Ministry for Primary Industries
Manatū Ahu Matua

Assessing the effects of mobile bottom fishing methods on benthic fauna and habitats: report from a workshop in 2015

New Zealand Fisheries Science Review 2016/2

R.B. Ford,
W.N.S. Arlidge,
D.A. Bowden,
M.R. Clark,
M. Cryer,
A. Dunn,
J.E. Hewitt,
J.R. Leathwick,
M.E. Livingston,
C.R. Pitcher,
A.A. Rowden,
S.F. Thrush,
G.A. Tingley,
I.D. Tuck,

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

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

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Preface

The Ministry for Primary Industries and its predecessor, the Ministry of Fisheries, have conducted fully-independent expert reviews of stock assessments, research methodologies and research programmes since 1998. We also run specialist technical review workshops to further advance fisheries and other marine science methodologies and techniques. These fully-independent reviews and technical workshops are separate from, but complementary to, the annual Science Working Group processes that are used to ensure the objectivity and reliability of most of our scientific research and analyses.

A new publication series, Fisheries Science Reviews, was initiated in 2015 to ensure that reports from these reviews are readily accessible. The series will include all recent and new fully-independent reviews and technical workshop reports, and will also incorporate as many historical reports as possible, as time allows. In order to avoid confusion about when the reviews were actually conducted, all titles will include the year of the review. They may also include appendices containing the Terms of Reference, a list of participants, and a bibliography of supporting documents, where these have not previously been incorporated. Other than this, there will be no changes made to the original reports composed by the independent experts or workshop participants.

Fisheries Science Reviews (FSRs) contain a wealth of information that demonstrate the utility of the processes the Ministry uses to continually improve the scientific basis for managing New Zealand’s fisheries.

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EXECUTIVE SUMMARY

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

A diverse group of experts (Appendix 2) convened in February 2015 for a workshop to address the issue: "*What is the best scientific approach to assessing trawl and dredge impacts on benthic fauna and habitats in New Zealand in the short, medium and long-term?"* The current approach uses the overlap of the trawl/dredge footprint with Marine Environment Classification (MEC), Benthic Optimised MEC (BOMEC), depth classes or more arbitrary divisions. This approach is not universally accepted by the various stakeholders and practitioners in New Zealand, and opinions diverge on its value (from adding some value to adding no value).

In the short to medium-term the workshop participants reached a compromise that a fishing impact/productivity modelling approach to benthic risk assessment was a useful starting point. This approach would be broadly similar to the south-eastern Australian marine region (SEMR) approach, where overlap is gauged between fishing footprint and distribution of species or habitats. The SEMR approach assumed that a level of impact from fishing on a species could then be estimated for some taxa, groups or habitats based on the fishing gear used and the functional traits of the organism, e.g. fragility and position in/on the seafloor, etc. The relative fishing impact rate (here called *I*) can then be calculated across the entire area and divided by some management target or threshold measure of productivity (here called *P*). In the SEMR approach, the productivity *P* was defined as equal to the estimated natural mortality rate of the organism[1]. Then the ratio *I/P* becomes a measure of relative impact, with those taxa with the highest values of *I*/*P* deemed most at risk. Values of *I/P* greater than one imply an impact above the management threshold; and values less than one imply an impact below the management threshold. In the SEMR approach the value of *I*/*P* equal to one can be thought of as conceptually equivalent to Maximum Sustainable Yield (MSY). This approach makes the assumption that recovery rate is correlated with productivity, i.e. as productivity decreases (for longer-lived organisms) then taxa take longer to recover following disturbance. Notably, the *P* of the slowest recovering species present, e.g. cold water coral colonies on seamounts, can also be used as a proxy for recovery rate of habitats. Comparing *I* to *P* within this framework is powerful as a screening device for prioritising which species or habitats are most likely to warrant research or management attention.

This fishing impact/productivity modelling approach would help address the management need to ensure sustainability of benthic impacts by taking a quantitative risk-based approach. This would be a feasible, short-term solution which uses a population productivity based assessment to identify fishing impacts in a spatially-explicit manner. This approach will allow for disaggregation of the impacts both spatially and by impact type (e.g., fleet, gear, etc.). However, this approach assumes additivity of risks and does not cater for interactions between risks that are antagonistic (decreasing in combination) or synergistic (increasing in combination), cumulative impacts or environmental thresholds. In the longer-term we aim to develop risk assessments or management approaches that can more realistically capture or cater for these complexities.

This approach will rely on using available information, so there are a number of limitations to this approach, or assumptions that will need to be made. Distributions of many species (particularly species deeper than 1500 m) are not well known. Predictive models of species or habitat distributions will need to be used where trusted, otherwise expert judgements about distribution and abundance may be needed. For rarer species, that have few sampled occurrences, this technique may not be applicable (although it may be possible to assess impacts on functional traits or species richness). For species where mortality rate or functional traits are unknown, they may need to be assumed from related taxa, or expert judgement may be necessary. Where expert judgement is required expert workshops will be convened and informed by whatever data are available. Where assumptions are necessary, these will be documented and transparent, so that better information can be substituted if it becomes available. Quantifying uncertainty remains difficult, but not impossible using this impact/productivity modelling approach. Using a number of fundamentally different modelling approaches was favoured to understand the true uncertainty, and the influence of alternative modelling assumptions on the resulting estimates of relative risk. In practice, this means a range of predictions of risk are likely to be generated along with different estimates of uncertainty.

[1] Notably care will be needed in determining an appropriate mortality rate for colonial or habitat forming organisms in particular if there functional roles are to be recognised.

The above process is fit for purpose and comparable to other fisheries risk assessments in New Zealand and elsewhere, but relies on a number of development steps which may take a number of years. In the mid-term it is hoped that improved information on factors like distribution, fragility and recovery rate could be generated to replace assumptions within any risk assessment framework.

Qualitative risk assessment workshops, based on available data, have been used by the Ministry for Primary Industries to provide short-term, less robust and transparent assessments of risk in some environmental areas. However, given the high numbers of benthic species for which information is not available, the logistics of assembling such an expert workshop (or more than one workshop) has yet to be evaluated, and are likely to be highly challenging. No alternative shorter-term solutions were identified by the workshop participants to evaluating benthic risk from trawling and dredging in New Zealand.

The *I/P* pragmatic approach, if effective, will advance current practice, but lags behind best practice as it does not consider cumulative impacts, impacts on biodiversity, habitats or ecosystems for which impact is poorly known or unpredictable. The suggested qualitative risk assessment is also unlikely to deal well with cumulative effects, ecosystem change or threshold responses in ecosystems, due to our lack of data on these. In the longer-term it is hoped that these factors can be addressed when considering benthic impacts, but it will require additional data and methods to achieve this synthesis and these are also still under development.

1 INTRODUCTION

Bottom trawling and dredging are used to harvest a wide variety of fish and invertebrate species in many of New Zealand's coastal and offshore fisheries (MPI 2014, 2015). The implications of seabed disturbance by these gears vary by habitat type, gear type, mode of use, and the frequency and intensity of disturbance (see MPI 2014 for a brief summary).

New Zealand has environmental obligations under the Fisheries Act (1996) to avoid, remedy or mitigate adverse effects of fishing by maintaining associated/dependent species above a level that ensures long-term viability, and protect habitats of significance for fisheries management. Overall, the aim of these obligations to ensure the maintenance of marine biodiversity, including protected species, in the region. New Zealand is also a signatory to a number of international agreements, which have their own environmental requirements, which relate to the potential impacts of fishing. These include the United Nations Convention on the Law of the Sea (UNCLOS), Convention on Biological Diversity (CBD), International Plan of Action – seabirds (IPOA-seabirds), International Plan of Action – sharks (IPOA-sharks), Regional Fisheries Management Organisations (RFMOs), Convention for the Conservation of Antarctic Marine Living Resources (CCAMLR), and the Agreement on the Conservation of Albatrosses and Petrels (ACAP).

In addition, a number of New Zealand fisheries have Marine Stewardship Council[2] certifications (MSC) or other market-driven environmental certifications which demand their own and often higher levels of environmental performance. For example, there is the expectation that in approximately two years the MSC certified fisheries will have to consider cumulative impacts on fish and protected species across all their certified fisheries in an area. As many of New Zealand's deepwater fisheries are currently certified (or applying for certification) this expectation heightens the need for an effective and verified assessment tool for assessing the effects of bottom fishing on the benthos.

Due to these legal and regulatory obligations fisheries management agencies need to ensure that benthic impacts are sustainable. At present no quantitative fisheries management objective for determining the effect of bottom fishing on the benthos exists. However, the biomass that supports maximum sustainable yield (MSY) is used as a target for fish species stock assessments and potential biological removals (PBR) are used as a reference point for seabird populations (Richard & Abraham 2013). Therefore an analogous concept to MSY, PBR, or both of these, for the benthos may be a useful starting point to develop a method for assessing benthic interactions.

The Ministry for Primary Industries (MPI) is taking a risk assessment-based approach to evaluate a variety of environmental impacts of the seafood sector. Risk assessment can be used as a tool to prioritise different management or research options. Alternately, if agreed standards exist, risk assessment can also allow an assessment of the need for, and relative prioritisation of, management or research interventions. To assess risk, MPI uses a framework comparable to the Australian Ecological Risk Assessment for the Effects of Fishing (ERAEF, after Hobday et al. 2007, 2011). This recognises three levels of risk assessment, each with their own strengths and weaknesses (Table 1). Progress in New Zealand has been made on assessing risk using different levels of assessment across a number of environmental components at a population level (Table 2). Progress has yet to be made when assessing risk at the level of the ecosystem, or for individual groups considering cumulative impacts in anything but an additive fashion (Table 2). The most realistic assessment would potentially allow risks to be additive, synergistic (increase in combination) or antagonistic (decrease in combination). For management application risk scores should be able to be disaggregated to different fisheries so that management measures can be focussed on those fisheries creating the risk. The current approach to risk, is acknowledged to be overly simplistic, but has however proved useful, for example, the results of the seabird risk assessment (Richard & Abraham 2013) allowed managers to focus attention on those birds most at risk, out of the 70 species considered.

A key aspect of assessing the risk to benthic habitats is a good understanding of the spatial and temporal distribution of fisheries, and how this can be compared to the distribution of the benthos in a biologically meaningful way.

The distribution and intensity of bottom trawling in both coastal and offshore trawl fisheries in New Zealand (see Figure 1 for examples) is now well described at a 5 km by 5 km grid scale (Baird et al. 2002, 2011; Black & Tilney 2015; Baird et al. 2015). Shellfish dredging is still reported with low spatial precision (MPI 2014), but

[2] https://www.msc.org/ and http://deepwatergroup.org/certification/

most of the frequently-fished beds are persistent and well-known (Williams 2009, Williams et al. 2007, 2013, 2014), so reasonable inferences can be made about the distribution of fishing.

Table 1: Levels of ERAEF risk assessments, advantages and disadvantages (after Hobday et al. 2007, 2011).

Type	Advantages	Disadvantages
Qualitative (Level-1)	Cheap, quick, comprehensive	Subjective on reference points, not repeatable, not additive
Semi-quantitative (Level-2)	Objective, repeatable, can be comprehensive and/or additive	Somewhat data hungry, need for proxies and assumptions
Quantitative (Level-3)	Additive[3], accurate, can be used to make and test predictions	Onerous requirements for data, money, skills and time

Table 2: Environmental risk assessments completed in New Zealand and their ERAEF level (see Table 1)[4]. Blank cells indicate no progress and QMS = Quota Management System.

Group	**Level-1**	**Level-2**	**Level-3**
Seabirds	all spp.	all spp.	5 spp.
Mammals		all spp.	2 spp.
Sharks	all spp.		some QMS
Fish			some QMS
Benthos	some	some	
Ecosystem			

There have been a number of attempts to categorise New Zealand's marine benthos on large scales (hundreds of kilometres) on the basis of benthic assemblage composition data and/or information on inferred environmental drivers of the benthic composition. For example, for marine protected area planning, a coastal classification and mapping scheme has been developed based on environment type, depth, exposure and benthic habitat type (Ministry of Fisheries and Department of Conservation 2008). More sophisticated modelling approaches have also been completed which relate benthic community composition to environmental gradients. This has resulted in the general-purpose Marine Environment Classification (MEC, Snelder et al. 2005, 2007), a demersal fish-optimised MEC (Leathwick et al. 2006a) and the Benthic-Optimised MEC (BOMEC, Leathwick et al. 2012). The BOMEC has been evaluated as being a better predictor of environmental classes at scales of hundreds of kilometres than the original MEC (Bowden et al. 2011). None of these habitat classifications was designed to predict at a fine scale or "point" habitats like seamounts or biogenic habitats, although separate seamount classifications have been developed for New Zealand (Rowden et al. 2005) and globally (Clark et al. 2011).

Predictive modelling of species distributions (as distinct from habitat classification) has also been employed in New Zealand, for selected taxa. For example, for deepwater corals (Tracey et al. 2011a, Anderson et al. 2014), and the overlap of coral distributions with fisheries footprints (Tracey et al. 2011b).

[3] This approach considers impacts as additive across fisheries only, whilst ignoring any potential synergistic or antagonistic interactions.
[4] Notably those assessments completed are at a population level and therefore do not incorporate cumulative or ecosystem effects within these groups.

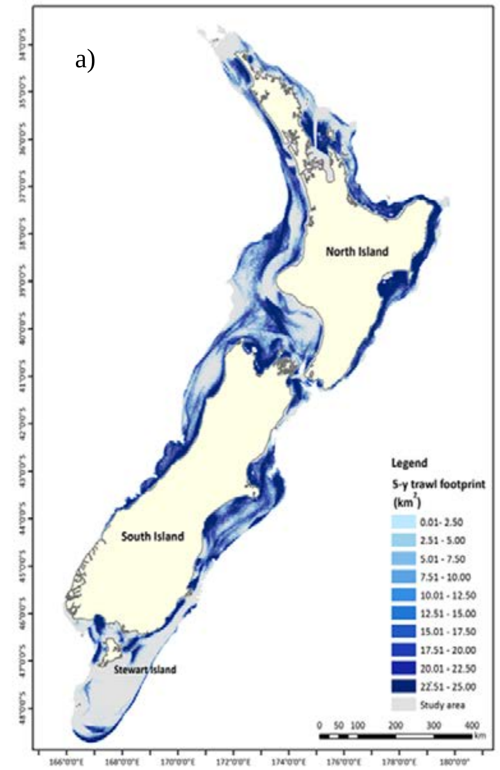
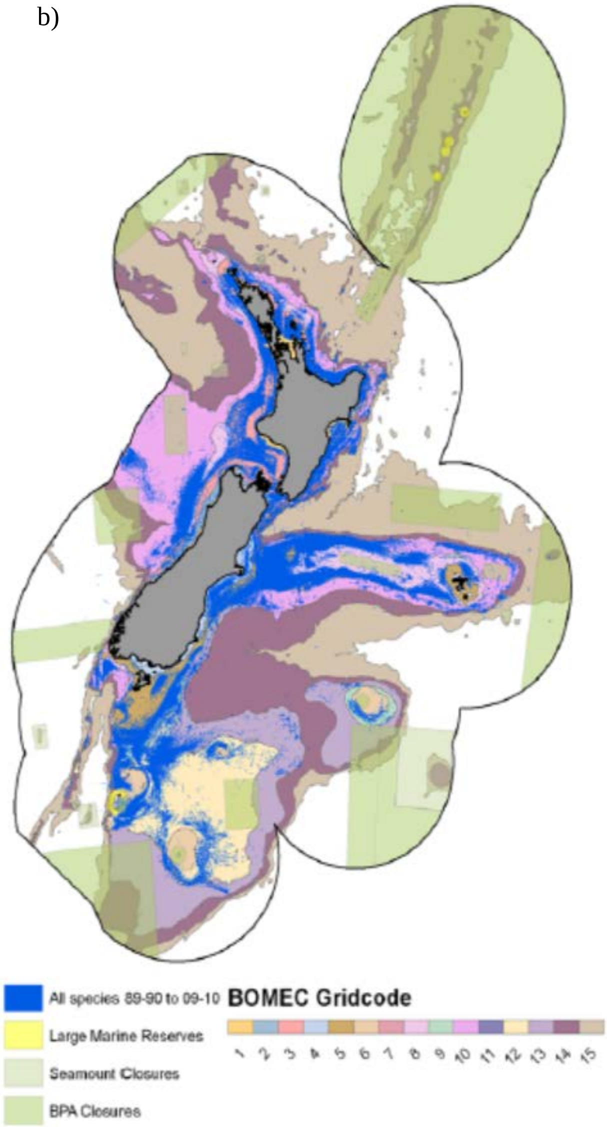


Figure 1: Visual representations of the trawl footprint from a) the coastal zone to 250 m depth from 2008 to 2012 (Baird et al. 2015), and b) deepwater fisheries (overlaid on the BOMEC) from 1989/90 to 2010/11 (Black & Tilney 2015).



Experimental assessments of trawl and/or dredge impacts (whether manipulative or mensurative) provide indications of the degree of change and, less frequently, the recovery trajectory for particular habitat types, species or species assemblages. Several such studies have been done in New Zealand. Tuck et al. (in prep.) summarises work on soft-sediments. Work on seamounts is contained in Clark et al. (2010) and in Williams et al. (2010). Results from these studies are largely consistent with overseas work and are summarised in MPI (2014), which generalises that shellfish dredges have the greatest effect of the various mobile bottom fishing gears, biogenic habitats are the most sensitive to such disturbance (especially for attached fauna on hard substrates) and shallow frequently wave disturbed sands are the least sensitive. Recovery from disturbance events can take months to decades, depending on the combination of fishing method and benthic habitat type. This is a particular issue in deep-sea habitats (Clark et al. 2015a).

While considerable information is available, it is not clear how the results outlined above can best be generalised to cover all fisheries and all benthic species and/or habitats that occur in New Zealand waters, especially where there are biogenic habitat forming organisms, and/or there is small scale heterogeneity in community composition and structure.

A number of recent developments have cast doubt on the use of both classification approaches and species distribution modelling as a basis for predicting biologically-meaningful categories for comparison with the distribution of bottom fishing effort, or for use in risk assessments:

- The BOMEC, although the best tool available for assessing benthic impacts at a New Zealand-wide scale, should not be interpreted as a map of benthic habitats and has limited explanatory power (Bowden et al. 2011). The BOMEC was therefore judged by the MSC, when assessing regional fisheries within the New Zealand EEZ against the Marine Stewardship Council sustainability standard, as needing to be interpreted with caution.

- Analyses in inshore habitats and fisheries have shown poor correlation between the sensitivity of fauna and predicted environmental classes from BOMEC (Baird et al. 2015).

- The 2014 ICES symposium "Effects of fishing on benthic fauna and habitats" in Norway[5] highlighted the need for good data coverage to produce reliable species distribution models, and doubt has been expressed by experts that enough data is available across a range of taxa to support these models.

- There were few detections of live stony coral thickets/reefs in locations modelled to support these in the SPRFMO area (Clark et al. 2015b)., and a formal test of the utility of these has shown that large-scale regional models perform poorly when applied to localised areas where there are limited underlying data (Anderson et al. 2016).

Now that the distribution of fishing is well-documented, and there have been a number of approaches to comparing this footprint with affected habitats and/or species (both in New Zealand and overseas), MPI convened a group of experts (see Appendix 1) in February 2015 to address the issue:

> **"What is the best scientific approach to assessing trawl and dredge impacts on benthic fauna and habitats in New Zealand in the short, medium and long-term?"**

The primary focus of the workshop was to advise MPI on the strengths, weaknesses, opportunities, and costs of different approaches to assessing trawl and dredge impacts on benthic fauna and habitats using existing data. The immediate use of this advice will be to aid the design of an initial risk assessment for benthic ecosystems to be conducted from 2016. For longer term (about 10 years) advice, the workshop focus was on describing what scientific approaches and data could underpin alternative types of management of benthic impacts. The workshop also sought to assess if substantial improvements in our ability to understand benthic impacts are potentially possible in the medium term (3–5 years), such that advice on how to reach that goal could be developed to guide MPI research planning. The terms of reference for the workshop, including an agenda and participant list, are attached (Appendix 1).

2 PRESENTATIONS

A number of presentations were viewed, after the workshop context was given, in the following order:

- Existing data sources in New Zealand – David Bowden
- Update on understanding of benthic impacts in New Zealand – Ian Tuck
- Approaches to benthic species/habitat prediction/classification
    - Marine Environmental Classifications (MEC) approaches (John Leathwick)
    - Benthic risk assessment in Australia and beyond (Roland Pitcher)
    - Species distribution modelling in New Zealand (Ashley Rowden)
    - Spatial priority ranking software approaches, e.g. Zonation (John Leathwick)
    - Direct modelling of consequences (Alistair Dunn)
    - Science to support managing for resilience (Simon Thrush)

These will be summarised in order below.

[5] http://www.ices.dk/news-and-events/symposia/Effects/Pages/Effects%20of%20fishing%20on%20benthic%20fauna%20habitat%20and%20ecosystem%20function.aspx

2.1 Existing data sources in New Zealand – David Bowden

Data to inform our knowledge of the distributions of benthic fauna in New Zealand waters are available from a number of sources. An overview of the range of existing data, methods of collection, and repositories is provided in Table 3. These data span many decades of collection effort for many different purposes but are predominantly from research trawl surveys, dedicated benthic sampling, and bycatch from commercial fisheries through the MPI Observer Programme. As a consequence of the extended time-span and disparate sources over which these data have been accumulated, there can be considerable variation in taxonomic resolution, representation of taxonomic groups, size ranges represented, availability of quantitative data and comparability. Caution is required, therefore, in the use and interpretation of such data when combining across surveys and regions for analyses at broad spatial scales.

The *Specify* database of the NIWA Invertebrate Collection (NIC) currently holds more than 90 000 records of benthic invertebrate fauna, each recorded with details of taxonomic identity (and identifying authority), sampling method, survey code, date of collection, depth, and geographic location. This total represents only approximately one third of the physical specimens preserved at NIC and work is on-going to register all historical specimens in *Specify*, in addition to adding new records as they are collected. Records span the entire south-west Pacific region but most are from within the New Zealand EEZ, with the highest densities of samples in areas of the major commercial fisheries (Figure 2). Because *Specify* is primarily a taxonomic database, set up to enable efficient curation of specimens for systematics research, and contains records compiled over many decades from a wide range of sources, most records do not span the whole community and are not quantitative. However, for dedicated benthic biodiversity research voyages run from RV *Tangaroa* during the past decade, including all Ocean Survey 20/20 (OS2020), Seamounts, Vulnerable Deep Sea Communities, and Vulnerable Marine Ecosystems voyages, *Specify* holds quantitative, whole-catch information. *Specify* also holds data on invertebrate bycatch from the research trawl surveys but only where physical specimens have been retained from the catch and registered at NIC. Thus, *Specify* holds a subset of the full data recorded in *Trawl db* (see below) but at finer taxonomic resolution because retained samples are identified by NIC taxonomists.

Information on benthic invertebrates is also held in the databases *trawl*, *cod* and *biods*, all hosted at NIWA. The taxonomic resolution in all these databases is variable, with clear increases in resolution for most taxonomic groups over time (for an example from the Chatham Rise trawl surveys see Figure 3), and community representation is limited because sampling is by demersal fish trawls rather than dedicated benthic gear. There are more than 100 000 records in *trawl*, including records of invertebrate bycatch weights and identifications from 1992 onwards from all research trawls. There are also more than 100 000 records in *cod*, which records information from the MPI Observer Programme. *Biods* holds more than 90 000 records of taxa and station details from biodiversity surveys, currently including all OS2020 and Seamount voyages, with most data being whole-community and quantitative. The taxonomic resolution of invertebrate data in *biods*, in most cases, should be comparable with that in *Specify* because all retained specimens are first lodged with and identified by NIC before data are submitted to *Biods*. Similarly, data on invertebrate taxa in *trawl* and *cod* will have overlap with *Specify* because all retained physical specimens from the research trawl and Observer Programmes are lodged with NIC.

Differences in sampling and processing methods have a strong influence on which components of the benthic community are represented in the available data, and on the degree to which they can be considered quantitative. Most records of benthic invertebrates in existing databases are from trawls, epibenthic sleds, grabs and corers. Trawls and sleds sample epifauna (fauna that live on or above the substrate) over relatively large seabed areas (hundreds to thousands of square metres), yield (at best) semi-quantitative data, and do not represent smaller fauna (less than about 10–50 mm, depending on gear specification) well. The highest density of trawl-sampled benthic invertebrate data is from fisheries trawls used in the research trawl programme but, as noted above, these are not well-suited for sampling benthos. Dedicated benthic trawl (mostly using beam trawl) and sled sampling have much lower coverage than fisheries trawl across the EEZ (Figure 4).

Grabs and corers, particularly the latter, sample primarily smaller macro-infauna (fauna that live within the sediments and are retained in these surveys on a 0.5 or 0.3 mm mesh, depending on survey) at smaller spatial scales (less than 1 m^2), and typically yield fully quantitative data. Sediment corers and grabs have been used extensively within the EEZ and beyond (Figure 4), but mainly for geological rather than biological sampling. Where coring for biological research (using multi-corers and box-corers) has occurred, the resulting data have often been fully quantitative for macro-infauna (and increasingly for meio-infauna), and sediment properties. For samples where biological material from sediment coring has been retained, specimens are stored at NIC and data in the *Specify* database.

Seabed camera systems are also used increasingly to gather data on benthic communities. When analysed appropriately, photographic samples can yield fully quantitative data on epifaunal distributions, including potentially useful ecological measures of spatial relationships between fauna and habitats. Various underwater camera systems (both still and video) have been deployed in New Zealand over several decades and the coverage of these is shown in Figure 4. Prior to 2006, these cameras mainly collected still photographs at varying levels of image resolution, mostly from scampi and middle depths fisheries surveys and geological surveys. From 2006, most imagery collected from deeper waters (surveys using RV *Tangaroa*) have been acquired using NIWA's Deep Towed Imaging System (DTIS, Hill 2009), while scampi surveys have continued in shallower depths. DTIS remains the primary benthic imaging system for deep-sea surveys (OS2020, commercial work, Cold Seeps, studies, and Seamount voyages; see Figure 4 for coverage) and records both still and video imagery. Deployment methods and image resolution (1080i video and 10MP stills) remained consistent up to 2016, except for the first voyage (TAN0604), which used a 5MP stills camera. DTIS imagery allows quantitative analysis of epibenthic megafauna, bioturbation marks, and substrate type but the actual data available vary depending on the analytical requirements of individual research programmes. Two broad levels of benthic faunal data are derived from DTIS deployments: real-time observations at coarse taxonomic resolution recorded during deployments, and post-voyage observation analysis at higher taxonomic resolution which are more reliably quantitative. Real-time observations, complete with full navigational data, are recorded for all DTIS deployments but post-voyage analyses are project-specific and thus vary in scope depending on the research aims. A purpose-designed database is under development at NIWA to store all data derived from seabed imagery collected using DTIS (or other comparable systems). At present, however, while OS2020 DTIS data are stored on *biods* and, together with all DTIS still images, accessible via NIWA's NIWA's Coastal and Marine Data Portal (http://www.os2020.org.nz/), most DTIS analysis data are stored on a project-by-project basis at NIWA.

A number of surveys since 2007, notably under the OS2020 programme, have employed integrated benthic survey plans in which two or more sampling methods have been used at each site to provide a more complete representation of size classes (e.g., mega-, macro-, and meiofauna) and living modes (e.g. infauna and epifauna). For example, the inaugural OS2020 voyages to the Chatham Rise (TAN 0705) and Challenger Plateau (TAN 0707) used DTIS, beam trawl, epibenthic and hyperbenthic sleds, and multi-corers to sample epifauna, infauna, hyperbentic fauna (living immediately above the seabed), sediments, and sediment oxygen demand. Data from these and other biologically-focussed OS2020 voyages are stored variously in *Specify* and *biods*, and are accessible via NIWA's Coastal and Marine Data Portal. The advantage of these integrated sampling programmes is that they allow some comparison of patterns across different components of the biotic and abiotic ecosystems e.g. between sediment, epifauna and infauanal compositions.

In their evaluation of the MEC and BOMEC, Bowden et al. (2011) concluded that "further OS2020-style surveys could be effective for expanding the scope and generality of existing marine environment classifications" and specifically, that "a single planned OS2020 sampling programme could provide the same level of spatial discrimination as the compiled historical data used in the BOMEC", even for the most data-rich area of the EEZ (Chatham Rise). In a more detailed review of data from the inaugural OS2020 project to the Chatham Rise and Challenger Plateau, Bowden & Hewitt (2012) concluded that increasing the density of sampling is likely to be the most effective way of improving the accuracy of biodiversity maps and that sampling methods that combine small spatial lag between samples with fine taxonomic resolution provide the most useful data.

Table 3: Benthic data types, mechanisms of collection and repositories, text in italics indicates a database name[6].

Type of data	Mechanism of collection	Data repository
Biology	Dedicated benthic research surveys	*Specify db, biods db,* www.os2020.org.nz
	Research trawl surveys	*trawl db*
	MPI Scientific Observer Programme	*cod db*
Sediments	Geological surveys	archived at NIWA
	Dedicated benthic research surveys	archived at NIWA
Environment	Bathymetric surveys	NIWA/LINZ/GNS
	Oceanographic models	NIWA

[6] The NIWA "Seamounts db" also holds physical and environmental data on both seamount-knoll-hill features, and hydrothermal vent locations. NIWA also hold seep and canyon inventories. For all these data types there is no specific mechanism of collection.

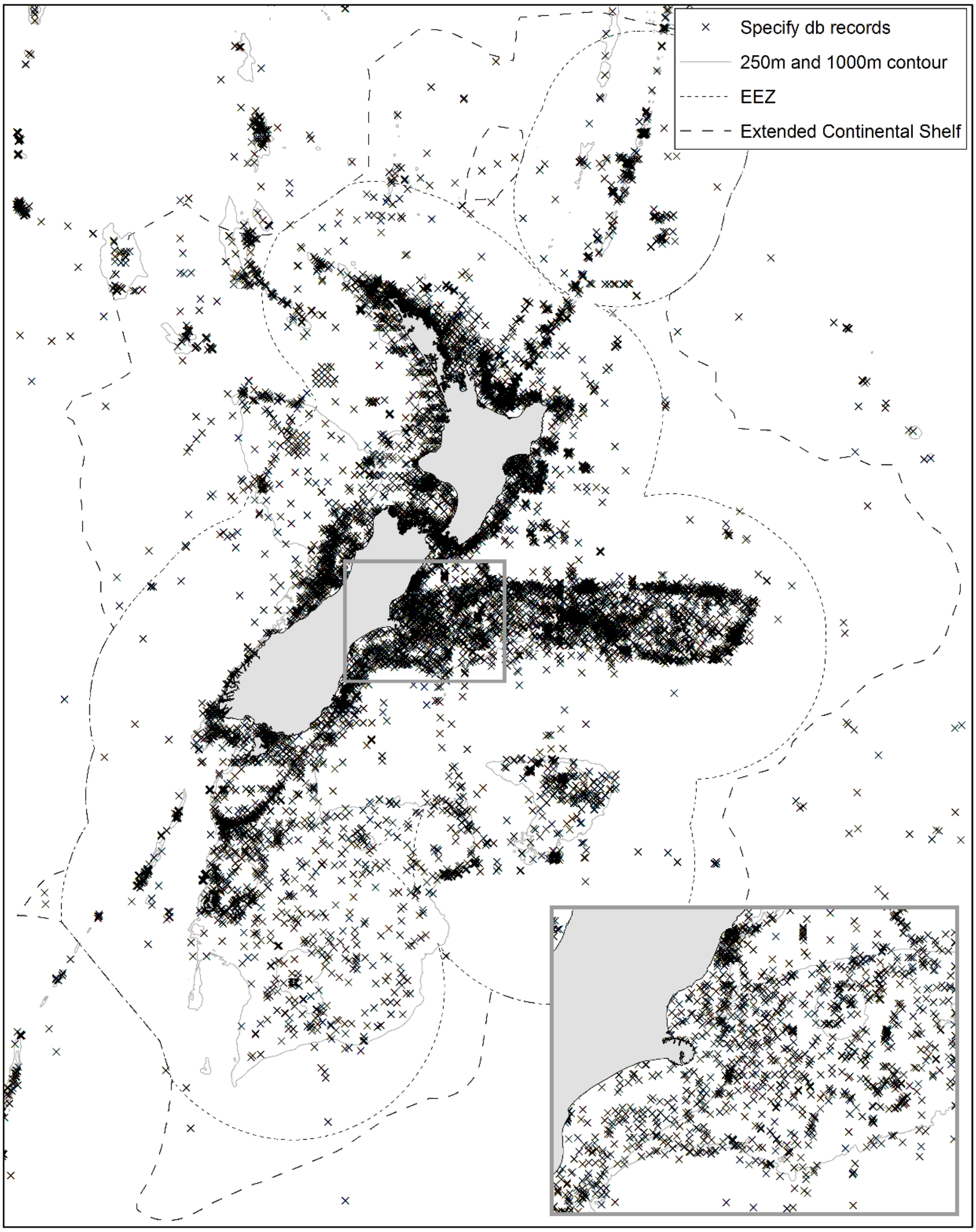


Figure 2: Locations of stations with records of benthic invertebrate fauna in the *Specify* database of the NIWA Invertebrate Collection. Inset shows the Mernoo Gap region between Banks Peninsula and Chatham Rise enlarged.

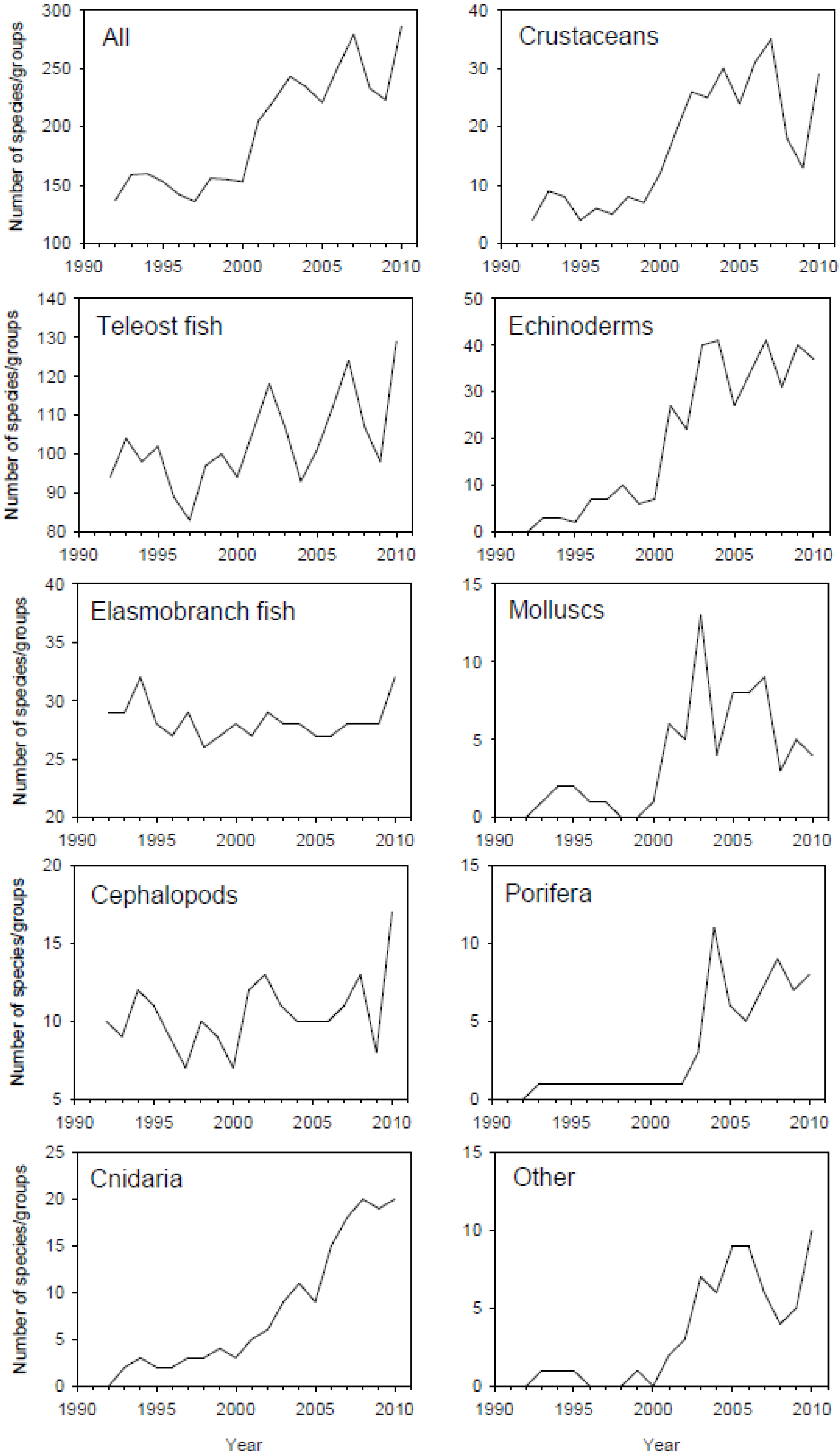


Figure 3: Number of invertebrate species or taxonomic groups identified from research trawl surveys on the Chatham Rise from 1992–2010. Data are from all stations where invertebrate catch was recorded and may include some tows outside the core survey area (O'Driscoll et al. 2011).

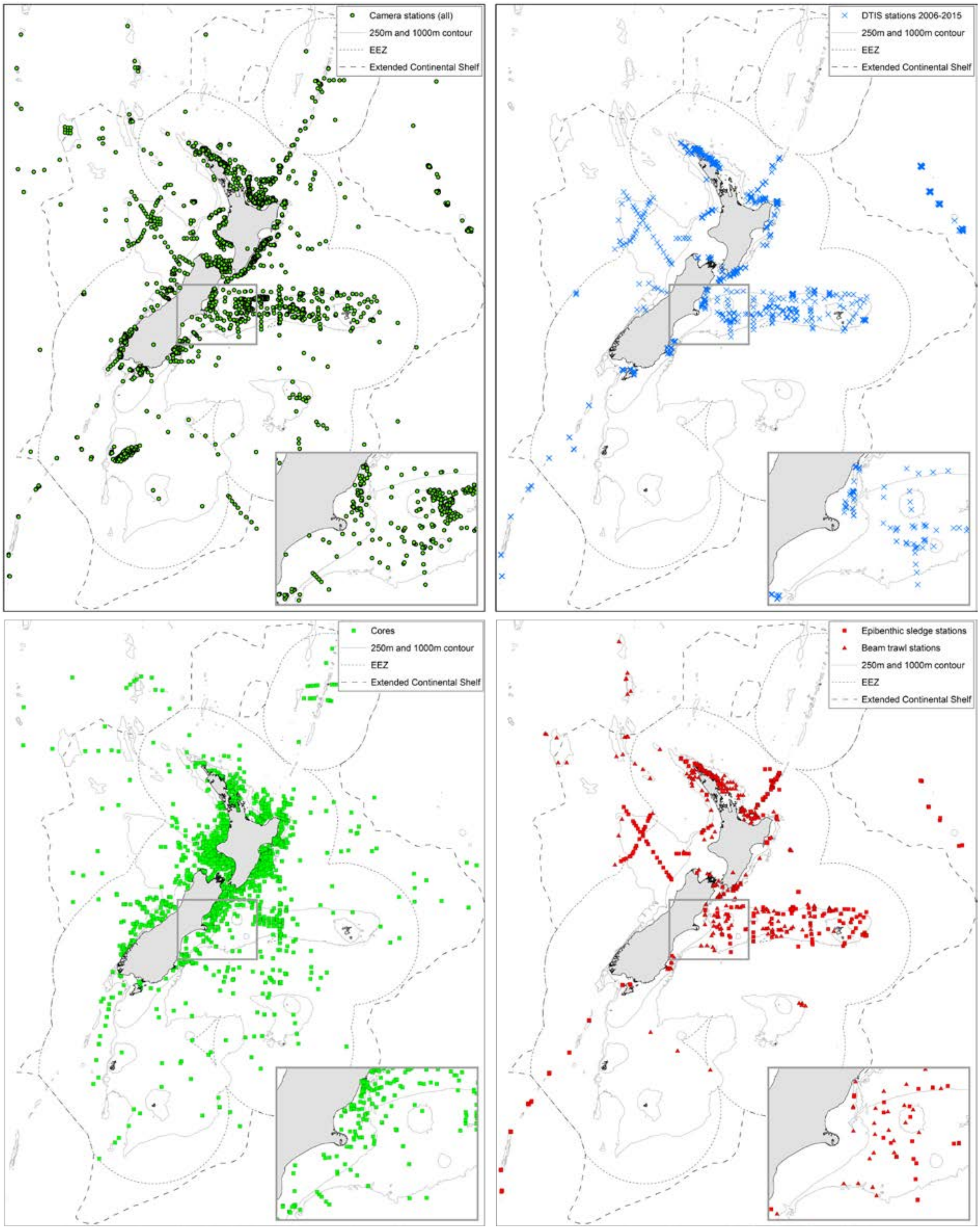


Figure 4: The distribution of data from all cameras (top left), Deep Towed Imaging System (DTIS, top right), sediment corers (bottom left) and beam trawl and epibenthic sled (bottom right).

2.2 Update on understanding of benthic impacts in New Zealand – Ian Tuck

Much of our information on trawl impacts is summarised for MPI in the Aquatic Environment and Biodiversity Annual Review (AEBAR) (MPI 2014), however, work is also ongoing in order to assess changes in seafloor communities and quantify key processes affected by disturbance from bottom fishing. This is the case for soft sediments under the project "Assessing the effects of fishing on soft sediment habitat, fauna, and processes" (BEN2007-01) and for hard substrate cold water coral communities under the project "Monitoring recovery of benthic fauna from the Graveyard complex" (BEN2014-02).

The gradient approach, in which benthic assemblages are compared across gradients of fishing pressure, is considered the most statistically powerful way to analyse the effects of trawling or dredging at the fishery scale. This analysis choice is also driven by the lack of comparable areas that are historically unfished and the availability of fishing intensity data. The gradient approach compares habitats or communities across gradients of environmental drivers and fishing pressure to determine effects. This approach has been used successfully for New Zealand fisheries (Thrush et al. 1998, Cryer et al. 2002) and other anthropogenic studies (Ellis et al. 2000, Hewitt et al. 2005), but is limited by a mismatch between the scale fishing is recorded at, and the scale biological communities respond to fishing at and co-varying drivers of benthic patterns. A useful knowledge of effort distribution can be garnered from three standard fishery reporting sources (TCER, TCEPR and dredging reporting). TCEPR (Trawl Catch Effort Processing Reports) report start and end positions of tows and have been available from the deepwater fisheries since 1990. TCER (Trawl Catch Effort Reports) report start positions of tows and have been available from the inshore fisheries since 2007. Dredge fisheries report positions by statistical area, which is not as precise as either the TCERs or TCEPRs, but when combined with fisher knowledge, implied gradients in effort can still be estimated.

The specific objectives of the MPI research project BEN2007-01 are:

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

This project aims to achieve these objectives by drawing together previous studies and information for New Zealand, and combining this with information from two case studies (Tasman Bay/Golden Bay (TBGB) and the south Canterbury bight (SCB), Figure 5) funded within this project. The Chatham Rise/Challenger Plateau OS2020 voyage data[7] will be analysed for the effects of fishing and combined with data on trawl survey invertebrate bycatch at the same location. Data collected for other purposes will also be utilised from elsewhere in New Zealand, e.g. trawl surveys and scampi photographs.

The TBGB case study has been completed, while the SCB study is ongoing. Recent fishing effort was consistently important as an explanatory factor of community structure in the TBGB study, generally explaining 15 – 20 % of the variance. Fishing patterns explained slightly more variance in epifaunal communities (community and diversity measures) than in infaunal communities in the TBGB study. The fishing effort variable, when applied to trawl survey bycatch at this location, was retained in the model and explained 9% of the variance. Infaunal and epifaunal analysis is underway for the SCB study. Trawl bycatch data has been analysed using the gradients of effort for the SCB study area and the fishing effort terms were not retained by the model, which suggests that fishing was not important as a causative factor of the observed benthic patterns at this site.

The OS2020 Chatham/Challenger Plateau voyage was not designed with fishing effort gradients in mind. Fishing effort in the OS20/20 project was either retained for various biological components (larger epifauna or survey bycatch) but explained little variance (1 – 2 %) or was not selected by the model (for the hyperbenthos). A newer project (ZBD2012-03), is looking for gradients in benthic assemblages correlated with fishing effort and attempts to detect these using multiple sampling devices over multiple spatial scales, but results are not yet available.

[7] www.os2020.org.nz

Functional trait based analysis for both the OS2020 Chatham/Challenger Plateau voyage and the TBGB analysis follow expected patterns in relation to bottom fishing pressure, i.e., that fragile, emergent and slow recovering fauna were most impacted by fishing pressure (Lundquist et al. 2013).

Previous New Zealand studies (Thrush et al. 1995, 1998, Cranfield et al. 1999, 2001, 2003, Cryer et al. 1999, 2000, 2002, Michael et al. 2006, Clark et al. 2010, 2011, Tuck et al. 2010, 2011,Williams et al. 2011,Tuck & Hewitt 2013) when looked at as a whole in combination with the analysis from this project have shown some results that are generally consistent with what has been reported from studies overseas:

- dedicated sampling of infauna and epifauna is better at detecting the effects of fishing than non-dedicated data, but non-dedicated data can provide some insight, although analysis of this could be more focussed in future;
- fishing explains between 0 and 40 % of variance in community composition in New Zealand (Tuck et al. In Prep). Notably fishing effort has also been used as a predictor in Vulnerable Marine Ecosystem (VME) work outside New Zealand waters (Anderson et al. 2016);
- epifauna tends to be more sensitive to fishing than infauna;
- recovery times are hard to quantify and are species dependent. For example, effects of fishing were evident up to a decade after fishing for long-lived fauna in Spirits Bay but are barely detectable in the south Canterbury Bight;
- fishing reduces the number of echinoderms, long-lived benthic surface dwellers, the total number of species, individuals, and the diversity of the community;
- fishing increases the proportional representation of opportunists and small individuals within the community;
- species/group sensitivity to fishing can be predicted on the basis of life history traits;
- the ecological implications of the changes to community composition from fishing are not well understood.

Ongoing studies of New Zealand seamount communities for seamount communities show that trawling has a strong impact on corals (Figure 6). Time series survey data show changes in community structure after fishing ceases (e.g., Williams et al. 2010, Clark et al. 2010), but no evidence of any stony coral “recovery” after periods of 5–10 years (Williams et al. 2010, NIWA unpublished data).

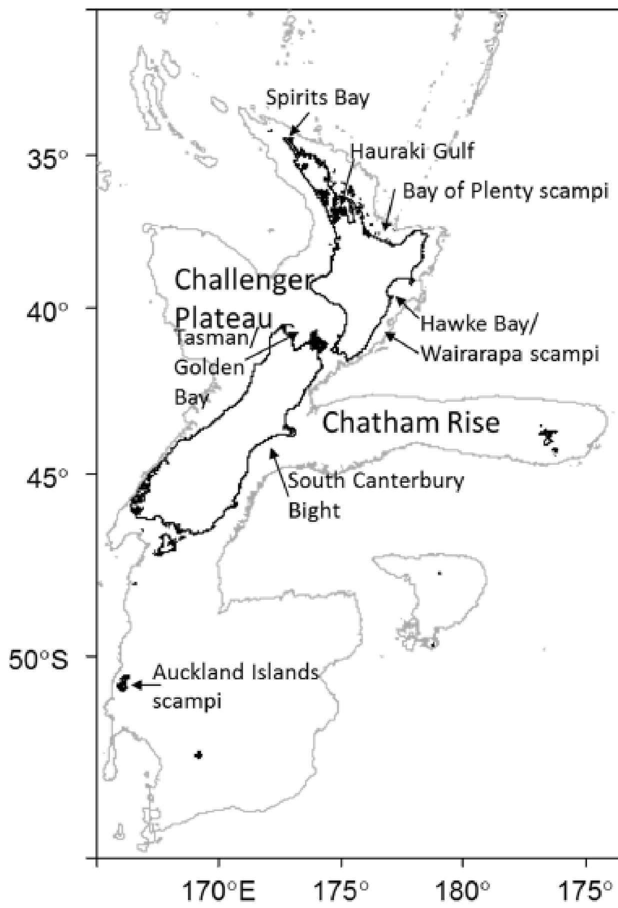


Figure 5: New Zealand locations where the impact of bottom contact trawling or dredging has been assessed. The grey line represents the 1000m depth contour (Tuck et al. In Prep).

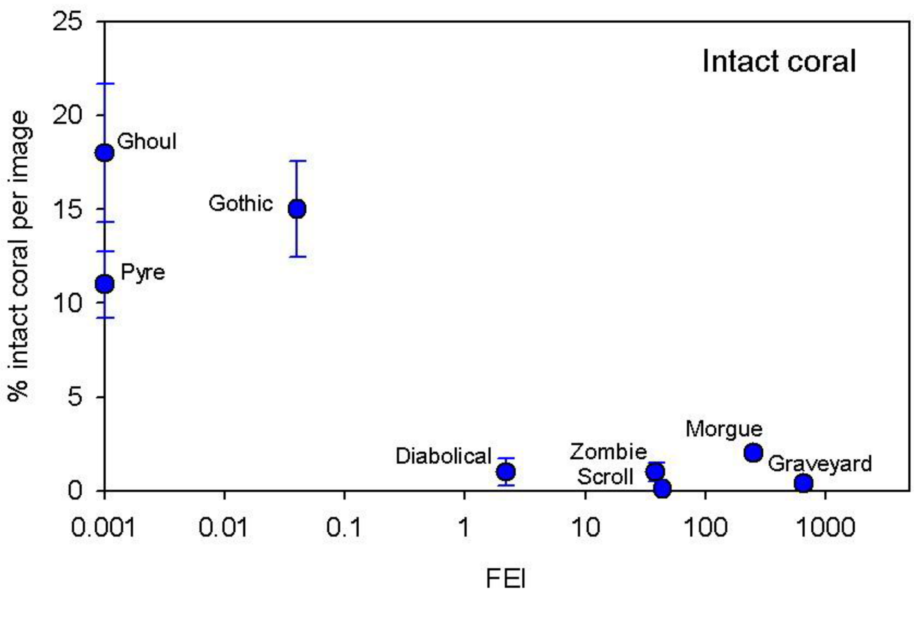


Figure 6: Intact coral correlation with a Fishing Effort Indicator (FEI) on a log scale (Clark et al. 2010).

2.3 Marine Environment Classifications (MEC) approaches - John Leathwick

Marine Environment Classifications provide a framework for the analysis of bottom fishing impacts upon benthic communities that facilitates:

- identification of biologically similar communities, independent of geographic location;
- identification of rare, distinctive or particularly sensitive biological assemblages;
- interpolation in the absence of comprehensive geographical coverage;
- and increased power of statistical analyses through efficient stratification.

In practice, however, there are a number of issues that need to be overcome before an effective marine classification can be developed for New Zealand. The quality of biological data is variable, sample numbers are low by comparison to terrestrial classifications and the numbers of samples tends to decline as depth increases. These samples often come from different sized areas, vary in taxonomic resolution, may have inherent collection biases (e.g. sessile taxa may be better sampled than mobile taxa), or may not have environmental data at matching resolutions. Although the broad ocean climate is well documented the composition of seafloor sediments (an important driver of benthic community composition), is less well known at a scale that is useful for marine environmental classifications.

There have been a number of different approaches to the challenge of classifying the biology of the marine environment, including (in order of increasing utility): rule-based Marine Environment Classification (MEC), Automated classifications, Biologically informed classifications using techniques such as General Dissimilarity Modelling (GDM) and Gradient Forests.

Rule-based MEC attempts to identify the major environmental drivers then subdivide and overlay these across the domain in question. This methodology assumes that environmental factors influence biological distributions independently of each other, and that these influences can be captured by combining classified descriptions of these factors in a simple, additive fashion. It requires subjective decisions on classification break points and weighting of factors. Automated classifications can be hierarchical, non-hierarchical or some combination of these. A simplistic application of this technique assumes that environmental distance is equivalent to biological distance, that all predictors contribute equally and that there is constant biological turnover along gradients. This technique is more statistically elegant than rule-based MEC, but contains important assumptions that are ecologically naïve. Biologically-trained classifications occur where the analytics are informed by biological data to address critical questions related to which environmental drivers to include, their relative influence and how they are transformed and combined to maximise the ability of the resulting classification to describe biological turnover.

GDM uses matrix regression to accommodate non-linear relationships between environment and species turnover, the differing influence of environmental factors and non-linear responses to individual factors. Matrix regression is also used to weight and transform predictors and classify the transformed data, but GDM still assumes additivity in predictor space. GDM has been applied to a number of New Zealand datasets including a riverine classification (trained against fish and macroinvertebrate data), the fish-based MEC (trained on research trawl data) and the Benthic-optimised MEC (BOMEC). The BOMEC involved separate GDM analyses for six biotic groups with transformations from these averaged across the predictors prior to classification. The BOMEC should be used with caution, it could be improved upon and should not be used in all circumstances (the right tool to use depends on the question being asked), but should be useful for gap identification and designing surveys to address those gaps. Gradient forests is conceptually more complex and is an extension of random forests, a machine-learning-based regression tool. It analyses a set of classification tree-based models relating variation in species composition to a related set of underlying environmental predictors, creating a set of transformations of the environmental data, prior to classification, as is done for GDM. Given its more individualistic approach to the analysis of relationships between species and the environment, it appears more capable of capturing the contributions of a full range of environmental data.

One alternate approach to ecological classification is to use environment-based prediction of species, followed by classification of the species predictions. A statistical model is first used to describe the relationship between each individual species and environmental predictors at some set of sample locations. This is then combined with spatially comprehensive environmental information to produce predictions of species occurrence or abundance across the entire geographical space. These predictions are then classified in order to predict a biological classification. Conceptually, this can be seen as directly modelling differences in species composition, rather than relying on environmental distance as a surrogate for ecological distance. It may help to illustrate this technique using a marine example. Approximately 22 000 research trawls were used to predict distribution and abundance of approximately 123 fish species using 10 environmental predictors (e.g. depth, a sediment layer) as well as trawl parameters. A 'delta-lognormal' approach was utilised with a two-step modelling process. The first step used a binomial (presence-absence) model to predict occurrence across the entire data set. The second step used a gaussian model on log (catch) in order to predict the abundance where catches occurred for that species. Using this method, predictions were made for 1.9 million 1 km^2 cells within trawlable depths in the Exclusive Economic Zone (EEZ), including interpolating to sites not sampled by trawls. These combined predictions of presence/absence and catch used trawl parameters so can be seen as providing effort-standardized catch estimates for each species. These predictions were classified at a number of class levels, the classification tree is shown in Figure 7 (at the 16 group level), and the spatial distribution of these BOMEC groups is shown in Figure 1b. Experience with a number of such classifications suggests that where comprehensive biotic data is available it will generate better predictions of variation in species composition or turnover than more indirect approaches based on some set of underlying environmental variables (Figure 8).

It was noted that, some of the classifications that have been developed have included information on fish catch as input data (fish-based MEC; BOMEC). Using these classifications as tools to evaluate fisheries impacts is likely to be inappropriate as the scale and intensity of the historic fisheries is likely to have been a factor in determining the density-distributions of the fish component used as an input to the classifications. An option for reworking the BOMEC without a fish component was discussed.

In summary, having biological data to support these models is crucial. Environmental classifications need to be tuned against biotic data, and biological classifications are inherently reliant upon these data. Where comprehensive biological data are available, biologically based classifications, although more computationally intensive, are likely to outperform environmentally driven classifications, in part reflecting their greater ability to account for interactions between environmental drivers (see Figure 8). Biologically trained gradient forests or species based approaches were recommended over techniques that have more limiting assumptions, like the additivity assumption inherent in GDM.

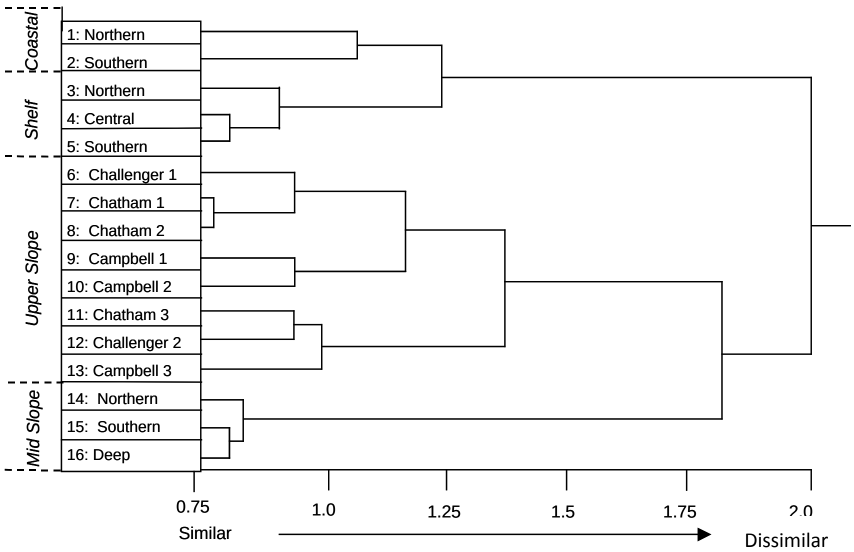


Figure 7: Dendrogram of classification results used in the fish-based MEC at the 16 class level.

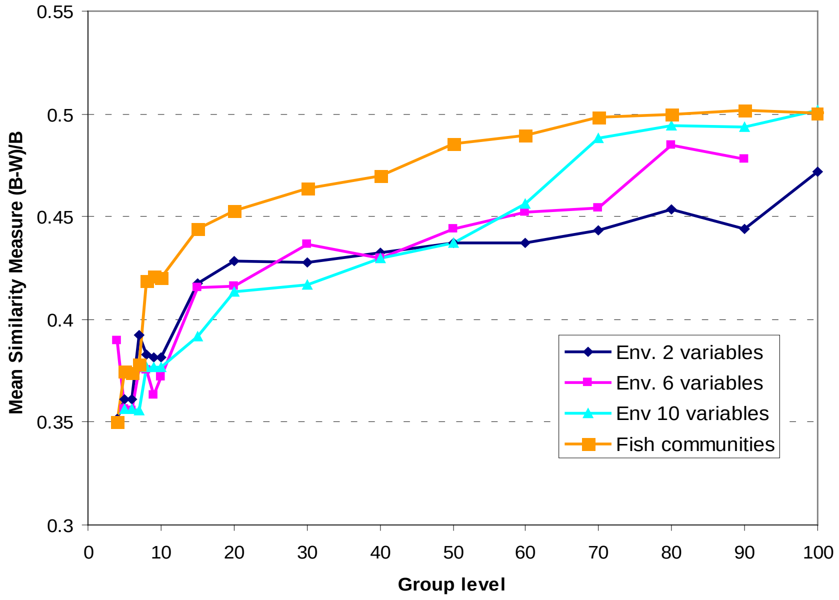


Figure 8: Mean similarity measure explained using a mix of environmental (Env.) variables or fish communities in the fish-based MEC.

2.4 Benthic risk assessment in Australia and beyond - Roland Pitcher

Two examples of the application of species predictive modelling will be discussed here, a well-studied example summarising fifteen years of research from the Great Barrier Reef (GBR) and a less-studied example from the south-eastern Australian marine region (SEMR). The Trawling Best Practices (TBP) project will also be briefly discussed.

2.4.1 Great Barrier Reef (GBR)

A series of related research studies over 15 years assessed the effects of prawn trawling on sessile megabenthos in the GBR, to support management for sustainable use in the World Heritage Area (summarized in Pitcher et al. 2016). These large scale studies:

- estimated impact rates on benthos;
- monitored subsequent recovery rates and measured natural dynamics;
- mapped the distribution of seabed habitats and megabenthos species; and
- integrated these results in a dynamic modelling framework together with spatio-temporal fishery effort data and simulated management, to estimate the regional scale time-series of status for megabenthos species.

Typical impact rates were between 5–25% per trawl, recovery rates ranged from several years to several decades. Most sessile megabenthos were however distributed in areas where trawling did not occur and so had low exposure to trawling. By simulating trawl impact and recovery on the mapped species distributions, the model estimated the regional scale cumulative changes due to trawling. The model also evaluated the expected outcomes for sessile megabenthos in response to major management interventions implemented over about a 10 year period, including closures, effort reductions and protected areas. Effort reductions made the biggest positive contribution to benthic status for all taxa, with closures making smaller contributions for some taxa.

A large amount of data were collected to support modelling of species distributions for the GBR. Trawl and epibenthic sled samples were taken across the extent of the GBR (see Figure 9 for an example – notably these samples have been identified to species). Trawling sampled fish approximately seven times better than epibenthic sleds, for crustaceans the trawls and sleds performed similarly, and for all other groups epibenthic sleds were generally superior at sampling. Camera gear observed some attached epifauna that were not sampled well by sled or trawl, and they could also observe over rough ground where sleds could not, however, the level of taxonomic identification able to be extracted from video was generally low due to poor image resolution. This highlights the need to use a range of gear if sampling across a range of biodiversity is required, or to use the correct gear to sample specific fauna if sampling is to be more selective.

Twenty-eight explanatory variables were also collated for use in species predictive modelling, these included the following variables (as well as seasonal and/or interrannual variability in these where appropriate):

Sediment type	Turbidity	Temperature	Benthic irradiance
Depth	Nutrients	Oxygen	Carbonate percentage
Current stress	Primary production (Chl *a*)	Slope	Salinity
Trawl effort			

These explanatory variables were combined with the species distributions in a two-stage generalised linear model (GLM) to generate models that allowed species distributions to be predicted along environmental gradients throughout the GBR. The first stage of this modelling was fitting a binomial logistic model of probability of presence to the observed presence/absence in trawl and sled data both separately and simultaneously. The second stage was fitting a log-normal model of biomass to the sampled biomass from trawl and sled data both separately and simultaneously. For both models, the best predictor covariates were selected from the available explanatory variables using the Bayesian Information Criterion (BIC). One key result of this prediction process was that contrasting distribution patterns were seen for species in the same genus. This demonstrates that distributions cannot always be assumed from higher taxonomy and that species-resolution is necessary for developing good biophysical prediction models.

Exposure to trawling could then be calculated, for the approximately 850 species where distributions were predicted, by multiplying the species abundance in 0.01° grid cells by the trawl swept-ratio of each grid cell then dividing by the total abundance (see Pitcher 2014) The relative catch rate (C) can then be calculated across the entire area and divided by an estimated mortality rate (M) per species. Those taxa with the highest values of the C/M ratios are those deemed most at risk. Notably M can also be used as a proxy for recovery of habitats (usually using the recovery of the slowest recovering species present, but this may need to account for functional role as well, particularly in the case of biogenic species). M can be used as a proxy for R as taxa with high M will tend to have high spawning/recruitment to deliver population replacement and this will tend to lead to shorter recovery times; taxa with low M will tend to be long-lived with low recruitment and thus will have longer recovery times. Notably there are some situations where this assumption will be erroneous. Two examples are where short lived creatures require biogenic structure to recover before they can and where large scale denudation occurs and therefore there is a delay in propagule availability (away from the edges of the denuded area) which slows recovery. Moving from mortality (P) to a recovery metric (R) is possible using impact recovery metrics from experiments. Comparing C to M within this framework is powerful as a screening device for prioritising which species are most likely to warrant management attention. Two drawbacks of this technique are that it relies on assumptions about mortality (where this is unknown) and that propagating uncertainty is also difficult using this technique.

Similarly, given that distribution information is known, species assemblages can be predicted from environmental data using techniques like regression trees. When this was done for the GBR the strongest environmental drivers were percentage mud (a breakpoint at 26.29% separated the deposit feeders from other feeding types) and depth (where different boundaries were identified depending on the sediment grain size). Other important predictors included; bottom stress; phosphate and the gravel/mud classification. Video data were analysed using a similar technique and the percentage mud was still the strongest predictor of habitats, but the data were coarser, harder to fit and missed some obvious patterns.

Negative correlations can also be examined between the predicted distributions and trawl effort. In the GBR example, 31 of the 850 species examined showed significant negative correlations with trawl effort. Management Scenario Evaluations (MSE) can be run using this information (Ellis et al. 2014; see Figure). This allows different management options to be evaluated on the basis of their impact upon response variables such as the relative biomass and density of the benthos.

2.4.2 South-eastern Australia’s Marine Region (SEMR)

The research in Australia’s South-East Marine Region (SEMR) set out to produce the first regional-scale distribution maps for benthos, and to assess the impacts of human uses and the efficacy of existing strategies for managing epibenthic fauna (see Pitcher et al. 2015).

The model incorporated predictions of biodiversity assemblages and habitat-forming benthos, and their exposure to fishing and levels of protection (derived from existing data sources). Survey data were collated for benthic species found on the continental shelf and upper slope, as well as information on their impact and recovery rates in relation to human uses. Biophysical modelling was used to characterise, predict and map patterns of biodiversity assemblages (spatially unique mixtures of all species, including mobile invertebrates and fish (Ellis et al. 2012; Pitcher et al. 2012), and the distributions and abundances of the major habitat-forming taxa such as sponges, coral, gorgonians and bryozoans. Fishing and other human activities that affect the SEMR seabed were mapped from collated data. For the fishing industry (particularly trawling) this included historical annual fishing effort by area, fishing operations and management actions (including effort reductions, closures to fishing and the Commonwealth marine reserves system). Information was also collected regarding oil and gas infrastructure.

The effects of fishing were modelled for 15 spatially unique species assemblages and 10 habitat forming benthos taxa types that had been predicted and mapped from survey data. A complex picture emerged, with patterns and responses varying spatially according to the distribution of benthos taxa types, trawling distribution, and type of management action. Had none of the management actions been implemented, benthos status was predicted to stabilise or recover slowly, and with all management actions in place, recovery was quicker. Reductions in trawl effort universally improved the status of habitat-forming benthos, with the 2006 commercial fishing licence buy-back leading to greater improvements than the 1997 commercial fishing license buy-back. In some cases, spatial management that excluded trawling also led to improved status of some benthos taxa types, particularly the deepwater fishery closures. Most fishery closures and Commonwealth Marine Reserves (CMR) had little detectable influence on status. However, there were also some cases where closures worsened the status of some taxa in some locations, because displaced trawl effort moved to areas where some taxa were more abundant.

A number of methodological lessons can be taken from the SEMR experience.

- Weighted ensemble maps were used to predict distributions from combinations of separate models because no single dataset (or model) provided complete coverage of the region.
- Many collated datasets had poor taxonomic resolution of sessile benthos (including those based on video), so data was aggregated to a higher taxonomic level in this analysis, this means the analysis is coarser than it could be for certain taxa, but at a uniform level throughout.
- Effects may differ within cells but this would not be modelled using this approach.
- This approach was unable to propagate errors from ensemble distribution models or from impacts with multiple sources (e.g. other fishing, oil and gas) but did consider cumulative impacts across sectors and over time, outputting a cumulative equilibrium status (as if the impact continued at that level).
- GLMs were not used. These can predict beyond the observed data range, so results should be interpreted cautiously.
- Random forest modelling was used. These are a bootstrap method, with each sample and model fitting holding out approximately 30% of the data from model generation for use in model cross-validation testing.
- Modelled recovery rates were used here, these did not explicitly include interspecies interactions, and therefore may differ from ecological observations of recovery.
- Exposure to the pressure is a useful first screening tool for impact upon an assemblage, then susceptibility and recovery of key species that characterise assemblages can be examined.

2.4.3 Trawling Best Practice (TBP)

The trawling best practices (TBP) project[8] established a working group of experts in ecology and fisheries management to provide a scientific basis for evaluating policies on trawling. It has five phases to:

1. compile and examine data on the area trawled, the habitats trawled and the intensity of trawling for as much of the world as possible. Particular attention will be paid to identifying data on the trends in the extent and frequency of areas trawled, and the distribution of trawl footprint across different habitat types.
2. compile and evaluate data on the impact of trawling on the abundance and diversity of biota, looking especially at the key factors of intensity of trawling and type of habitat trawled. Where possible, responses of key ecosystem services to trawl disturbance will be compiled or inferred from published studies.
3. use information from the first and second phases to develop methods for benthic risk assessment and conduct a benthic risk assessment of the effects of trawling and illustrate trends in risk of change to seabed habitats and communities among fisheries both spatially and temporally.
4. look at the medium- and long-term impacts of trawling on the productivity and sustainable yield of different target species and the ecosystem. It seems likely that trawling benefits some species and is detrimental to others. How does trawling affect the long-term sustainable yield of aquatic resources from an ecosystem? How does trawling affect other ecosystem services?
5. identify and test a range of management options and industry practices that may improve the environmental performance of trawl fisheries; with a view to defining ‘best practice’. For each option or practice, the impact on biota, sustainable food production, ecosystems and ecosystem services will be evaluated, along with changes in fuel consumption and other costs and impacts.

Some of the detail of this project was discussed, particularly in relation to the third phase (risk assessment) and points from the TBP project relevant to the New Zealand situation are discussed here.

- A range of recovery metrics (average and others) will be used.
- Sediments will be included as a continuous variable to compare against biological complexity, with sediment being lumped into categories as late as possible in the analysis.
- The form of impact assessment used here is to estimate equilibrium status assuming that the current level of effort is applied indefinitely; thus approximating cumulative effects over time.
- The project will conduct analyses at approximately 1 km grid scale, and is aware of sub-grid scale effects, where assessment of trawling hotspots would suggest that no fragile taxa would exist, whereas in reality, such taxa may persist at smaller scales.
- The meta-analysis generalizes impact and recovery rate parameters over multiple studies – due to the lack of biological data – limiting how precise advice to managers can be. Therefore, advice about relative benefits of different management options is likely to be our best approach.
- Where multiple sectors overlap, cumulative impacts across sectors will be approximated by combining effort and impact parameters via logistic equations.
- One of the outcomes of the risk assessment will be to highlight data gaps, enabling better target data collection.
- The sedimentary habitat-level risk assessment will, where data are available, be extended to groups of species within benthos classes. This will include prediction of distribution patterns using methods similar to those used for the GBR and SEMR described above.

[8] Officially known as “Trawling: finding common ground on the scientific knowledge regarding best practices” see https://trawlingpractices.wordpress.com/ for more information.

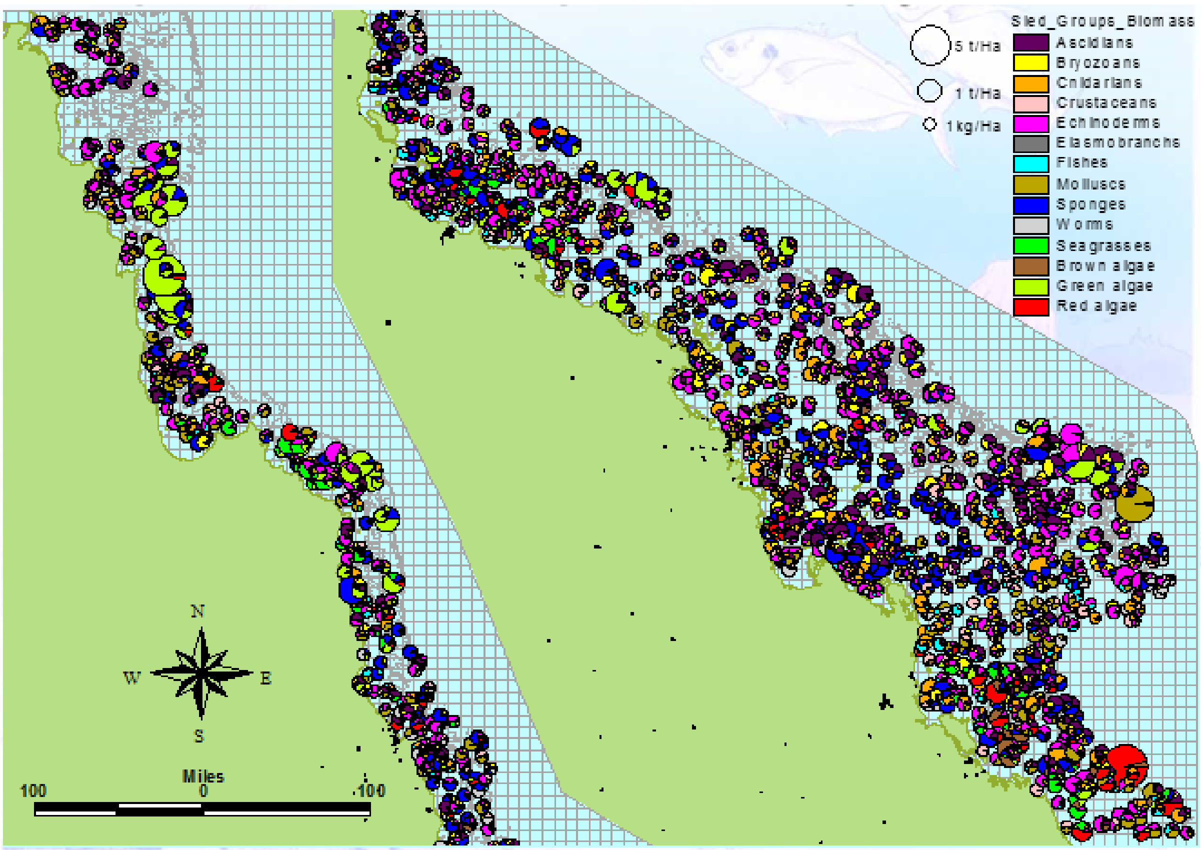


Figure 9: Epibenthos Sled sample biomass from the GBR at the phylum level

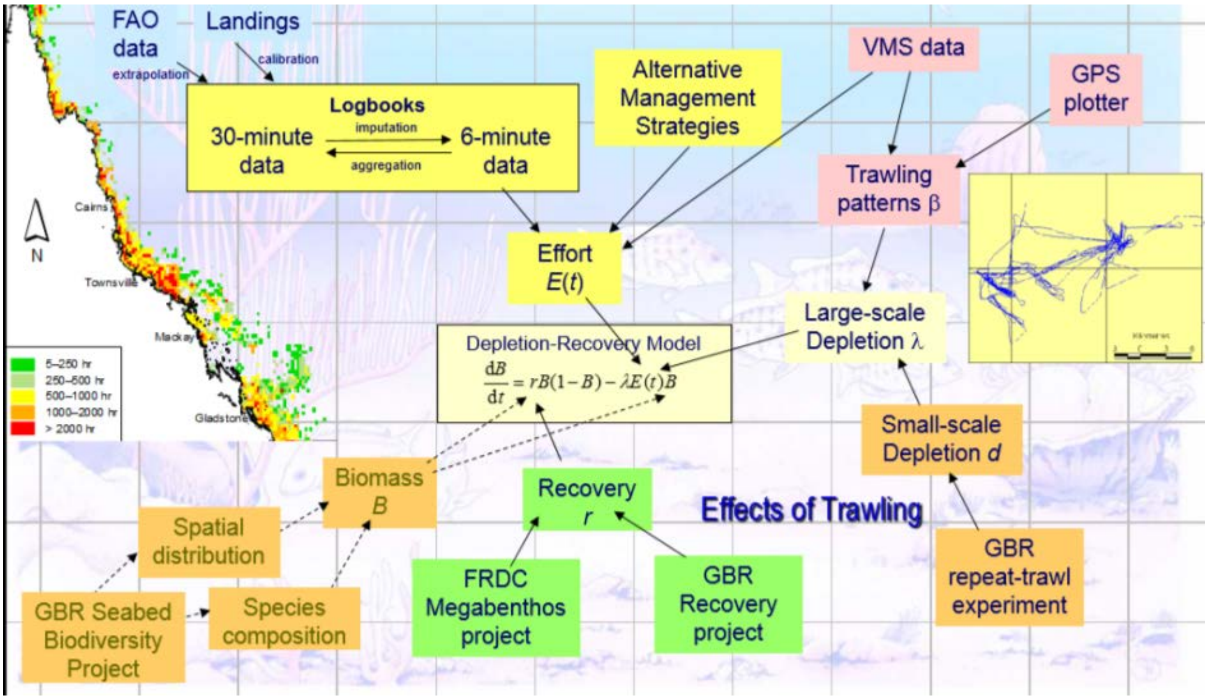


Figure 10: Diagram of research from the GBR over fifteen years and how it was combined to inform the depletion-recovery model, including a MSE component.

2.5 Species distribution modelling in New Zealand - Ashley Rowden

There have been a number of applications of species distribution (or habitat suitability) modelling that are relevant to New Zealand from global to more local scales. At a global scale these include predicted distributions of deepwater stony corals (Clark et al. 2006, Tittensor et al. 2009, Davies & Guinotte 2011), and octocorals (Yesson et al. 2012) that are based on a significant amount of data from the New Zealand region. Ocean-scale species distribution models that include a relatively large amount of data from the New Zealand region have also been produced to identify biogeographic regions (O'Hara et al. 2011), and to develop spatial management options for bottom fisheries with respect to protecting vulnerable marine ecosystems (Rowden et al. 2015). At a local scale these model applications include boosted regression tree (BRT) predictions for framework forming deep-sea corals (Tracey et al. 2011a) as well as models for selected taxa (e.g. protected taxa) or groups of taxa (i.e. benthic assemblages/communities or species richness) (Leathwick et al. 2006a, Baird et al. 2012, Compton et al. 2013, Rowden et al. 2013, Rowden et al. 2014, Anderson et al. 2014, Leduc et al. 2015). These models combine usually sparse occurrence data with environmental variables to predict distribution using environmental variables across areas where sampling does not exist (Figure 9). These predictions have been used to assess overlaps of predicted distribution with forms of benthic protection and fishing effort (Baird et al. 2012, Wood et al. 2013) (Figure 12) or proposed mining activities (Rowden et al. 2014), construct classifications (Leathwick et al. 2006a), and inform ecological risk assessments (Clark et al. 2014).

This modelling approach has a number of strengths and weaknesses.

- The conceptual simplicity of this type of modelling encourages uptake for management purposes (e.g. to assess the overlap with different management regimes or trawling footprints, guide monitoring strategies, and identify data gaps) but it can be difficult to convey the limitations of models.
- This modelling requires little data (as a rule of thumb a minimum of 10 observations per fitted parameter) and can use presence only data (which is sometimes the only data available for the deep sea), but models using absence and/or abundance data perform better.
- The management value of models made for higher level taxonomic groupings is questionable (i.e. models produced are often for very large environmental niches).
- Model performance can be good, but uncertainty can also be high.
- Model performance and predictions vary by modelling method.
- Some modelling techniques are prone to over-fitting of data and can thus provide 'unrealistic' predictions.
- The lack of post-model field validation (only recently completed for South Pacific scale models of vulnerable marine ecosystem (VME) indicator taxa, Anderson et al. 2016) affects the 'confidence' in use of species predictive models for management by stakeholders.
- All models are limited by the type and quality of data available.

New Zealand data for input to these models is often spatially and temporally biased, lacking true absence data or abundance/biomass data and there is often limited environmental data or a mismatch between the scale of the biological data and the environmental data. Predictions of distribution are then made at a common scale, so this raises a number of questions:

- Given the mismatch in scales of data, or absence of all potential suitable predictors, how useful are the predictions?
- How accurate are predictions in less sampled areas (usually deeper areas and also less fished areas)?
- Will the temporal bias allow predictions about more recent impacts, (e.g. fishing) as the baseline data may already reflect an impacted state?

Some possible advances in this area were identified, but also a number of unresolved issues. Compiling new faunal datasets and/or updating environmental data layers and making them easily available could stimulate modelling, or make new modelling more reliable. The group noted that plotting actual presences on predictive maps was considered good practice, but not always achieved. Species distribution or habitat suitability maps are not generally considered useful for rare species, but the GBR example did show that species richness predictions were highly correlated with the number of rare species. In recent times in New Zealand there has been an emphasis on BRT modelling approaches, but other approaches may be better and multi-model or ensemble model approaches may be more useful for management purposes (Oppel et al. 2012). It is not known how much data is needed to make reliable models, and at what spatial scale such data should be, for each environmental gradient of interest. Furthermore, systematic filling of data gaps is difficult due to the fragmented nature of funding, particularly for deep-sea sampling. The questions were also raised of how the following factors can best be incorporated into the modelling:

- confidence and uncertainty
- connectivity/dispersion data
- species interactions.

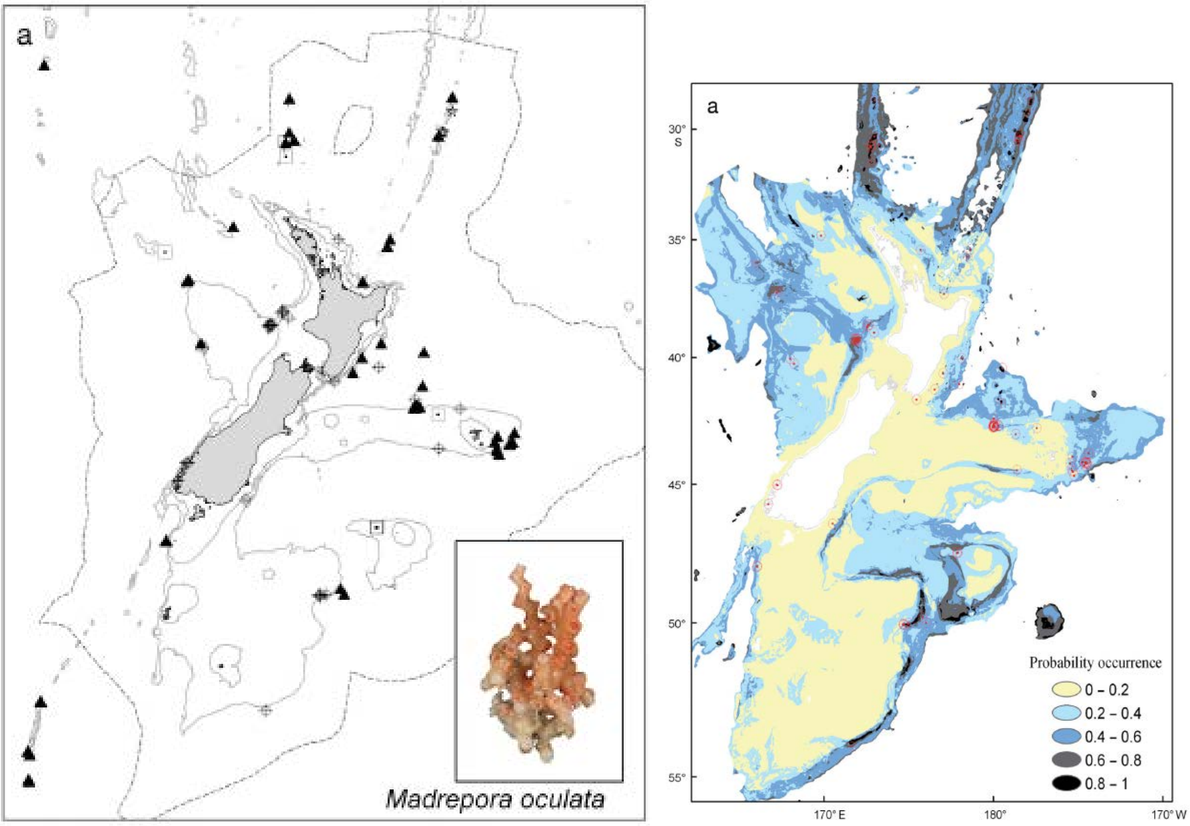


Figure 9: Distribution records and subsequent prediction of probability of occurrence predictions from BRT presence-pseudoabsence models for 1 species of habitat forming stony coral (Tracey et al. 2011a).

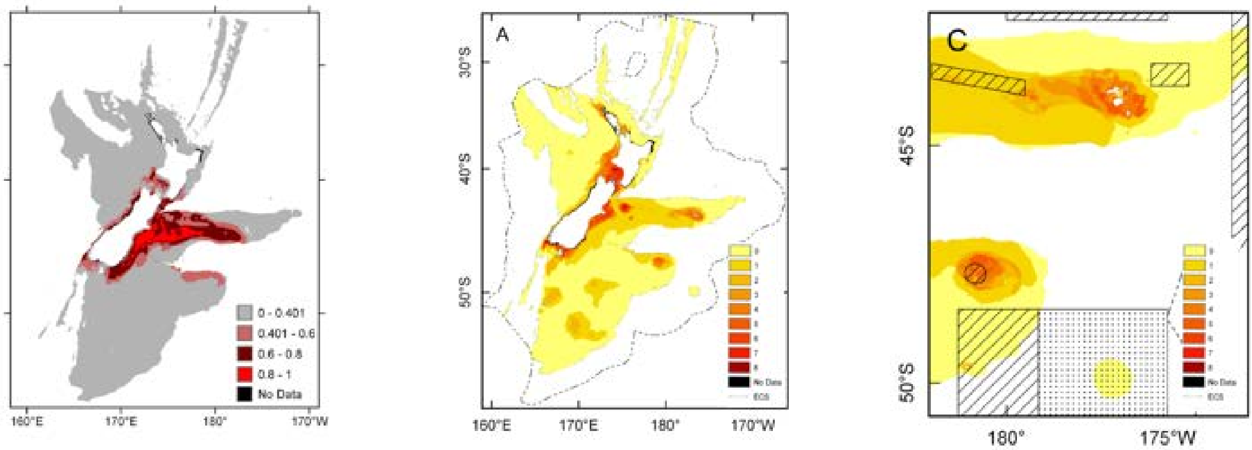


Figure 10: Predictions of *Celleporina grandis* distribution (left), hotspots (middle) and hotspots in relation to coverage of Marine Protected Areas (right) (Wood et al. 2013).

2.6 Spatial priority ranking software approaches - John Leathwick

Spatial prioritisation software allows spatial planning goals to be objectively maximised given various layers of GIS information describing some set of biodiversity features (e.g., species, habitats, etc.). In New Zealand, these types of software have mainly been used for biodiversity protection planning, reflecting their ability to identify subsets of sites that maximise the representation of a full range of biodiversity features. These programs can include costs to identify the most cost-effective solution and can consider the difference that landscape retention (preserving large tracts as opposed to smaller ones) will make to the optimal solution.

This presentation focused on the two main programs used in this area *Zonation* and *Marxan*, these were also the ones John Leathwick has experience with. The key difference between these two programs is that *Marxan* is target based, whereas *Zonation* produces continuous rankings. That means that *Marxan* requires specification of fixed targets for biodiversity features (e.g. preservation of X% of each habitat), whilst *Zonation* can be implemented at varying levels and covers the entire area of interest. In New Zealand, *Zonation* in has been used for approximately eight years in marine, terrestrial, freshwater and historic settings. The earliest of these was a marine analysis comparing benthic protection areas (BPAs) with those that would be optimal for biodiversity protection (Leathwick et al. 2006b). The Department of Conservation (DOC) is currently using *Zonation* to prioritise biodiversity management, it is also being used by several regional councils and was used in the Chatham Rock Phosphate Environmental Protection Authority (EPA) application[9].

Zonation uses gridded layers to describe the distribution of biodiversity features, these features can be species and/or classification groups and these layers can be weighted to indicate their relative importance. Cells are then removed using a backwards stepwise procedure, eliminating the cells with the lowest marginal contribution at each selection step. The mapped output then indicates the relative importance of various locations within the landscape, as a continuous, nested ranking, i.e., the top 10% of sites is contained within the top 20% of sites, and these in turn are nested within the top 30% of sites, etc. Associated tabular outputs summarise the representation of various features that would be achieved given different degrees of implementation.

A number of options are available to implement greater realism within the application of *Zonation.* Condition layers can be used to indicate less impacted sites and these can be preferentially retained. Aggregation options are available (using planning units, smoothing or adjacency options) to produce a more coarse-grained (i.e. larger protected areas) solution. Multi-criteria prioritisation can be implemented to identify trade-offs between conflicting goals, e.g. to maximise biodiversity protection whilst minimising the impact upon current fishing effort. Mask layers can be utilised to impose a particular order of removal, this allows assessment of existing or planned reserves. Retention layers can also be used to predict the expected fate of biodiversity with or without management. Point records can also be used for rare species, as opposed to the gridded layers normally utilised. Regional sub-units can also be used to allow for local geographic considerations, e.g. the desire for greater protection closer to population centres.

A deliberate approach is required to these analyses, but when these types of software are utilised they can deliver substantial benefits, but with appropriate caveats. Adding layers in a stepwise fashion, with realistic but gradually increasing weights, allows trade-offs to be explored between potentially conflicting goals, e.g., biodiversity protection versus fishing. In the BPA analysis, a rearrangement of the protected areas resulted in 2.5 times the benefit of the present BPAs being realised without affecting fishing. The caveats that need to apply to these outputs are:

- The outputs are critically dependant on the quality of the input data (although consideration can be made for uncertainty in the data layers).
- These techniques work best for sessile species as these have static distributions as adults (although they may better for adults than juveniles, which may be more mobile).

9 http://www.epa.govt.nz/EEZ/EEZ000006/EEZ000006_19AppEvidence_Rowden_Ashley_Benthic_communities_and_spatial_planning.pdf

- For mobile species the results are less reliable, particularly where harvesting targets spawning aggregations. These techniques can include seasonal distributions, but there are not usually data to support these for most species - recent developments in the application of *Zonation* have made substantial progress in the consideration of mobility effects, but have not yet been implemented in New Zealand.

In summary, this type of spatial prioritisation software is a useful decision support tool for identifying sites for protection, but needs expert implementation to fully capture biological complexities. It is very useful as an objective means of identifying possibilities and evaluating options for protection.

2.7 Direct modelling of consequences - Alistair Dunn

Model-based approaches have some potential advantages for assessing benthic risk. They are quantitative and typically employ Management Strategy Evaluation (MSE) frameworks, which are generally based on a range of underlying model frameworks (at differing levels of complexity). They allow explicit examination of management objectives, outcomes and trade-offs, whilst using feedback loops to determine management actions. They supply relative (rather than absolute) results allowing comparison of scenarios, but not evaluation of absolute consequences. This presentation will give an example of how this has been applied in a simple scenario, there are similar approaches elsewhere in the literature (Ellis et al. 2008, Fujioka 2006, Gribble 2003, Hiddink et al. 2006a, Hiddink et al. 2006b, Lundquist et al. 2013).

The general modelling approach applied is outlined below. The model assumed a discrete number of cells that did not interact[10]. Each cell was initialised (or seeded) with biological components representing a benthic community or population. A series of processes were then applied to those populations:

- *Production* (growth and recruitment)
- *Mortality* (natural)
- *Impact* (anthropogenic mortality from bottom impact gear).

Simulations were investigated that had different levels of *impact*, *production* (based on maximum age and a population model) and potential to include interacting effects, larval dispersal, etc. This allows the model to assess how possible management actions change outcomes under different assumptions.

This modelling approach was then applied to two different case studies Dunn et al. (2010) and Mormede & Dunn (2013), which were different areas on the Chatham Rise (Figure 11). Both of these areas were in BOMEC class K, so it is assumed they have similar habitats, but they differed in terms of their fishing effort (the eastern site had “high” effort and the western site “mixed” effort). Two types of communities were simulated at each site, with a maximum age of 50 or 500 years. Then two types of fishing mortality were added with 50 or 80% mortality with each ‘pass’ of the bottom gear. Therefore, eight different simulations were run in all (2 sites × 2 community ages × 2 levels of fishing mortality).

The model was implemented with set parameters for scale, populations, fishing mortality and management scenarios. The spatial scale of each site was 4200 km^2, over which a grid of 168 squares of 25 km^2 was superimposed; results were then summed across each area. In this case study, communities were assumed to be uniform everywhere. This was acknowledged as unrealistic, but the next step would be to introduce more realistic communities. Notably, this may need to occur in a separate model (at a finer scale), and the results from that input into the scenario modelling, so that it is computationally practical. Formal MSE approaches were then applied that used feedback loops to update knowledge, and determine management actions. A simple population model was then applied that assumed biological productivity parameters (*M* assumed from maximum age, etc.). It was assumed that these parameters were representative of the populations at the sites. Mortality due to fishing was applied yearly as an instantaneous event and was a factor of:

- the amount trawled (percentage of seabed area trawled from Baird et al. (2011))
- how much it was trawled (cumulative area trawled each year (from Baird et al. (2011))
- the impact of the trawls (arbitrarily chosen here as 50% or 80% - this could potentially be derived from experimental or other studies).

[10] Non-interaction was not a requirement of the model but was assumed for this illustrative case.

Management scenarios were simulated for either 50 or 500 years and were:

1. 1990–2011 with actual fisheries effort;
2. continued fishing effort as 2011;
3. closure of 40% of the area to fishing with displacement of effort;
4. cessation of fishing.

This framework enables relative performance of different management approaches to be evaluated. For example a 40% closure in the high fishing effort area displaced 24% of effort (Figure 12), as opposed to the same closure in the mixed fishing effort area which displaced 3% of effort (Figure 13). This framework is fast and simple to implement and understand, allows scenario exploration and quantified comparisons of outcomes, but makes strong assumptions of the parameters and functional relationships.

Notably, there are a number of improvements that could be implemented within this modelling framework, for example making cells non-independent or communities heterogeneous.

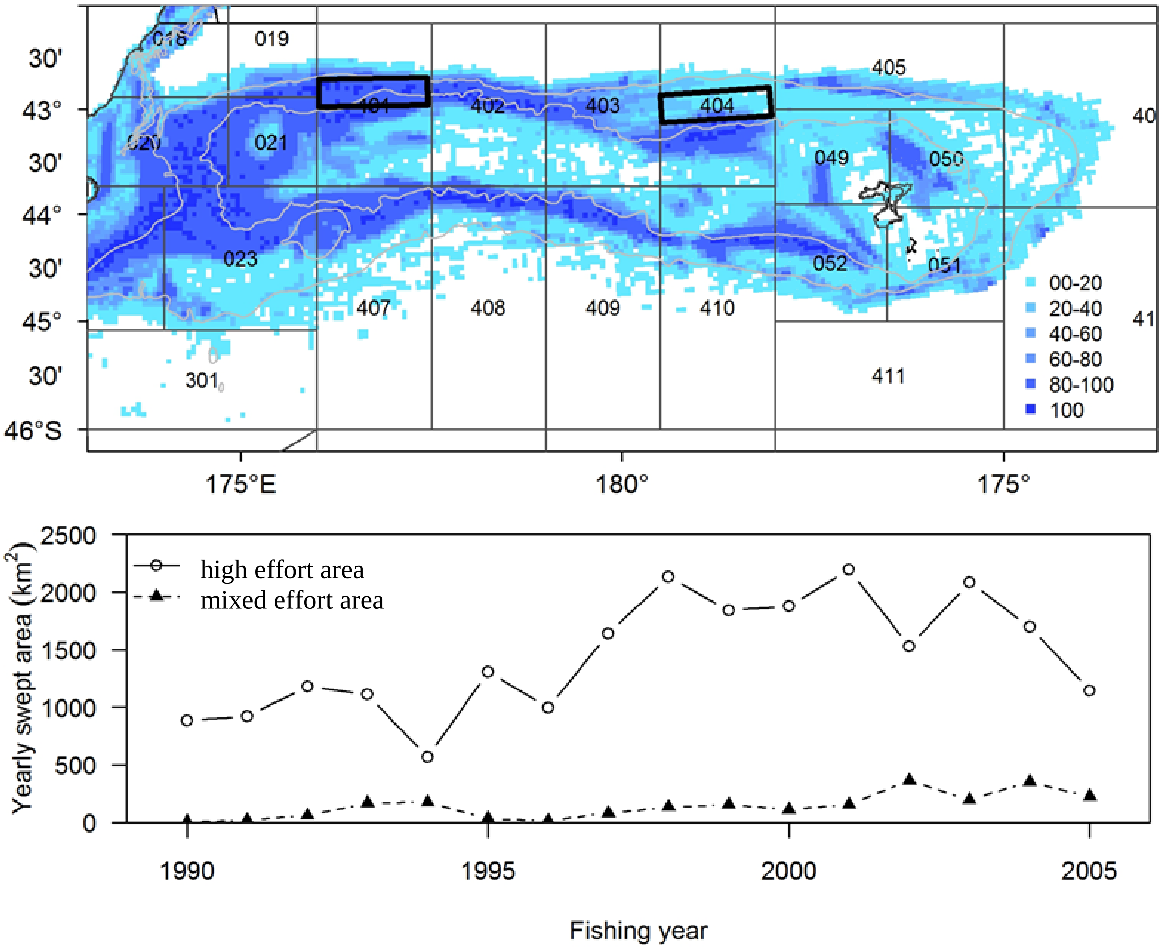


Figure 11: The two case-study areas for application of the model on the Chatham Rise outlined by rectangles (top panel) and the level of fisheries effort within each of these (bottom panel). The darker the shade of blue (in the top panel) the higher the potential fisheries impact, therefore the high swept-area area on the bottom panel represents the boxed area to the East on the top panel. Numbers on the top panel indicate fisheries statistical reporting areas (Mormede & Dunn 2013).

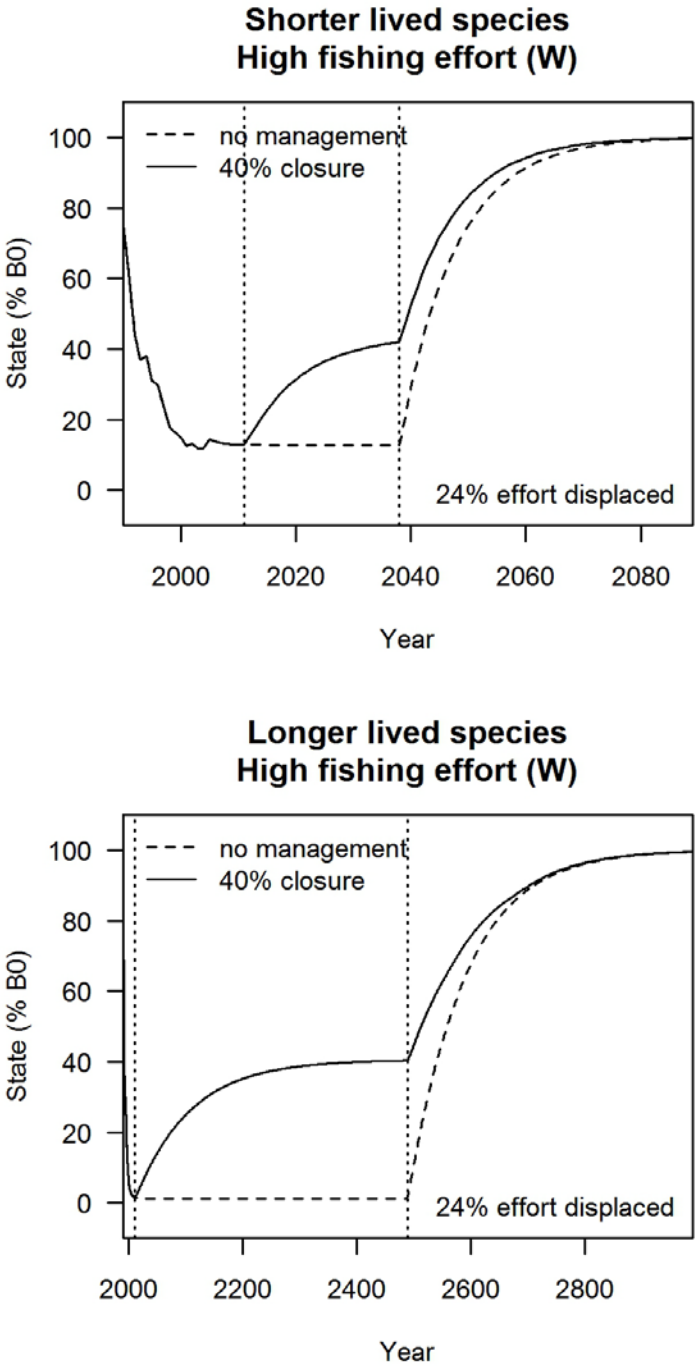


Figure 12: Graphics of relative response of modelled species state (in terms of percentage of initial biomass ($\%B_0$)) in the high fishing effort area to a 40% area closure (implemented between the dashed vertical lines) for two species with different life spans.

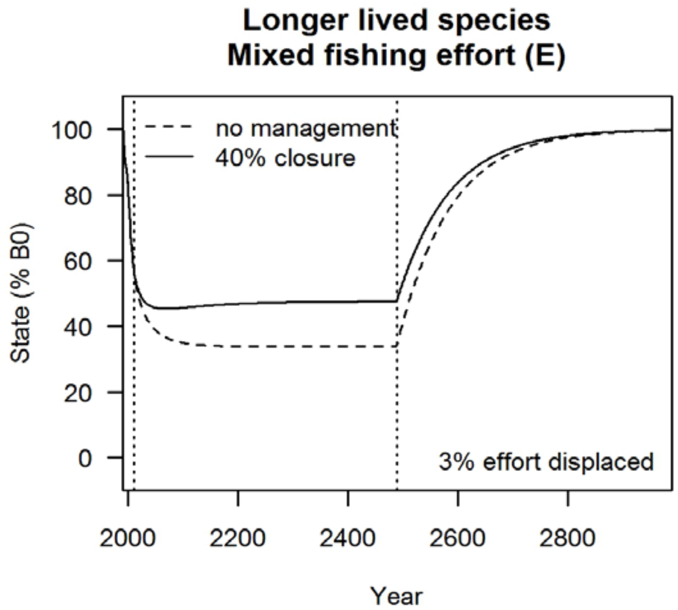


Figure 13: Graphics of relative response of modelled species state (in terms of percentage of initial biomass (%B_0)) in the mixed fishing effort area to a 40% area closure (implemented between the dashed vertical lines) for a longer lived species.

2.8 Science to support managing for resilience - Simon Thrush

This presentation focuses on ecosystem management and thresholds for ecological systems, this informs when activities should be considered high risk within ecological systems.

Effective resource management is complex due to multiple societal-ecosystem interactions. A good example is in the Hauraki Gulf where management interventions have been relatively minor, there is a lack of clear and universally agreed environmental goals, and there are key gaps in management response, implementation, fragmentation of management and roadblocks to effective management (Hauraki Gulf Forum 2011). Ecosystem-based management (EBM) is the current approach favoured to try to best incorporate the complex of societal-ecosystem interactions. Many different definitions of ecosystem-based management (EBM) exist, but in general, EBM tries to bring multiple stakeholders together to understand different viewpoints and explore the full range of benefits that ecosystems can provide and the trade-offs between these benefits inherent in management. Common themes from studies of EBM are listed in Figure 14. EBM includes explicit recognition of ecosystem services that may not immediately be obvious (e.g. nutrient cycling as opposed to fish for food), yet may underpin our use and enjoyment of the area and be impacted by management and different resource users.

Ecosystems provide a range of services, e.g. food, nutrient cycling, recreational opportunities, waste disposal. Ecosystems respond to change from a wide range of both natural and human activity; response prediction, away from catastrophic endpoints, is especially difficult in the 'real world' where there are often multiple interacting chronic and potentially cumulative stressors. Diversity is important in ecosystems, and ecosystems change over time. Ecosystem processes and structural components interact with varying strengths across a range of spatial and temporal scales to determine system responses. Ecosystems have thresholds and when they are approached variability in function is sometimes seen (Figure 15), although empirical support for this is scarce (but see: Litzow et al. 2008; Lindegren et al. 2012; Hewitt & Thrush 2010). Notably, the same indicators will not necessarily show variability prior to thresholds in different systems, emphasising the need for an ensemble of indicators (Dakos et al. 2015). When ecological thresholds are crossed, re-crossing these can be difficult (Holling 1973; 1996; 2001; Scheffer et al. 2001; Scheffer et al. 2012; Carpenter 2013). In addition, ecosystems should not be considered in isolation, their boundaries are 'leaky', and this leakiness may change spatially and temporally. The capacity of an ecosystem to resist system change is known as its resilience. But the extent of resilience is likely to change over time, due to the changing ecosystem components and processes, and in combination with changing stressors this can lead to ecological thresholds being crossed unexpectedly and ecosystems changing into different stable states (Figure 15, Figure 16).

Disturbance can be an important structuring force in benthic marine communities. Seafloor disturbance (such as trawling) has important implications for the dynamics of patches and landscapes. The time-scale of recovery for even simple benthic communities can be longer than one year in some circumstances and initial and subsequent disturbance events in the same place may not have the same effects. Far field effects are important, two examples of this are benthic impacts from elevated suspended sediment concentrations and lost fishing gear. Multiple resource users may also affect the seafloor's disturbance regime in different ways, e.g. fishing may have a more dispersed and temporally sporadic footprint in an area than mining which may be more localised and frequent.

Ecosystem management is made difficult in New Zealand by (amongst other things) our limited information base. Ideally ecosystem time series would be available, like the joint IMR/PINRO ecosystem surveys[11] of the entire Barents Sea. These types of surveys become particularly valuable when attempting to interpret system change. In the absence of good monitoring data, we must be especially focused on thinking about ensuring the sustainability of ecosystems and their ecosystem services. This means managing for surprise. Even given data like that collected in the Barents Sea, we should still be asking whether this data is sufficient to answer the questions we have, or if we need new studies or novel ways of addressing questions for this new (EBM) approach to management. This is particularly so as many ecosystem models are data intensive, poorly verified and limited in scope. An example of a novel approach of generating data for ecosystem management are the maps of ecosystem services for the Hauraki Gulf compiled using an ecosystem principles approach (Figure 17). Adaptive management is also consistent with EBM principles, but may take a change in mind-set or mandate of management agencies as the current management arrangements are usually slow and fragmented.

The 'Sustainable Seas[12]' National Science Challenge (SSNSC) is New Zealand's approach to implementing marine EBM. This will provide a strategy for the integrated management of natural resources that recognises the full array of interactions within the ecosystem, including human, and promotes both sustainable use and conservation in an equitable way. It will:

- assess the cultural, economic and environmental values of our oceans and coasts;
- investigate and describe the impacts of natural and human stresses on marine ecosystems;
- identify options for environmental mitigation or restoration; and
- overcome impediments to enhanced resource use.

It aims to do this using transformational projects in five programme areas:

1. **Our Seas** - new and effective ways of engaging with, and embedding knowledge in, our society
2. **Valuable Seas** - defines the value of our marine environment, and fosters the connections between multiple societal values.
3. **Tangaroa** – incorporation of Māori values into the management and governance regimes for our seas.
4. **Dynamic Seas** – incorporation of the wider ecosystem (including services) and the full range of impacts in current New Zealand management models.
5. **Managed Seas** - bringing all of the EBM components together into integrated decision support frameworks.

A focal area from North Taranaki across to the Chatham Islands has been defined as the main (but not the only) research area for the SSNSC[13] (Figure 18).

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[11] http://brage.bibsys.no/xmlui/handle/11250/106622
[12] http://sustainableseaschallenge.co.nz/
[13] http://www.sustainableseaschallenge.co.nz/sites/default/files/Sustainable%20Seas%20Research%20Plan%20July%2019%202015.pdf

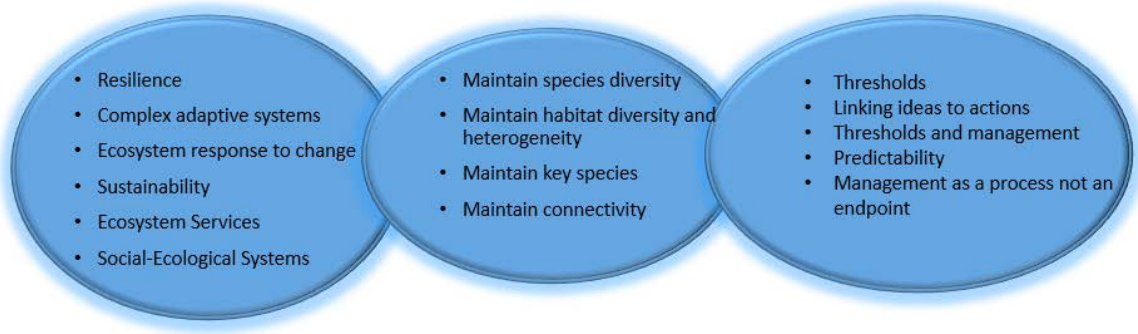

Figure 14: Common themes from studies on Ecosystem-based management (EBM) from Levin & Lubchecko 2008 (left bubble), Foley et al. 2010 (middle bubble) and Samhouri et al. 2010 (right bubble).

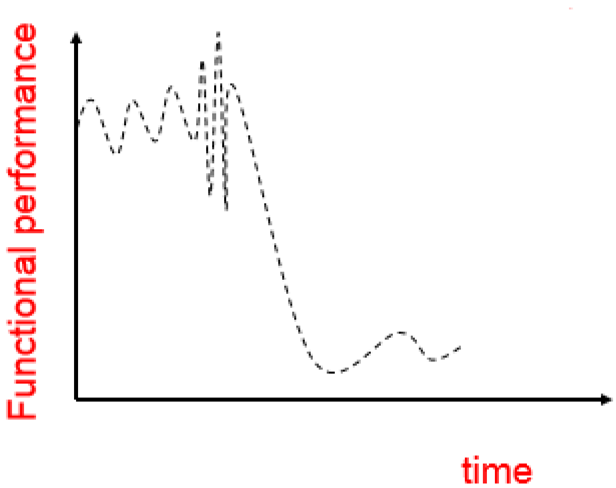

Figure 15. Conceptual diagram of the impact of a temporally sustained stressor on the function of an ecosystem (Thrush et al. 2009).

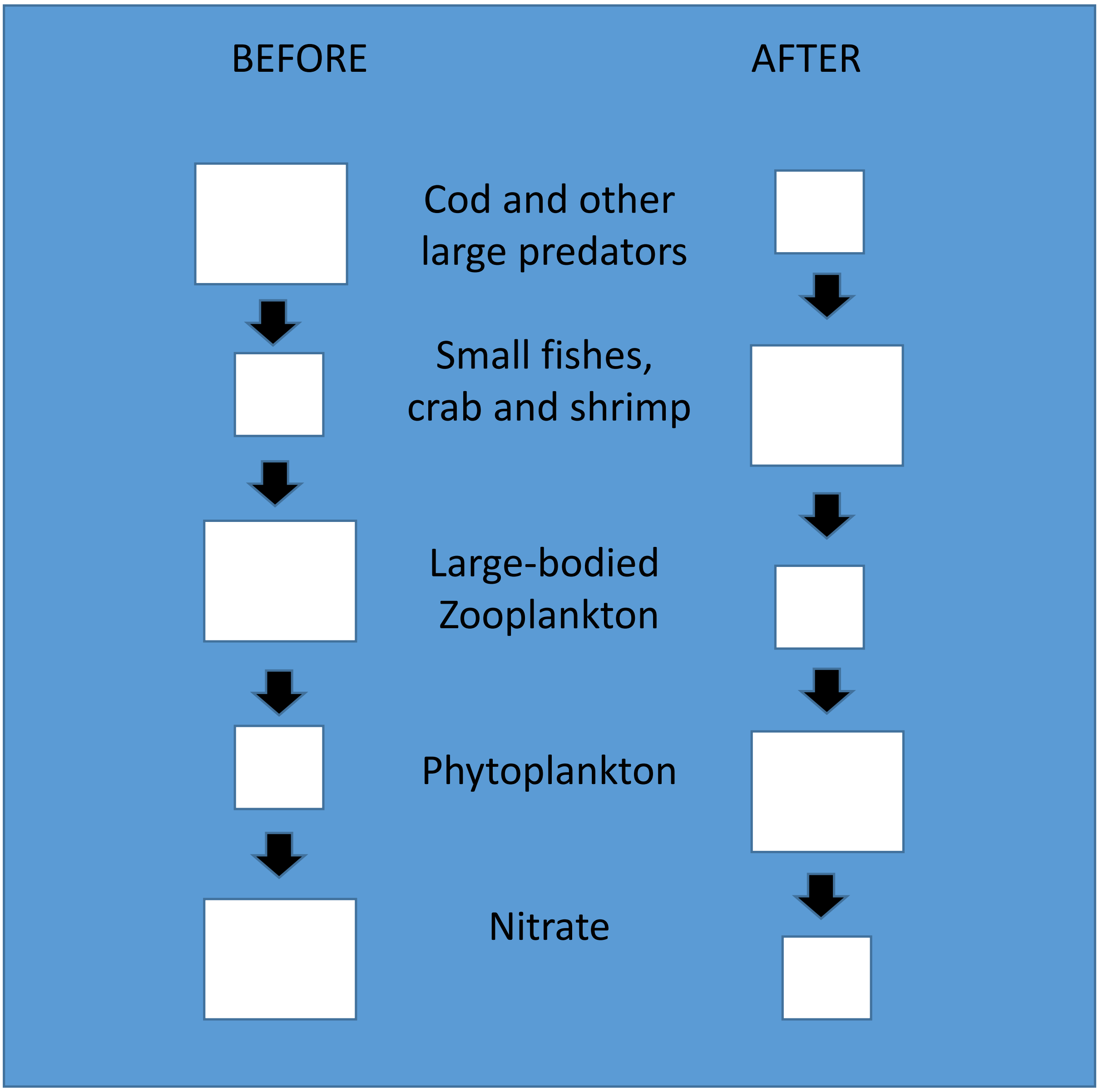

Figure 16: The cascading effect of the collapse of the cod and other large predatory fishes on the Scotian shelf ecosystem during the late 1980s and early 1990s. The size of the boxes represents the relative abundance of the corresponding trophic level. The arrows depict the inferred top-down effects (redrawn from Scheffer et al. 2005).

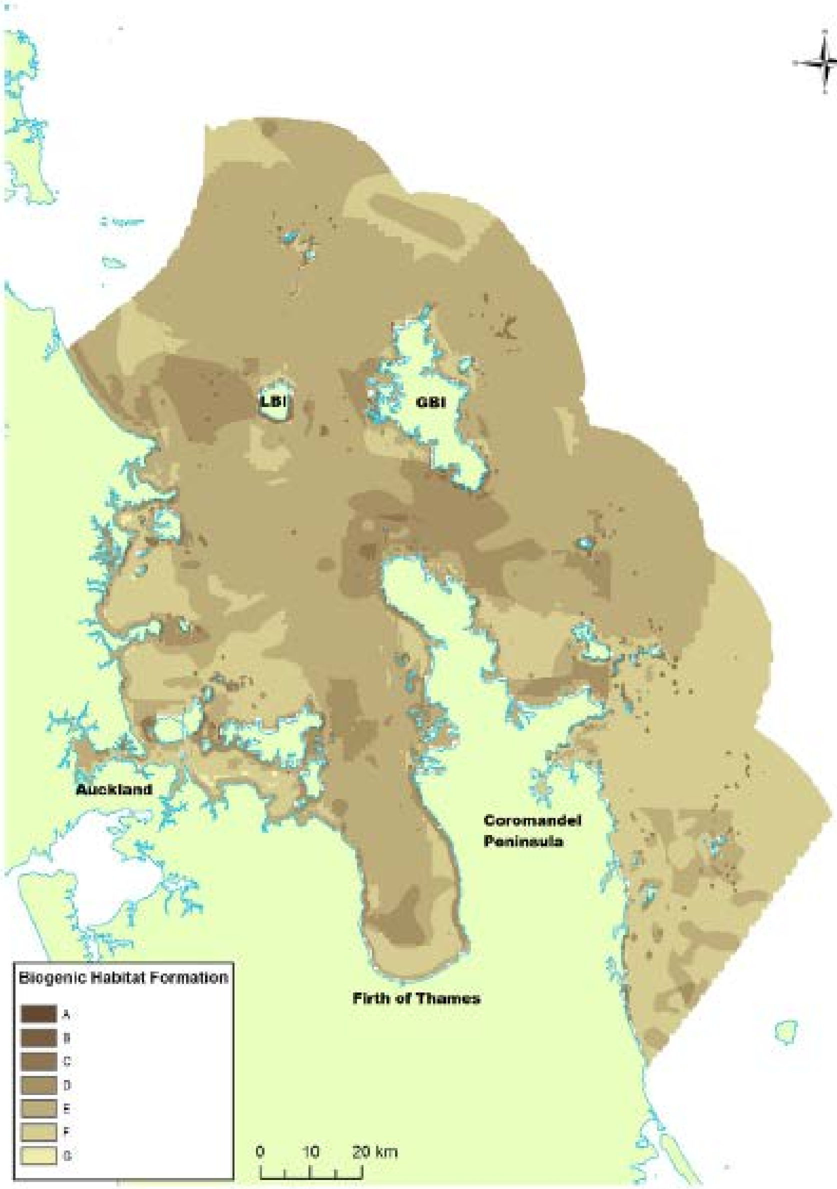

Figure 17: Diagram of predicted level of likelihood of biogenic habitat formation, an ecosystem service in the Hauraki Gulf, as predicted from an Ecological Principles approach, darker colours indicate higher likelihoods (Townsend et al. 2011).

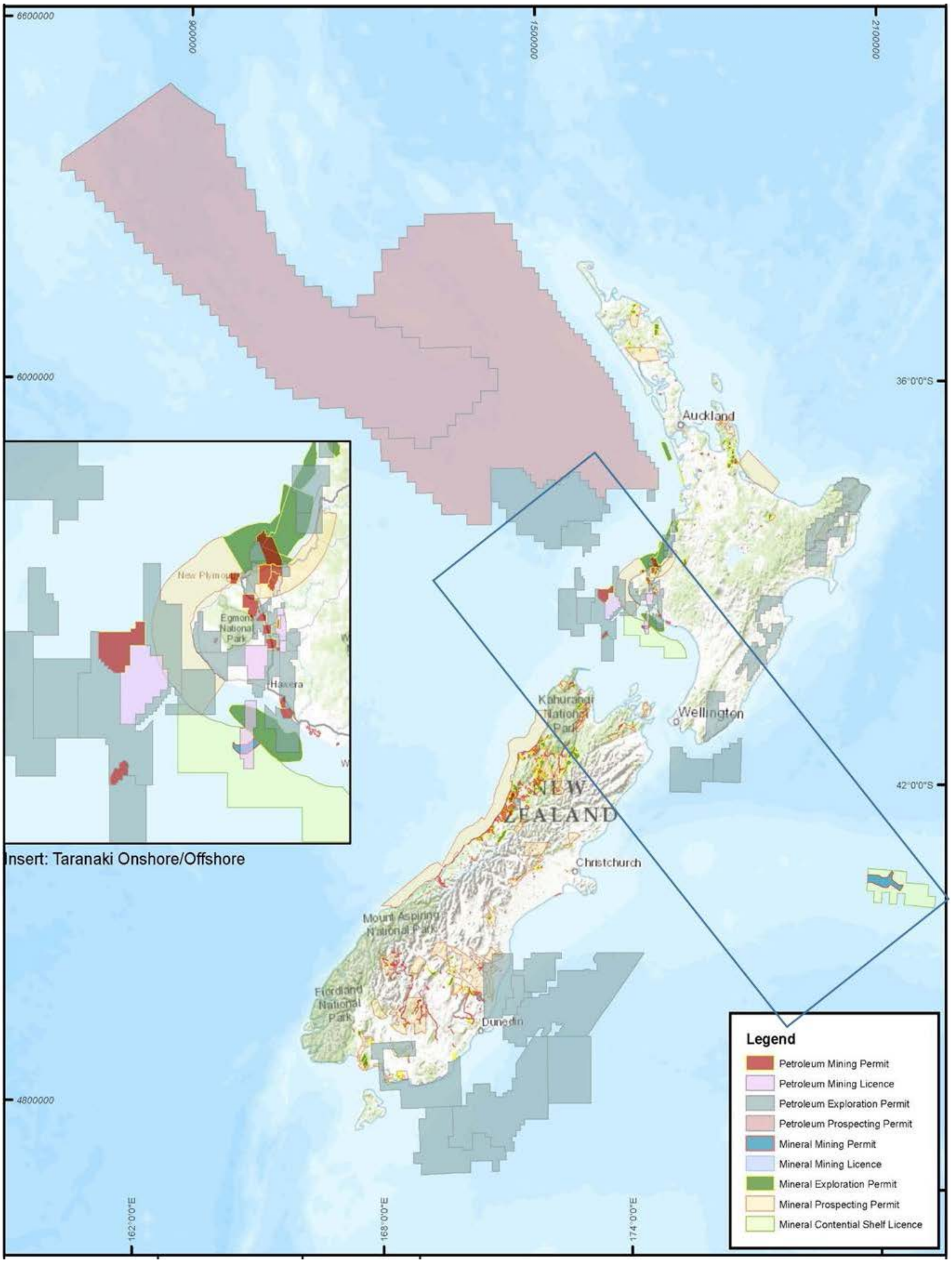

Figure 18: Focal region for research within the SSNSC[14].

[14] http://www.sustainableseaschallenge.co.nz/sites/default/files/Sustainable%20Seas%20Research%20Plan%20-%2030%20September%202015.pdf

2.9 General Discussion and Recommendations

The presentations described above formed the basis for the workshop participants to evaluate some of the key data sources, and a range of approaches and methods that could help inform the subsequent discussions. These discussions considered what scientific approaches and data could underpin robust and effective types of management of benthic impacts and could be developed in future MPI research planning.

In the short to medium-term the workshop participants reached a compromise that a fishing impact/productivity modelling approach to benthic risk evaluation was a useful starting point. This approach would be broadly similar to the SEMR approach in Australia (Pitcher et al. 2015) where overlap is gauged between fishing footprint and distribution of species or habitats. A level of impact from fishing could then be estimated based on the fishing gear used and the functional traits of the organism, e.g. fragility, position in/on the seafloor. The relative fishing impact rate (I) can then be calculated across the entire area and divided by a productivity rate (P) per species[15]. Those taxa with the highest values of I/P are deemed most at risk, a value of I/P equal to one is conceptually equivalent to Maximum Sustainable Yield (MSY), when P is assumed to equal the natural mortality. This approach makes the assumption that recovery rate is linearly correlated with natural mortality, i.e. as mortality rate decreases (for longer-lived organisms) then taxa take longer to recover. Notably, P can also be used as a proxy for recovery of habitats (using the functional or numeric recovery of the slowest recovering species present). Comparing I to P within this framework is powerful as a screening device for prioritising which species are most likely to warrant more research or management attention.

This fishing impact/productivity modelling approach would address the management need to ensure sustainability of benthic impacts by taking a risk-based approach. This would be a feasible, short-term solution which is a spatially explicit population-based assessment. This approach assumes additivity of risks and does not cater for interactions between risks that are antagonistic (decreasing in combination) or synergistic (increasing in combination), cumulative impacts or environmental thresholds. In the longer-term it is hoped we can develop risk assessments or management approaches that can more realistically capture or cater for these complexities.

This fishing impact/productivity modelling approach will rely on using available information, so there are a number of limitations to this approach, or assumptions that will need to be made. Distributions of many species (particularly species deeper than 1500 m) are not well sampled; predicted species distributions or habitat suitability maps will be used where available and considered reliable, otherwise expert judgements about distribution and abundance may be needed. For rarer species that have few sampled occurrences, this technique may not be applicable (although it may be possible to assess impacts on species richness as a whole). For species where the natural mortality rate or functional traits are unknown they may need to be assumed from related taxa, or expert judgement may be necessary. Where expert judgement is necessary expert workshops will be convened informed by whatever data are available. Where assumptions are necessary, these will be documented and transparent so that better information can be substituted if it becomes available. Quantifying uncertainty remains difficult, but not impossible using this fishing impact/productivity modelling approach, but using a number of fundamentally different modelling approaches was favoured to deal with different assumptions, or model structures, that are likely to be needed in this process. In practice, this means generating a range of predictions of risk and estimates of uncertainty.

The above process is fit for purpose in the short-term and comparable to other fisheries risk assessments in New Zealand and elsewhere, but relies on a number of model and prior development steps which are likely to take some years. The work will need to be scoped, tendered and contracted. Iterative presentations of methodology will be needed, data will need to be compiled and expert workshops convened to decide on suitable assumptions where data are scarce or found unusable. In the mid-term it is hoped that improved information on factors like distribution, fragility and recovery rate could be generated to replace assumptions within any risk assessment framework.

In developing this approach, there are a number of questions that will need to be addressed. The spatial scale of the model will need to be carefully considered given limitations in the quality and distribution of available data, and the different spatial scale of data on effort data, species, habitats and functional processes. Trade-offs will

[15] Notably care will be needed in determining an appropriate mortality rate for colonial or habitat forming organisms in particular if there functional roles are to be recognised.

therefore need to be made for this model in terms of the chosen scale of analysis. The scale of application in New Zealand (about 4 million km^2) is much larger than in south-east Australia (250 000 – 300 000 km^2), therefore, the workshop participants favoured examining a range of spatial scales. Separate models could be run on different biogeographic regions and perhaps even nesting some finer scale analyses (e.g. focused on seamount communities) within the coarser scale. It was recommended that a broad range of predicted species distribution methodologies be tested. Associated risk assessments could be regionally focused (e.g. Chatham Rise), versus nationally focused using coarse habitat descriptors (e.g. depth zone and/or sediment type); the trade-offs involved in these choices will need to be explored. There was also discussion around what modelling framework to use. The workshop participants agreed that an ensemble modelling approach (where different model types are used and results synthesised) would be the most appropriate, as this may help address any artefacts of the modelling methodology used.

The proposed process for development of this work would be iterative with presentations to either Working Groups or expert panels to refine details and agree on assumptions.

The benefits of this approach are that:

- sufficient data are available to enable an assessment (although assumptions will be necessary);
- assumptions can be explicit and reversible, e.g. if an initial assessment is risk averse (assumes high risk in the absence of information), unknown values could be updated later with alternative assumptions or new information;
- sensitivity analyses can be run using potentially important information incorporated from overseas, to assess where additional New Zealand data should be collected to reduce uncertainty;
- it is an inexpensive approach in the short-term compared with costs of any significant new data collection, e.g. large scale benthic surveys, although in the longer-term we need to continue to improve our knowledge of the seafloor in order to improve our assessment and management of risk;
- it will highlight data gaps (both spatially and in terms of our understanding of important processes), and should help focus future sampling to reduce uncertainty;
- it can incorporate new data as they become available, e.g. new sediment data layers are being produced currently;
- it can incorporate many different variables and test the impact of these on results (including different response metrics, e.g. benthic diversity versus function), for example:
    - different spatial scales of analysis, e.g. an analysis could be at a regional level or the national level initially then triaged and focused on more discrete areas as a later iteration,
    - levels of analysis (population, community or ecosystem functions[16]),
    - more data on rare distributions,

Potential drawbacks of this approach are:

- that it will be a complex ecological model (see Section 2.4.2 for an overview) and therefore not easy for all to understand;
- uncertainty has been included in the Australian modelling approach through the use of MSEs which may be difficult to incorporate within the New Zealand context. Although some types of uncertainty may be able to be built in or incorporated by producing a range of predictions using alternative methods;
- This approach needs careful consideration for application to biogenic habitats as the provision of structure (which may be dead) and reliance of other organisms on this structure adds an additional layer of complexity, which may not be well captured in this approach;
- it will take a number of years to generate risk assessment results (therefore it will not be available to fisheries management before then);
- although this approach is likely to be less expensive than the cost of multiple new surveys to establish benthic impacts specific to particular fisheries and locations, it will still take a concerted research effort over a number of years (so it will not be a trivial or inexpensive solution);
- as rare species abundance and distributions are not well known, this will limit assessing their risk, although proxies for rare species such as species richness may be trialled;
- a single-impact-focused risk assessment approach (as applied here) does not deal well with interactions, cumulative effects and ecological thresholds unexpectedly being passed. Therefore MPI hopes in the

[16] Notably diversity and function would be considered here as emergent properties of singular population assessments.

longer-term to continue developing methods of assessing or managing risk to better deal with interactions, cumulative effects and ecological thresholds.

In the longer-term, adaptive management is an option for dealing with uncertainty. This approach can assist in learning about system behaviour, but is a challenge for current management agencies which need to regulate these processes. There is also a need for funding programmes that span multiple impacts and governances to support work which will have broad relevance across a number of marine (and possibly terrestrial, e.g. forestry) industries and management agencies. The Sustainable Seas National Science Challenge (SSNSC) is a central government funding approach to try to address ecosystem-based management within the New Zealand context in the longer term. Attempts should be made to align this benthic risk assessment process with the SSNSC work. It is hoped that some of the weaknesses of addressing risk in the proposed approach may be overcome by the SSNSC. Societal views of acceptable levels of risk may also change in the medium- to longer-term, which could affect not only acceptable levels of risk but acceptable ways of estimating it as well, i.e. single issue calculations of risk may become unacceptable.

No shorter-term solutions, other than those already applied, were identified as adequate for the MPI objectives to evaluate benthic risk from trawling and dredging in New Zealand given the present level of information. It should be noted that a similar medium-term quantitative approach is being taken to assess the risk to fish species (including sharks) from fishing within New Zealand. In that instance, the National Plan of Action – Sharks (NPOA-Sharks) required a shorter-term assessment of risk to chondrichthyans (sharks, skates and rays and chimaeras) in order to prioritise management actions. This resulted in a qualitative (but data informed) assessment of risk by an expert panel (Ford et al. 2015) that is being utilised to prioritise management actions until a more robust and transparent quantitative risk assessment is completed. This approach could be utilised if necessary to get a shorter-term, but less robust or transparent, measure of risk to the benthos from fishing. However, due to the hundreds of species likely to be assessed (on the Great Barrier Reef more than 800 taxa were assessed and the shark process took a week to assess 85 taxa) the number of experts required and the need to compile data, this should not be considered a simple or cheap alternative. The logistics of assembling such an expert workshop or workshops for the time required has yet to be evaluated, and are likely to be highly challenging, if only because of expert availability.

In the longer-term it is anticipated that quantitative risk assessment methods can be developed, informed by the results of this initial workshop. It is also hoped continued benthic sampling that will improve information on factors such as distribution, abundance, fragility and recovery rate over time to replace assumptions within this or any other approach to assessing benthic risk.

3 ACKNOWLEDGMENTS

NIWA is acknowledged for providing the venue, and the Ministry for Primary industries funded the participation of experts in the workshop.

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APPENDIX 1 TERMS OF REFERENCE

Ministry for Primary Industries (MPI) Terms of Reference

Science workshop on assessing the effects of mobile bottom fishing methods on benthic fauna and habitats (18–19 February 2015)

1. Background

Bottom trawling and dredging are used to take a wide variety of fish and invertebrate species in many of New Zealand's coastal and offshore fisheries. The implications of seabed disturbance by these gears vary by habitat type and the frequency and intensity of disturbance (see MPI 2014 for a brief summary, noting that NIWA is relatively close to finalising project BEN2007-01 which will provide a much more detailed review).

The distribution and intensity of bottom trawling in both coastal and offshore trawl fisheries is now well-understood and mapped (Baird et al. 2002, 2011; Black et al. 2013; Baird et al. 2015). Shellfish dredging is still reported with low spatial precision, but most of the frequently-fished beds are persistent and well-known, so reasonable inferences can be made about the distribution of fishing. However, it is not clear how the distribution of bottom disturbance by fishing should best be compared with the distribution of the benthos in a biologically meaningful way.

There have been a number of attempts to categorise New Zealand's marine benthos on large scales (ca. 100s km) on the basis of the available information on benthic assemblage composition and/or physical drivers of this benthic composition. This has resulted in the general-purpose Marine Environment Classification (MEC, Snelder et al. 2005, 2007), a demersal fish-based MEC (Leathwick et al. 2006a) and the Benthic-Optimised MEC (BOMEC, Leathwick et al. 2012). The BOMEC was later judged as the best predictor for benthic habitats at scales of ca. 100s of km (Bowden et al. 2011). However, none of the habitat classifications was designed to predict "point" habitats like seamounts or biogenic habitats.

Species predictive modelling (as distinct from habitat classification) has also been employed in New Zealand, for example by Tracey et al. (2011a) for deepwater corals, and their overlap with fisheries footprints has been assessed (Tracey et al. 2011b).

Experimental assessments of trawl and/or dredge impacts (whether manipulative or mensurative) provide indications of the degree of change and, less frequently, the recovery trajectory for particular habitat types or species assemblages. Several such studies have been done in New Zealand, and these are largely consistent with overseas work, but it is not clear at this stage how these results can best be generalised to cover all fisheries and all benthic species and/or habitats that occur in New Zealand waters.

A number of recent developments have cast doubt on the use of either classification approaches or species predictive modelling as a basis for predicting biologically-meaningful categories that might be compared with the distribution of bottom fishing effort, or used in risk assessments:

- The BOMEC was judged unsuitable for certification of the Hoki fishery by the Marine Stewardship Council (MSC).
- Analyses in inshore habitats and fisheries have shown poor correlation between the sensitivity of fauna and predicted habitat classifications from BOMEC (Baird et al. in press).
- The recent ICES trawl impacts workshop in Norway highlighted the need for good data coverage to produce reliable species predictive models.

- There are some doubts about the utility of species predictive models for deep-sea corals for SPRFMO fisheries (although a formal test of the utility of these has yet to be completed).

Now that the distribution of fishing is well-documented, and a number of approaches to comparing this footprint with affected habitats and/or species have been attempted (both in New Zealand and overseas), MPI considers it appropriate to "step back" and consider what is likely to be the most fruitful approach for the future in New Zealand. A project to conduct a risk assessment for benthic systems has been approved for 2014/15, but its final design will be influenced by the deliberations and outcomes of this workshop.

2. Workshop Terms of Reference

Purpose

The purpose of the workshop is to address the issue

"What is the best scientific approach to assessing trawl and dredge impacts on benthic fauna and habitats in New Zealand in the short, medium and long term?"

In addition, MPI would like to understand how different forms of data (e.g. video imagery and bycatch information) could potentially better inform our knowledge of benthic impacts within this context.

Scope

The primary focus of the workshop is to advise MPI on the strengths, weaknesses, opportunities, and costs of different approaches to assessing trawl and dredge impacts on benthic fauna and habitats using existing data. The proximate use of this advice will be to aid the design of a risk assessment for benthic systems to be conducted in 2015 and 2016. For longer term (~10 years) advice, the workshop focus should be on describing what scientific approaches and data could underpin alternative broad types of management of benthic impacts. If substantial improvements in our ability to understand benthic impacts are potentially possible in the medium term (3–5 years) then advice on how to reach that goal would be extremely useful to guide MPI research planning.

Out-of-Scope

This is entirely a scientific discussion and advisory forum to aid MPI's thinking and has no decision-making powers. Recommending management approaches or management actions to address particular impacts of fishing on the benthos are out of scope, as is the final decision on the design of the risk assessment project. It is recognised that these are relevant, and are likely to be discussed; however, decisions on these matters are out of scope and the focus will be on the science of assessing benthic impacts.

Participants

Attendance at the workshop is by invitation only. Invited participants are:

Participant	Role	Affiiliation
Richard Ford	Chair	MPI
Martin Cryer	Presenter/Participant	MPI
Mary Livingston	Participant	MPI
Will Arlidge	Participant	Department of Conservation
Geoff Tingley	Participant	Private consultant
John Leathwick	Presenter/Participant	Private consultant
Simon Thrush	Presenter/Participant	University of Auckland
Roland Pitcher	Presenter/Participant	CSIRO (Australia)
David Bowden	Presenter/Participant	NIWA
Malcolm Clark	Presenter/Participant	NIWA
Alistair Dunn	Presenter/Participant	NIWA
Judi Hewitt	Presenter/Participant	NIWA
Ashley Rowden	Presenter/Participant	NIWA
Ian Tuck	Presenter/Participant	NIWA

Protocols

All workshop participants will commit to:

- participating in discussions in an objective, unbiased, and non-personal manner;
- resolving issues;
- following up on agreements and tasks;
- adopting a constructive approach;
- facilitating an atmosphere of honesty, openness and trust;
- having respect for the role of the Chair; and
- listening to the views of others, and treating them with respect.

The workshop will be run formally based on the Terms of Reference and the agenda, notes will be taken and a formal report will be generated. Participants who do not adhere to the standards of participation may be requested by the Chair to leave a particular part of the workshop or, in more serious instances, will be excluded from the remainder of the workshop.

Chairperson

The roles of the technical workshop Chair include that of a facilitator, and the Chair is responsible for:

- setting the rules of engagement consistent with the workshop's purpose and scope;
- promoting full participation by all members;
- facilitating a constructive discussion per the workshop's protocols;
- focusing the workshop on relevant issues; and
- working with the panel members to achieve the workshop's objectives consistent with the workshop's approach.

The Chair is responsible for working towards an agreed view of the workshop participants, but where that proves not to be possible then the Chair is responsible for making any final decisions and recording differences of opinion.

Conflicts of Interest

Panel members will be asked to declare any "actual, perceived or likely conflicts of interest" before involvement in the workshop, and any new conflicts that arise during the process should be declared immediately. These will be clearly documented in the notes of the workshop. Management of conflicts of interest will be determined by the Chair.

Documents and record-keeping

The overall responsibility for record-keeping rests with the Chair and any facilitation staff, including:

- Recording the relevant approaches to the issue including
    - Strengths, weaknesses, data needs, possible improvements and suitability for risk assessment or longer-term.
- In cases designated by the Chair, recording the extent to which consensus was achieved, and recording any residual disagreement.

The findings of this workshop will be documented in a report, whose drafting and compilation will be overseen by the Chair, with feedback and agreement sought from all participants. Individual panel members' views may be recorded as part of the workshop, but will not be released in the final report. Until that report is released publicly, advice from the workshop should be considered draft and remain confidential.

3. Agenda

Location and timing: Brodie boardroom, NIWA, Greta Point, Wellington (18-19 February 2015, from 0930 to 5pm if necessary).

Could visitors to the site please sign in at the reception in the Allen Building.

Proposed structure:

Welcome and Housekeeping

Presentations of 20-30 minutes each (unless otherwise stated) with 10 minutes for discussion following each, in the following proposed order:

1. Brief introduction covering where MPI sees the status of the science, potential risk assessment approaches, and existing management objectives (Martin Cryer)
2. Brief summary of existing data sources in New Zealand; e.g. cores, imagery, bycatch data (Dave Bowden)
3. Brief update on project BEN2007-01 and our understanding of bottom trawl and/or dredge impact on benthic fauna, habitats, and processes in New Zealand (Ian Tuck)
4. Approaches to benthic species/habitat prediction/classification (all of these talks should cover strengths, weaknesses, data needs and possible advances)
    a. Marine Environmental Classifications (MEC) approaches (John Leathwick)
    b. Species predictive modelling approaches overseas (Roland Pitcher - 1 hour)
    c. Species predictive modelling in New Zealand (Ashley Rowden)
    d. Spatial priority ranking software approaches, e.g. Zonation (John Leathwick)
    e. Direct modelling of consequences (Alistair Dunn)
    f. Science to support managing for resilience (Simon Thrush)

Please suggest any other presentations you think would aid our discussions (particularly, but not exclusively, if you are happy to give the presentation).

We anticipate that these talks and discussions will take (at least) the whole of the first day to complete. We will then organise a structured discussion on the different approaches to identify (for each):

- Data requirements
- Strengths and weaknesses
- Applicability to a risk assessment in 2015

- Suitability as a future approach (improvements over approaches feasible in the short term) and developments necessary for that to happen

MPI will write up the deliberations and conclusions in a draft *Aquatic Environment & Biodiversity Report* (AEBR) or a *Fisheries Science Review Report* (FSRR) following the meeting. Input will be sought from participants on the accuracy of that summary before publishing the report.

4. References

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

Bowden, D., Compton, T., 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.* 27p.

Leathwick, J., Dey, K., Julain, K. (2006) Development of a marine environmental classification optimised for demersal fish. *NIWA Client report HAM2006-063.*

Leathwick, J., Rowden, A., Nodder, S., Gorman, R., Bardsley, S., Pinkerton, M., Baird, S., Hadfield, M., Currie, K., Goh, A. (2012) A Benthic-optimised Marine Environment Classification (BOMEC) for New Zealand waters. *New Zealand Aquatic Environment and Biodiversity Report No 88.* 54p.

Ministry for Primary Industries (2014) Aquatic Environment and Biodiversity Annual Review 2014. Compiled by the Fisheries Management Science Team, Ministry for Primary Industries, Wellington. 560 p.

Snelder, T. H., Leathwick, J. R., Dey, K. L., Rowden, A. A., Weatherhead, M. A., Fenwick, G. D., Francis, M. P., Gorman, R. M., Grieve, J. M., Hadfield, M. G., Hewitt, J. E., Richardson, K. M., Uddstron, M. J. & Zeldis, J. R. (2005) The New Zealand Marine Environmental Classification. *Unpublished Report for the Ministry for the Environment.*

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Tracey, D., Rowden, A., Mackay, K., Compton, T. (2011a) Habitat-forming cold-water corals show affinity for seamounts in the New Zealand region. *Marine Ecology Progress Series,* 430**,** 1-22.

Tracey, D., Baird, S., Sanders, B., Smith, M. (2011b) Identification of Protected Corals: distribution in relation to fishing effort and accuracy of observer identifications. *Draft Final Report prepared for Marine Conservation Services (MCS), Department of Conservation, Te Papa Atawhai.*