitats derived by training species-environment models with data layers compiled from all available sources of benthic invertebrate data (Leathwick et al 2012).
Demersal fish	NIWA	EEZ	1 × 1 km	count/weight	17 000	Data layer: distributions of 38 demersal fish species throughout New Zealand waters (bounded by the 3000 m contour) prepared from the fish_comm db held at NIWA (research trawl survey data 1979–2005) for use in training the BOMEC.
Scleractinia	NIWA	EEZ	1 × 1 km	count	354	Data layer: distributions of six scleractinian coral species throughout New Zealand waters (bounded by the 3000 m contour) prepared from a variety of sources held at NIWA (including *Specify* db) for use in training the BOMEC.
Octocoralia	NIWA	EEZ	1 × 1 km	count	725	Data layer: distributions of 23 octocoral families throughout New Zealand waters (bounded by the 3000 m) prepared from a variety of sources held at NIWA (including *Specify* db) for use in training the BOMEC.
Porifera	NIWA	EEZ	1 × 1 km	presence	146	Data layer: distributions of sponge species (based on counts) throughout New Zealand waters (bounded by the 3000 m contour) prepared from a variety of sources held at NIWA (including *Specify* db) for use in training the BOMEC.
Polychaeta	NIWA	EEZ	1 × 1 km	count	2,114	Data layer: distributions of all polychaete records throughout New Zealand waters (bounded by the 3000 m contour) prepared from the MS Access *AllSeaBio* database held at NIWA (all records also in *Specify* db) for use in training the BOMEC.
Foraminfera	NIWA	EEZ	1 × 1 km	count	244	Data layer: distributions of foraminifera throughout New Zealand waters (bounded by the 3000 m contour) prepared from NIWA data and data supplied by B. Hayward/OBIS for use in training the BOMEC.
Ocean Survey 20/20						
Chatham-Challenger						http://www.os2020.org.nz/chatham-challenger-project/
Epifauna (sled and trawl)	NIWA	Chat-Chall	Station	count	1 196	Identifications and abundances of all benthic epifauna sampled by epibenthic sled and beam trawl during OS 20/20 voyages TAN0705 and TAN0707 to Chatham Rise and Challenger Plateau, respectively. Separate datasets give number of individuals of each organism by site for the two sampling methods.
Epifauna (DTIS stills)	NIWA	Chat-Chall	Station	count	1 238	Identifications and abundances of benthic epifauna from DTIS still camera photographs during OS 20/20 voyages TAN0705 and TAN0707. Number of individuals per sq.m per image for each station and site for 152 sessile taxa. A total of 924 images for CHAT and 314 for CHAL. Taxonomic information is also stored in the dataset.
Epifauna (DTIS video)	NIWA	Chat-Chall	Station	count	387 stations	Identifications and abundances of benthic epifaunal taxa from DTIS video camera transects during

Dataset	Contact	Extent	Resolution	Parameters	No. records or stations	Description
						OS 20/20 voyages TAN0705 and TAN0707.
Diversity indices	NIWA	Chat-Chall	Station	Index	359 stations	A range of univariate metrics describing benthic faunal biodiversity across Chatham Rise and Challenger Plateau based on data collection over the OS20/20 surveys.
Taxonomic distinctness	NIWA	Chat-Chall	Station	Index	346 stations	Taxonomic distinctness metrics calculated for selected benthic faunal groups across Chatham Rise (TAN0705) and Challenger Plateau (TAN0707).
Biotic Habitats	NIWA	Chat-Chall	1 × 1 km	Class	N/A	Biotic habitat classes derived from statistical classification of benthic fauna and substrate data from DTIS and seamount sled samples across Chatham Rise (TAN0705) and Challenger Plateau (TAN0707).
Bay of Islands Coastal Survey						http://www.os2020.org.nz/bay-of-islands-coastal-survey-project/
Epifauna (DTIS video+stills)	NIWA	BoI	Station	count	58 stations	Identifications and abundances of benthic epifauna from DTIS video camera transects during OS 20/20 voyages KAH0905 and TAN0906.
Epifauna (sled and trawl)	NIWA		Station	count	89 stations	Identifications and abundances of all benthic epifauna sampled by epibenthic sled and beam trawl during OS 20/20 voyage TAN0906.
Chatham Benthos						
Epifauna (DTIS video+stills)	NIWA	Chatham Rise	Station	count	59 stations	Identifications and abundances of benthic epifauna from DTIS video camera transects during OS 20/20 voyage TAN1306.
Infauna (macro)	NIWA	Chatham Rise	Station	count	35 stations	Identifications and abundances of benthic macro-infauna from multicorer during OS 20/20 voyage TAN1306.
Biodiversity databases						
Biods	MPI-NIWA	EEZ	Station	count/weight	>90 000	Station data and associated taxonomic records for catch or samples from biodiversity trips that targeted areas such as seamounts and seeps.
Specify	NIWA	EEZ	Station	count/weight	>90 000	Information about specimens held in the NIWA Invertebrate Collection. Benthic invertebrate specimens returned from research trawl, biodiversity, and geological surveys, and from MPI observers on commercial fishing trips are recorded in this database. The taxonomic level of each record represents the lowest taxonomic level currently known for a specimen. Records are updated (and the taxonomy is revised) on occasions when experts are able to check and verify the identifications. At a later stage, an updated record in specify may then be used to update the 'parent' information in *trawl*, *biods*, or *cod*. The station records for these invertebrate specimens are accessible in marineDB. This database superseded *AllSeaBio* (and contains an unknown proportion of *AllSeaBio* records).
AllSeabio	NIWA		Station	count	Unknown	Historic benthic data and NZ Oceanographic Institute benthic data and was 'decommissioned' in 1995. It has been replaced by *Specify* but an unknown proportion of the *AllSeaBio* records are now in *Specify*.

Dataset	Contact	Extent	Resolution	Parameters	No. records or stations	Description
Fisheries databases						
Trawl db	NIWA	EEZ	Station	weight	>100 000	Records from trawl surveys on NIWA or industry vessels. Some early (pre 2003) Biodiversity trips are also recorded in this database. All catch is recorded by weight and a code that represents the lowest taxonomic level to which the organism can be identified. A total of 1128 codes that represent the variety of catch: teleosts and elasmobranchs, cephalopods, Cnidaria, porifera, polychaetes, seaweeds, crustaceans, echinoderms, molluscs, plankton. Biological data are collected (length, weight, otoliths, gonad stage, sex, etc). Where an organism in the catch could not be identified at sea, it was generally returned for identification and given a placeholder generic code which is later updated once an expert has identified the specimen to whatever taxonomic level possible. These updates are not obvious in the database: i.e., the verified ids replace the original ids and thus it is not possible to extract verified data from the database as a simple select. The updating process is ongoing. Benthic invertebrate identification is being improved with the introduction of guides (from 2004).
Cod db	MPI-NIWA	EEZ	variable	weight	>100 000	Observer catch and effort data from trawl and bottom longline trips that give information on the position, date, gear type and parameters, and catch for each observed event. The data collection emphasis has been on the fish catch, though corals have often been recorded (using generic codes), but not consistently. In January 2008 a new form was introduced - Benthic Bycatch Form - and this has resulted in more benthic bycatch entries in the database. During 2007–08 to 2009–10, observers on vessels targeting deepwater fish species were requested to report all coral bycatch. Returned samples have been used to verify some observer ids: these samples and their ids are also stored in the Specify db. Identifications of benthos for catch not returned for expert examination are thought to be better and less generic from about 2004 when the first comprehensive benthic id guides for use at sea were introduced.
Warehou db	MPI-NIWA	EEZ	variable	weight	>100 000	Trawl catch effort and processing data collected on TCEPR forms and owned by MPI. Data include station details and the catch of the top five species for TCEPR and the top 8 species for TCER forms (latter usually used by smaller vessels unlikely to be fishing beyond 200 m. Fine scale position data need to be requested to get the best resolution (which equates to the nearest nautical mile). Start and finish positions and times allow for the development of swept area impact indices. Work from BEN200601 provides a dataset of TCEPR swept areas at the level of a tow or by 25 sq.km cell for fishing years 1989–90 to 2004–05. These commerical catch-effort data are used by GNS to create footprint of various fisheries, and in a recent iteration there has been some attempt to include a measure of intensity (final report not out yet), so presumably those layers would be available, from MPI. Bottom longline effort collected on LCERs are also available here.

10 APPENDIX 2

Table A 2: List of benthic invertebrate taxa present in beam trawl, rat-catcher trawl, and net bag samples from TAN1208.

Scientific name	Common name	Code
Acanthephyra spp.	Subantarctic ruby	ACA
Actinostolidae	Deepsea anemone	ACS
Acutiserolis spp.	Spiny serolid	ACU
Alcithoe wilsonae	*Alcithoe wilsonae*	AWI
Alcyonacea (Order)	Soft coral	SOC
Alcyonacea, Gorgonacea, Scleractinia, Antipatharia, Stylasteridae	Coral (unspecified)	COU
Amphipoda	Amphipod	APH
Anthozoa	Anemones	ANT
Aristaeopsis edwardsiana	Scarlet prawn	PED
Ascidiacea	Sea squirt	ASC
Asteroidea	Asteroid (starfish)	ASR
Bathyplotes spp.	*Bathyplotes* spp.	BAM
Benthopecten spp.	*Benthopecten* spp.	BES
Bivalvia	Bivalves	BIV
Bolocera spp.	Deepsea anemone	BOC
Brisingida	Brisingida	BRG
Brachyura	Crab	CRB
Bryozoa (Phylum)	Bryozoan	COZ
Camplyonotus rathbunae	Sabre prawn	CAM
Cirripedia (Class)	Barnacle	BRN
Coelenterata	Coelenterata	COE
Cosmasterias dyscrita	*Cosmasterias*	CDY
Crossaster multispinus	Sun star	CJA
Decapoda	Natant decapod	NAT
Diacanthurus rubricatus	Pagurid	DIR
Dipsacaster magnificus	*Dipsacaster*	DMG
Echinoidea	Echinoid (sea	ECN
Echinothuriidae & Phormosomatidae	Tam O'Shanter urchin	TAM
Enypniastes eximia	*Enypniastes*	EEX
Epizoanthus spp.	*Epizoanthus* spp.	EPZ
Euciroa galatheae	*Euciroa*	EGA
Euplectella regalis	Basket-weave	ERE
Flabellum spp.	*Flabellum* coral	COF
Fusitriton magellanicus	*Fusitriton*	FMA
Galatheidae	Galatheid	GAL
Gastropoda	Gastropods	GAS
Gnathophausia ingens	Giant red mysid	NEI
Goniocidaris parasol	Sea urchin	GPA
Goniocorella dumosa	Bushy hard coral	GDU

Scientific name	**Common name**	**Code**
Gorgonacea (Order)	Gorgonian coral	GOC
Gracilechinus multidentatus	Sea urchin	GRM
Henricia compacta	*Henricia*	HEC
Holothurian unidentified	Sea cucumber	HTH
Hormathiidae	Deepsea anemone	HMT
Hyalinoecia tubicola	Quill worm	HTU
Hydrozoa (Class)	Hydroid	HDR
Isopoda	Isopod	ISO
Laetmogone spp.	*Laetmogone* spp.	LAG
Lipkius holthuisi	Omega prawn	LHO
Lithosoma novaezelandiae	Rock star	LNV
Metanephrops challengeri	Scampi	SCI
Mollusca	Molluscs	MOL
Munida gracilis	*Munida gracilis*	MGA
Notostomus auriculatus	*Notostomus*	NAU
Nudibranchia (Order)	Nudibranchia	NUD
Ophiomusium lymani	*Ophiomusium*	OLY
Ophiophthalmus relictus	Deepsea brittle	ORE
Ophiuroidea	Ophiuroid (brittle	OPH
Oplophorus spp.	*Oplophorus* spp.	OPP
Paguroidea	Pagurid	PAG
Pannychia moseleyi	*Pannychia*	PAM
Paramaretia peloria	Heart urchin	PMU
Pasiphaea aff. *tarda*	Deepwater prawn	PTA
Penion chathamensis	*Penion*	PCH
Pennatula spp.	Purple sea pen	PNN
Phorbas spp.	Grey fibrous	PHB
Pillsburiaster aoteanus	*Pillsburiaster*	PAO
Plutonaster knoxi	Abyssal star	PKN
Polychaeta	Polychaete	POL
Polycheles spp.	Polychelidae	PLY
Porifera (Phylum)	Sponges	ONG
Priapulida	Penis worms	PDL
Provocator mirabilis	Golden volute	GVO
Pseudechinus flemingi	Sea urchin	PFL
Pseudostichopus mollis	*Pseudostichopus*	PMO
Psilaster acuminatus	Geometric star	PSI
Pycnogonida	Sea spiders	PYC
Pycnoplax victoriensis	Two-spined crab	CVI
Scleractinia	Stony corals	SIA
Sipuncula	Unsegmented worms	SIP
Spatangus multispinus	Heart urchin	SPT
Stephanocyathus platypus	Solitary bowl coral	STP
Sympagurus dimorphus	Pagurid	SDM

Scientific name	**Common name**	**Code**
Thenea novaezelandiae	Yoyo sponge	THN
Trichopeltarion fantasticum	Frilled crab	TFA
Vitjazmaia latidactyla	Deep sea spider crab	VIT
Zoroaster spp.	Rat-tail star	ZOR