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EXECUTIVE SUMMARY

Rowden, A.A.; Oliver, M.; Clark, M.R.; Mackay, K. (2008). New Zealand's "SEAMOUNT" database: recent updates and its potential use for ecological risk assessment.

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1) NIWA's SEAMOUNT database was enhanced in order to improve its utility for fisheries managers addressing issues around the effects of bottom trawling on "seamounts".

2) Forty-one additional data fields were erected and populated to create a second version of the database (now with a total of 72 data fields).

3) Access to the updated database is restricted at present to NIWA and MFish. Broader public access is planned by 2010.

4) Development of the database has been possible through collaboration between this MFish project and NIWA's FRST-funded Seamounts Programme.

5) The potential uses of the various types of data included in the new version of the database are discussed in relation to their use as indicators or measures to assess the ecological risk to seamount biota from bottom trawling.

6) Indicators/measures of risk which can be derived from data that could be added to the database in the future are proposed.

7) Four types of methods available for the task of ecological risk assessment are reviewed, and their particular advantages and disadvantages discussed.

8) It is recommended that the preliminary evaluation of these and other ecological risk assessment methods is continued through a workshop forum.

9) Recommendations for continued development of the SEAMOUNT database are made.

1. INTRODUCTION

1.1 Overview

The New Zealand region, because of its geological setting and history, has a complex seafloor relief. Tectonism and volcanism since 300 million years ago, and crucially within the last 80–100 million years, have formed a seafloor bathymetry in which isolated submarine rises feature prominently (CANZ 1997). The major physiographic features were known by the early 1970s (Brodie 1964, Wanoa & Lewis 1972, Thompson 1991), but with the advent of GPS satellite navigation, use of multibeam swath-mapping, and declassification of satellite altimetry data (Sandwell & Smith 1997), the last 10 years has seen a significant increase in knowledge of the distribution of "seamounts"[1] around New Zealand (Ramillion & Wright 2000). Such data have produced detailed bathymetry of seamounts in some areas (Lewis et al. 1997, Wright et al. 2006), but most have not been mapped in detail.

Biological research published in the primary literature on seamounts of the New Zealand region is limited, and is largely directed at the fishery or fishing impact issues (Probert et al. 1997, Clark 1999, Clark & O'Driscoll 2003, Tracey et al. 2004). Only since 1999 has research been focused on assessing the diversity and ecology of seamount benthic macroinvertebrate fauna (Clark et al. 1999). Determining the identities of species sampled from such previously unexplored habitats is very time-consuming and the results of such research effort have only recently begun to be published in preliminary/interim reports (Clark & O'Shea 2001, Rowden et al. 2002, 2003, 2004, Rowden & Clark in press).

As part of the FRST-funded research programme "Seamounts: their importance to fisheries and marine ecosystems" NIWA undertook to identify and map seamount distribution and bathymetric structure for previously "unmapped" areas of the New Zealand region, as well as determine the size, origin, geological structure, and composition of these seamounts. This information has been stored in the database "SEAMOUNT". A preliminary synopsis of the physical characteristics of seamounts within the New Zealand region (taken as the area bounded by 24° S, 167° W, 57° S, and 157° E) was initially presented by Wright (1999), and a more extensive characterisation (of over 800 seamounts) and classification of a subset (over 400) of New Zealand's seamounts has been produced (Rowden et al. 2005). The latter procedure was conducted because, in the absence of extensive and consistent biological data, biologically focused classification of physical variables will advance understanding of seamount biodiversity patterns and improve the effectiveness of seamount conservation/management strategies (Stocks et al. 2004). Rowden et al. (2005) demonstrated the potential management utility of the SEAMOUNT database by identifying groups of seamounts with potentially different benthic biota, from which individual seamounts could be selected for representative protection.

Nonetheless, the utility of the SEAMOUNT database for fisheries management can be improved, and this is underway as part of NIWA's FRST-funded Sustainable Production Systems (SPS) research project "Effective management strategies for seamount fisheries and ecosystems" (hereafter this project and the previous FRST-funded project are referred to collectively as NIWA's Seamount Programme). For example, data for fish species occurrence at (or near) seamount features are available for inclusion/linkage from the Ministry of Fisheries (MFish) database "TRAWL", as are

[1] The term "seamount" as used in this report is synonymous with the term "underwater topographic feature" and refers to all underwater features with a vertical elevation greater than 100 m. See Rowden et al. (2005) for further discussion on what is considered a seamount in the New Zealand context.

data for macroinvertebrates (at relatively coarse levels of taxonomic identification) from various databases maintained at NIWA. In addition, the inclusion of (or links to) oceanographic data that relate to the ecology of seamounts is desirable, and to specifically improve the fishery management capability of the database, data or indices that relate to fishing effort can be added. Such an improved version of the SEAMOUNT database, with a suitable interactive interface, would then have greater usefulness for the development and preliminary implementation of a method to assess the effects of trawling on seamounts. Risk assessment methods can be used to assist in devising appropriate management strategies to avoid, remedy, or mitigate the adverse effects of fishing on seamounts.

The work presented here incorporates data obtained through FRST projects (C01808, C01X0028, C01X0224, C01X0508) as well as other MFish projects (ZBD2000/04, ZBD2001/10, ZBD2004/01, ENV2005/16), and has involved a collaborative approach by researchers from a wide range of disciplines.

1.2 OBJECTIVES

1.2.1 Overall objective

To provide an analytical tool to assist in the assessment, research, and management of the risks to benthic organisms and their habitats, from bottom trawling on underwater topographical features (UTFs; see footnote on previous page).

1.2.2 Specific objectives

1. To provide an atlas with an interactive database that identifies all known seamounts in the New Zealand region, encompassing the area from 24°00' – 60°00' S, 155°00' E – 165°00' W. The atlas will catalogue relevant data (e.g., physical, biological, location, fishing effort) for individual UTFs. The purpose of the database is to facilitate data analysis in relation to management options regarding the effects of bottom trawling on UTFs.

2. To develop a preliminary risk assessment model from data stored in the database to predict the effects of bottom trawling on the benthic environments of UTFs of New Zealand's EEZ, and to identify management options for these ecosystems.

It was determined early in the work programme that an atlas as such was not going to be a particularly useful product. Efforts under Specific Objective 1 were therefore focused on developing and enhancing the database, from which any combination of variables could be extracted and mapped according to the user's requirements.

The development of a risk assessment model was not possible given the data currently available, and the uncertainty about key management priorities that would help formulate the model. Thus Specific Objective 2 was directed towards exploring potential ecological risk assessment methodologies, and indicators for use in such models.

2. THE SEAMOUNT DATABASE (VERSION 1)

The type, source, and reasoning for data stored in the SEAMOUNT (v1) database has already been largely detailed by Rowden et al. (2005); however, it is repeated and augmented here for the sake of completeness and specifically to provide a context for those data added as part of this project. The names of the 31 data fields are italicised in the text below and are also listed in Figure 1 and Appendix I.

2.1 Physical data

Physical data on seamounts have been collated from existing sources used in the updating of regional bathymetry in 1997 (CANZ 1997), including data held by NIWA, the New Zealand Hydrographic Office, Royal New Zealand Navy, National Geophysical Data Centre (U.S.A), South Pacific Applied Geoscience Commission (Fiji), published scientific papers, and recent multibeam surveys funded by Institut Francais de Recherche pour l'Exploitation de la Mer (IFREMER, France). This information was supplemented by detailed data on smaller features from University of Kiel (Germany), Seabed Mapping New Zealand Ltd., and research surveys carried out over the last 20 years by NIWA and MFish (including multibeam surveys in the last 5 years).

Latitude and *longitude* of a seamount were based on the location of the summit, which was determined from actual bathymetric data wherever possible, or from the central point of the *minimum depth contour* derived from NIWA's regional bathymetric dataset. *Depth at peak* is the shallowest depth record known from the seamount. The *depth at base* of the seamount was generally taken from the deepest most complete depth contour (*maximum depth contour*) that encircled the entire seamount. In some cases, there was an appreciable difference between sectors of a seamount, where one side is, for example, up-slope of a broader feature like a rise. In these cases the mid-point between the shallow and deep basal depth was taken. *Elevation* was computed as the difference between depth at peak and depth at base. *Area* and *perimeter* were estimated from the polygon of the basal depth contour. Slope was calculated in two ways. First, echo-sounding data from ship tracks over seamounts were analysed and *maximum slope*, *minimum slope* (usually zero at the peak), and *mean slope* (and *standard deviation of the slope*) computed. For many seamounts, however, data are inadequate for this method, and hence slope was calculated from the seamount trigonometry using elevation and base radius to derive average slope. This method tends to underestimate the true slope on the flanks of seamounts, since most seamounts have broadly domed peak regions (i.e., the method tends to average the low gradients near the peak and higher gradients on the flanks).

2.2 Geological data

The *geological association* of seamounts has been broadly categorised as being associated either with the inner New Zealand continental margin (within the enclosing continuous 2000 m isobath) or with various types of ridge and rise systems on the surrounding oceanic seafloor. Most of the known seamounts have received little or no scientific study, and their *geological origin* is not definitive, but over 500 seamounts included in the database were classified on the basis of geological composition or location, i.e.; arc/mid-plate/oceanic plate/hotspot/rifted margin volcanoes, tectonic ridge, rifted continental block, or continental rise. Fewer than 10 seamounts in the New Zealand region have any form of direct radiometric age dating (Wright 1994, unpublished data, Mortimer et al. 1998), thus most *age* determinations were based on interpretation of magnetic anomaly and plate reconstructions and a regional assessment of seafloor volcanism (Sutherland 1999).

2.3 Oceanographic data

Which *water mass* overlies each seamount was determined by reference to the characterisation produced for the New Zealand region by Carter et al. (1998). Remotely sensed data were available to directly measure sea surface temperature and derive data that give temporally and spatially continuous variables that characterise different water masses. Data for *sea surface temperature* (SST) variables 'wintertime SST', 'annual amplitude of SST', 'spatial SST gradient', and 'summertime SST anomaly' were calculated from NIWA archived SST climatology dataset. Procedures for collecting satellite radiometer data, detecting cloud and retrieving SST data were described by Uddstrom & Oien (1999), and the calculation of the specified variables for the New Zealand region (at 1 km resolution) were detailed by Snelder et al. (2006). Patterns in wintertime SST are a proxy for water mass (which is related to nutrient availability); variations in the annual amplitude of SST are due to differences in stratification and wind mixing, that together produce the mixed layer across the region; spatial SST gradient recognises fronts in oceanic water masses (and is expected to correlate with variation in primary productivity); summertime SST anomaly is expected to recognise anomalies in temperature that are due to hydrodynamic forcing, such as upwelling and vigorous mixing due to eddies (areas with high values of this variable are expected to correlate with high primary productivity). All these parameters derived from SST data possibly influence the composition of pelagic and benthic assemblages (Longhurst 1998).

2.4 Biological data

The shortest *distance a seamount is from the continental shelf* was calculated using ArcGIS where seamount distances from the 250 m depth contour (which approximates the continental shelf edge) were calculated at a resolution of ± 1 km, based on an azimuthal equidistant projection (Central Meridian 171° E, Latitude of Origin 41° S, Datum WGS84). The composition of faunal assemblages on seamounts (which are generally features of the slope or deep-sea) is expected to be in part influenced by the degree to which faunal colonisation has been possible from the shallow-water of the shelf (Leal & Bouchett 1991, Gillet & Dauvin 2000). Thus, a measure of the shortest distance from the shelf edge is expected to be a reasonable proxy for the likely extent of seamount colonisation by shallow-water species. To some extent this measure is a proxy for the general degree of biological isolation, and therefore the relative level of likely endemism of a seamount's benthic fauna. Distance from the shelf edge is also a reasonable proxy for the existence of localised, biologically meaningful, hydrodynamic processes. The intensity of the current flow field near a seamount decreases with distance from continental margins (Smith et al. 1989), which concomitantly affects the development of hydrographic features (e.g., localised upwelling, Taylor Caps) that can influence primary productivity overlying seamounts (Comeau et al. 1995).

SeaWiFS 4 km L1A daily radiances for 1998–2002 were processed using the OC4v4 algorithm (Pinkerton et al. 2005) to derive surface chlorophyll a concentrations (mg m^{-3}). The resulting daily chlorophyll products were then further processed using the Fourier decomposition and objective analysis method of Uddstrom & Oien (1999) to generate a temporal *mean chlorophyll a* (and *SD and CV estimates about the mean chlorophyll a* measure) measure for the surface water overlying each seamount. The spatial resolution of these climatologies is about 8 km. Remotely sensed chlorophyll *a* data are generally related to the relative occurrence of phytoplankton in surface waters, and given reasonable assumptions are proxies for phytoplankton biomass in the ocean above the seamount (Martin 2004). The amount of phytoplankton (or rather primary productivity) associated with seamounts is likely to influence the diversity of pelagic, and subsequently the benthic faunal, assemblages (Piepenburg & Müller 2004).

2.5 Fisheries data

Whether or not a seamount has or is subjected to bottom *fishing* is noted. This condition was determined by direct plots of deepwater tow positions (determined from the Ministry of Fisheries Trawl Catch Effort Processing Return data), as well as assignment of trawls with a recorded start position within 3 nautical miles of a summit location, with a trawl duration of less than 30 minutes.

2.6 Descriptive data

Every seamount in the database has been assigned a *registration number*, and where known the official and/or unofficial *name* is indicated. Whether a seamount is located within the New Zealand *Exclusive Economic Zone* is recorded, as is the broad *geographic area* and the *source of the physical data* (e.g., regional bathymetry).

3. THE NEW SEAMOUNT DATABASE (VERSION 2)

An enhancement of the SEAMOUNT (v1) database, via the addition and population of new data fields, was planned as part of NIWA's Seamount programme. With the advent of the present MFish project a meeting was held between NIWA and MFish representatives to discuss the new data fields, to agree on a selection that would facilitate data analysis for management options for the effects of bottom trawling on seamounts. The selection of the additional data fields was subject to the availability of appropriate data, and the resource (time and money) limits of both the MFish project and NIWA's Seamount programme. The names of the 41 additional data fields are italicised in the text below and are listed in Figure 2 and Appendix I.

3.1 Additional oceanographic data

NIWA has developed regional climatologies (at 1 km^2 resolution) for *current speed*, *mean diurnal tidal flow* and *annual mean semi-diurnal tidal flow*, and *depth of thermocline* (G. Rickard et al., NIWA, unpublished data) - measures of these variables are included in the new version of the database for the position of individual seamounts. These measures are combined with physical variables (already in the SEAMOUNT database) that describe the seamount and its position in the water column to model the likelihood that a seamount could generate closed circulation. Whether a seamount possesses such a hydrographic feature ('Taylor Cap' or 'Cone') is considered important, for nutrient delivery from depth to the surface waters can be enhanced – thereby influencing the primary and secondary/tertiary production associated with the seamount (Rogers 1994). The formation of a Taylor Cap also has implications for recruitment to, and subsequent long-term stability of, seamount communities. Many seamounts are isolated from other seamounts or the nearest shallow water topographies by 100s or 1000s of kilometres, making them difficult for new recruits to reach. The greater the likelihood of a Taylor Cap forming over a seamount the more likely passing larvae are to be entrained in the seamount circulation and to settle there (Mullineaux & Mills 1996). Once populations are established, the strength of this same circulation system may retain larvae that would otherwise disperse away from the seamount and this retention potentially increases the proportion of endemic species found there (see arguments in introduction of Beckman & Mohn (2002)). In contrast, a seamount with a low likelihood of Taylor Cap formation is less likely to entrain passing larvae or retain those that are produced on the same seamount, and may have taxa more representative of the surrounding area. Two measures of the likelihood or probability of Taylor Cap formation have been generated for inclusion in the SEAMOUNT (v2) database; these are *probability of Taylor Cap* *formation in mean flow* and *probability of Taylor Cap formation in tidal flow*. These measures are derived from numerical studies of flow over seamounts, and have been adapted for the variables available in the present database. If either the mean flow or the tidal flow dominates, then the likelihood is that the nature of the Taylor Cap formation will be consistent with the dominant component. For seamounts where both components are equally significant it is expected that cap formation will still occur, but that the interactions between the forcing flows will result in more complex flow patterns. Further details of how these two measures were calculated are provided in the documentation that accompanies the database (Mackay 2007).

3.2 Additional geological data

The geological origin of a seamount has been further usefully qualified, for volcanic seamounts, by indicating whether a seamount is considered *active* (extinct, dormant, active), and specifically if it is thought or known to posses active *hydrothermal vents*. Submarine volcanic eruptions, though inherently difficult to record, have been interpreted from indirect hydroacoustic *T*-wave data, eruptive manifestations at the sea-surface, and even the emergence of ephemeral island volcanoes. For the New Zealand region, such recordings (although almost certainly under-reported) are restricted to seamounts along the Kermadec Ridge (Kibblewhite 1967, Davey 1980, Latter et al. 1992, Lloyd et al. 1996, Wright et al. 2006). Similarly, the discovery of hydrothermal venting at depths of 500–2000 m below the sea-surface has been difficult. However, more recent and systematic surveys of seamounts along the Kermadec Ridge using towed sensor arrays measuring water chemistry and optical properties have established the incidence of the more significant submarine hydrothermal venting (de Ronde et al. 2001, Baker et al. 2003). Some of these indicative signals of venting have been confirmed by visual observations at the seafloor, using either towed camera arrays, ROVs, or manned submersibles (Rowden et al. 2003).

To date there are no regional studies of seamount substrates within the New Zealand region. The only regional compilation of substrate type is for seafloor sediment composition (Mitchell et al. 1989), which is produced on a scale too coarse to realistically resolve sediment types for a seamount. At smaller spatial scales, modern swath imagery data (typically at an acquisition frequency of about 12 kHz), although restricted to relatively small areas, can provide important information on general substrate compositions at scales of 100–1000 m. Such swath mapping imagery has been acquired from only a few areas where significant numbers of seamounts exist (southern Kermadec/Colville Ridges and Havre Trough, eastern North Island and Chatham Rise; (Coffin et al. 1994, Blackmore & Wright 1995, Lewis et al. 1997, Lewis et al. 1999, Barnes et al. 1998)). These swath imagery data can differentiate broad areas of sediment and rock substrates (Orpin 2004) and the nature of large-scale degradation and mass-wasting of seamounts. More recently, as part of detailed geological investigations of specific seamounts along the southern Kermadec arc (Wright 1994, 1996, 2001, Wright & Gamble 1999) higher frequency and higher resolution multibeam systems (at 30 kHz) have been used (Wright et al. 2006). From these detailed investigations it is possible to describe substrate heterogeneity at scales of 10s to 100s of metres through integrating data from swath mapping backscatter imagery, seafloor photography, and/or seafloor sampling (Wright et al. 2002). Thus, included in the database is a field which indicates whether or not seamount-specific *substrate information is available*, and all available data are located on 'Tsunami', a mass storage device for multibeam data held by NIWA. However, where it is known that particular seamounts (or the areas in which they exist) possess (or can be predicted to possess) *substrates of potential interest for mining*, e.g., ferro-manganese crusts or sulphide rich deposits, this has been recorded in this data field.

An objective morphometric analysis of New Zealand seamounts (morphology) has not been undertaken during the project due to time constraints. Standard hydrographic classifications based on subjective interpretation of seamount morphology and elevation (e.g., seamount, knoll, guyot) could have been applied using the standard International Hydrographic Classification (IHO/IOC 1988). However, while this sort of 'analysis' has already been partly undertaken for submarine features of New Zealand (Eade & Carter 1975, Thompson 1991), such classifications are subjective and time consuming, requiring analysis of each feature, and, therefore, this morphological categorisation was not undertaken. Algorithms and/or GIS based morphometric analysis which could determine a seafloor feature's "footprint area", degree of elongation, elevation, slope, aspect, volume, and corresponding ratios of these parameters, would provide a quantitative and robust analysis of seamount morphology. Morphological measures would provide insight into the possible relationships between, for example, the size of a seamount and the composition of the associated biotic assemblage. However, at present there is a limit to which an objective determination of seamount morphology can progress due to the highly variable quality of the bathymetry data for the New Zealand region. Much of the existing seamount bathymetry is based on limited, poorly navigated, single-beam echo-sounding profiles, though more recent mapping has used modern multi-beam systems to provide 100% coverage of the seafloor. Newly compiled and updated bathymetry datasets could be used for a morphometric analysis of only a small proportion of the region's seamounts.

3.3 Additional biological data

Remotely sensed estimates of chlorophyll *a* are useful proxies for phytoplankton biomass, but chlorophyll *a* is not an ideal proxy for primary productivity. Measures of primary productivity will provide a better indication of the type of communities that a particular seamount environment supports. That is, the composition of faunal communities will in part be determined by the quantity of the potential phytoplankton food source that is in the water above a seamount, be it directly or indirectly available to the seamount associated fauna (Rogers 1994). Algorithms to estimate primary productivity have been developed (Behrenfeld & Falkowski 1997), and recently these global algorithms have been modified to estimate primary productivity for specific areas. Primary productivity estimates have been made for areas of the New Zealand region using two algorithms under development by Behrenfeld and collaborators. These primary productivity measures are derived from a Vertically Generalised Production Model (VGPM) and a Carbon-based Model (Carbon2). Recent work has suggested that the flux of primary production across the seamount (not just the amount of local primary productivity, which itself might be enhanced by the formation of a Taylor Cap – see earlier section) is a more sensitive/pertinent indicator of the type of community that one might expect to develop on a particular seamount (Bulman et al. 2005, Morato & Pitcher 2005). Thus the two estimates of primary production above a seamount have been spatially extended to determine measures of *net primary productivity-VGPM* and *net primary productivity-Carbon2* associated with a particular seamount. It is important to note that due to the lack of published validation results and the substantial differences between the two primary productivity algorithms, these data (and the derived net primary productivity estimates) should be treated as preliminary. Currently it is not possible to decide which of the primary productivity estimates is more meaningful, so both are included in the database. Detailed information of how the values underpinning the algorithms were derived are given in the documentation that accompanies the database (Mackay 2007).

Unfortunately, data for zooplankton (abundance or biomass) directly associated with seamounts in the New Zealand region are very scarce, and it is only recently that plankton data have begun to be systematically collected from above seamounts in the region. Thus it is unlikely that any such data will be added to the database for at least a few years.

Data for the benthic macroinvertebrates of seamounts in the New Zealand region are relatively sparse and generally unstandardised in their collection and taxonomic resolution. Whilst efforts are currently underway to improve data quality it was not possible to complete this exercise within the timeframe of this project. Nonetheless, it is possible to record in the database whether or not biological samples have been taken for a particular seamount (as *biology sampled*). With respect to management considerations, a separate data field has been created to note that records exist for the occurrence of those benthic macroinvertebrates thought particularly vulnerable to the impacts of bottom fishing and important for habitat structuring (e.g., see comments of Probert et al. (1997) for New Zealand context, and Section 5), in this case *structure-forming corals* (the matrix-forming scleractinian corals - *Solenosmilia variabilis, Madrepora oculata, Goniocorella dumosa, Enallopsammia rostrata, Oculina virgosa* (Tracey et al. unpublished data), all *octocorals* (Sanchez & Rowden 2006) and *sponges* (Kelly-Shanks, unpublished data, Rowden & Kelly-Shanks, unpublished data). The records for these taxa are maintained separately in other NIWA marine databases.

Occurrence data for fish fauna sampled from seamounts are more robust than data for benthic macroinvertebrates, and are already stored in the TRAWL database of MFish. Extracts have been made from this database to produce by individual seamount feature, counts of taxa comprising teleost fishes, elasmobranchs (sharks, rays, chimaeras, and ghost sharks), squid, and octopuses. These data (between water depths of 500 to 1700 m) are listed as *number of fish (from research surveys)*, *number of squid and octopus* and *number of research trawls*. This last variable can be used to assess the sampling effort that resulted in the counts.

3.4 Additional fisheries data

Recently, NIWA scientists have developed indices that evaluate the relative importance of seamounts for deepwater fisheries (Clark & O'Driscoll 2003) and intensity of fishing on seamounts (O'Driscoll & Clark 2005) in the New Zealand region. These indices, the *Fishing Importance Index* and *Fishing Effects Index,* are both included in the database. They are calculated from a number of catch and effort variables from Trawl Catch Effort Processing Returns which are stored separately on the MFish database "WAREHOU", but are also included in the SEAMOUNT database in order that the indices can be calculated and updated with relative ease. These variables are: by target species – *number of tows*, *catch* in tonnes, *number of years* in which there were 10 or more tows (all within 10 km of seamount centre); and by seamount – *summed tow length* in km of all tows within 10 km of the seamount centre and of tow length less than 5.6 km, *number of tow directions* (from 1 to 4) for each seamount that had more than five tows. The latter is derived from information for *direction of tow* (north, south, east, west – four separate data fields). In addition, data for the *year first fished* has also been included in the database.

3.5 Additional descriptive data

Whether a seamount is subject to any specific *mining interest* has been indicated with reference to current exploratory mining leases sourced from the Crown Minerals division of the Ministry of Economic Development. This data field refers specifically to mineral exploration and mining permits and does not relate to petroleum or hydrate exploration or production.

The *protection status* of a seamount has been derived from the Department of Conservation report describing all central government area-based restrictions for the New Zealand marine environment (Froude & Smith 2004). For 19 seamounts, there is a prohibition on the use of trawl nets by any commercial fisher to protect the benthic biota from the potentially damaging effects of bottom trawling (Froude & Smith 2004).

Variation in a number of physical and oceanographic parameters across the New Zealand region has been summarised on a spatial basis by the recent Marine Environment Classification or MEC (Snelder et al. 2006). The *environmental class*, according to this classification, to which an individual seamount belongs is noted in the database. The MEC can operate a variety of class 'levels'. The MEC classification strength analysis revealed that below the 15 class level the classification detail was not statistically significant for the benthic invertebrate test data. Between the 20 class level (when statistical significance is achieved for all three test data sets) and the 40–50 class level there was very little difference in classification strength. After the 50 class level there are notable increases in classification strength (with respect to the three test data sets) up to the 60–70 class level, thereafter (above 75 class level) there is little gain in classification strength (up to the maximum of 290 classes). Large bathymetric features such as the Chatham Rise and Challenger Plateau, become aligned with separate MEC classes at the 33 class level. Following the request of MFish, the identity of the class to which a seamount belongs is noted at two classification levels, 20 and 33. These two levels were chosen by MFish because they reflect the low level of spatial detail that can be readily appreciated by managers/stakeholders and easily incorporated into management plans, and, for the latter class level, distinguish between two areas thought to be ecologically distinct.

The classification of about half of the known seamounts in the New Zealand region, also based largely on physical and oceanographic parameters (in addition remotely sensed surface water chl *a*) (Rowden et al. 2005), provides an indication of the degree to which seamounts may provide a similar environment for benthic biota. The *seamount class*, according to this classification (which operates at a 12 class or group level), to which an individual seamount belongs is also noted in the database. Such information will be useful for considering the representative nature of a particular seamount and, because the classifications are designed to be biologically meaningful, also potentially representative of its associated fauna.

3.6 Database structure

The SEAMOUNT (v2) database is currently held as an Empress RDBMS database located in the MFish "snapper" server (managed by NIWA). The SEAMOUNT (v1) database was a spreadsheet and this is reflected in the schema of the second version of the database where there is one large table with many fields but which is now linked to other databases (see above). The overriding factor when designing the schema was the need to transfer data easily between local copy spreadsheets (that individual researchers might have) and the central database. Full details of the structure and content of SEAMOUNT (v2) database are provided in the documentation that accompanies the database (Mackay 2007).

3.7 Interactive capability

The SEAMOUNT (v2) database can be accessed using SQL commands via the dedicated network connection between MFish and NIWA. The data are currently versioned, with editable MS Excel spreadsheet versions held with various NIWA researchers. These editable versions are regularly reconciled and checked-in against the master database. Snapshots of data can be distributed on MS Excel spreadsheets, as they have been to MFish, but these are not reconciled.

4. ECOLOGICAL RISK ASSESSMENT

The context for this section includes the requirement for a risk assessment of the effects of fishing in New Zealand waters, the attendant concepts of ecological risk, and what is meant by measures and indicators of risk. These features of ecological risk assessment are discussed separately below.

4.1 The national requirement for risk assessment

The marine environment provides a number of resources for humankind. Fish, in particular, provide a valuable social and economic resource. Unfortunately, fishing, commercial fishing in particular, has and can significantly impact the seabed environment (see papers in Barnes & Thomas (2005), and references cited therein). Fishing activity in New Zealand is no exception as regards its influence upon benthic habitats, communities and species (e.g., Thrush et al. 1998). However, there is in New Zealand a body of legislation that aims to ensure that the impact of fishing activity is minimised while allowing for a sustainable fishery. The single most significant piece of legislation for this purpose is the Fisheries Act of 1996. This Act establishes a number of obligations, including a requirement "to avoid, remedy or mitigate any adverse effects of fishing on the aquatic environment". While there have been, and are, a number of initiatives to address specific issues that relate to the adverse effects of fishing, it is only relatively recently that MFish established an overall strategy specifically aimed at managing the effects of fishing. The Strategy for Managing the Environmental Effects of Fishing (SMEEF) sets out the approach that MFish is in the process of implementing in order to meet environmental obligations across all its activities and procedures (Ministry of Fisheries 2005). Fundamental to the SMEEF framework is the setting of 'Environmental Standards' which define "the point at which the effects of fishing on an element of the aquatic environment moves from being acceptable to unacceptable, or adverse." (section 2.2.2, p 6, Ministry of Fisheries 2005). As part of the "process a fishery manager should use to identify environmental standards relevant to a fishery and determine the appropriate management response", there is a requirement for a "risk assessment process by which species and habitats requiring standards as a high priority are identified" (section 2.3.3, p 8, Ministry of Fisheries 2005). It is important to note here also that the SMEEF implementation process (for setting standards) identifies that it will be necessary for MFish to "establish and maintain links with relevant research and management organisations", such as NIWA, and "develop systems to obtain necessary information on the threat status of species and habitats", such as the SEAMOUNT database (section 4.3, p 20, Ministry of Fisheries 2005).

4.2 Concepts of risk

The SMEEF notes that the system of determining and prioritising the setting of Environmental Standards, the fundamental unit of the strategy, "should be based on the level of risk to each species and habitat, including consideration of the likelihood of an adverse effect, the severity and reversibility of the effect, and the nature of available information." (section 2.2.3, p 7, Ministry of Fisheries 2005). Here, then, the SMEEF refers to the 'concepts of risk' – "likelihood" of an adverse effect, the "severity" of an effect and the "reversibility" of the effect".

MFish is not alone in attempting to address the risk that is posed by anthropogenic activities to the marine environment, and other bodies elsewhere have adopted similar concepts of risk when attempting to manage, conserve, or protect the environment (e.g., the United Kingdom and the Republic of Ireland's MarLIN scheme; see Hiscock & Tyler-Walters (2006) for most recent summary of this scheme). To some extent the terms used to describe these concepts have become standardised and thus the terms for the overarching concept of 'level of risk' or 'threat status' used by the SMEEF are hereafter (for the sake of commonality and possible comparability with already established schemes) referred to singularly as 'sensitivity'. The related risk concepts 'likelihood' and 'severity' are often combined under the term 'vulnerability', while 'reversibility' is more often termed 'recoverability', and so these more standard terms are also used here. The SMEEF is unusual in using the term "species and habitat" as a shorthand way of referring "to all the elements and relationships within the aquatic environment that may be affected by fishing" (section 1.4, p 3, Ministry of Fisheries 2005). In other words the term is intended to imply consideration of the "species' role in the ecosystem", for example, in the way in which it contributes to the biological unit commonly refereed to as a 'community'. Again for the sake of consistency with other schemes that seek to assess the risk to the marine environment posed by anthropogenic activities, hereafter reference will be made to the biological concept of communities.

4.3 Measures and indicators of risk

The United Kingdom and the Republic of Ireland's Marine Life Information Network (MarLIN)[2] has been at the forefront of attempts to develop various measures (and the means to assess them, see Section 6) which can be used in processes for the conservation and management of the marine environment (http://www.marlin.ac.uk/sah/baskitemplate.php?sens_ass_rat – hereafter referred to as MarLIN website). The concept of risk, for which MarLIN has developed a measure and an assessment technique, is **sensitivity** (Hiscock & Tyler-Walters 2006). MarLIN was by no means the first or only concerned body to devise a means to assess sensitivity (or a sensitivity index) (see, for example, MacDonald et al. (1996)). Other national schemes are under development in Canada (Arbour 2004) and Australia (Hobday et al. 2007). However, it appears that the MarLIN approach is currently the most well developed, and has already been implemented and incorporated into conservation/management practice in Europe (see examples in Tyler-Walters & Hiscock (2005)), and thus it will be used here as the basis for discussion. However, it is noted that the Australian scheme is the one that any scheme for New Zealand will need to be fully cognisant of, or even comparable to/compatible with [At the time this project report was submitted for publication the final CSIRO-AFMA report that describes the Australian scheme was not officially available for consultation.]

MarLIN notes that "sensitivity is dependent on the intolerance of a species or habitat to damage from an external factor and the time taken for its subsequent recovery. For example, a very sensitive species or habitat is one that is very adversely affected by an external factor arising from human activities or natural events (killed/destroyed, 'high' intolerance) and is expected to recover over a very long period of time, i.e., >10 or up to 25 years ('low'; recoverability). Intolerance and hence sensitivity must be assessed relative to change in a specific factor" (MarLIN website). Thus, sensitivity involves measures of two other concepts of risk, **intolerance** and **recoverability**. MarLIN defines intolerance as "the susceptibility of a habitat, community or species (i.e., the components of a biotope) to damage, or death, from an external factor. Intolerance must be assessed relative to change in a specific factor"; and recoverability as "the ability of a habitat, community, or species (i.e., the components of a biotope) to return to a state close to that which existed before the activity or event caused change" (MarLIN website). In other sensitivity assessment schemes, the concept of intolerance is replaced with the related (and sometimes synonymous) concept of **vulnerability** (e.g., DFO 2005). The concept of vulnerability can capture not only the intolerance of a biological unit to disturbance, but also "the likelihood that a [biological] component will be exposed to some impacting factor" (DFO 2005). In the context of the present project's aim to identify measures suitable for the

[2] The UK's MarLIN is not to be confused with the MFish meta database managed by NIWA and also called MarLIN.

assessment of risk pertaining to the effects of fishing disturbance on the benthic communities of seamounts, it seems sensible to adopt the risk concept of sensitivity, and the associated concepts of vulnerability and recoverability. However, the above definitions of these concepts can be modified according to the specific context in which they are to be used.

Here, sensitivity is defined as:

> The vulnerability of a habitat, community, population, or individual (or individual colony) of species to disturbance from the direct, or indirect, effects of fishing, relative to its recoverability from such a disturbance.

The component concept of vulnerability is defined as:

> The likelihood and degree of disturbance to a habitat, community, population, or individual (or individual colony) of species from fishing activities;

and recoverability is defined as the:

> Ability of a habitat, community, population, or individual (or individual colony) of a species to return to a state close to that which existed before fishing activities caused change.

It should be noted that while the MarLIN sensitivity assessment scheme offers useful guidance for the development of measures/indicators for an ecological risk assessment of seamounts, the other related national schemes or international sensitivity initiatives which are somewhat more complicated in structure also provide helpful direction. These schemes concern themselves with identifying areas, for example, of "particularly high ecological or biological significance" (DFO 2004), often with the view of affording protection to these areas, and include an assessment process that examines data under a number of "criteria". Of these criteria, at least one – **representiveness** – is not directly or indirectly captured within the concepts of vulnerability and recoverability, and yet a measure of this concept could be useful for the development of an ecological risk assessment for disturbance by fishing of seamounts. As such, MFish requested that representativeness to be specifically considered by this project. Representiveness is variously defined (e.g., "Typical of a feature, habitat or assemblage of species. Representative examples are identified from the range of natural or semi-natural habitats and associated communities (biotopes) within a biogeographically distinct area or the boundaries of a national territory", MarLIN website), but in the context of the present issue it could be more specifically defined as:

Typical of a seamount or habitat within an environmentally distinct area.

5. DEVELOPING INDICATORS AND MEASURES OF RISK

The MarLIN scheme and others identify that in order to assess the sensitivity of a habitat, community, etc, there is a need to collate "key information" which can be used as indicators or measures of the various risk concepts. In the MarLIN scheme, this information is systematically collected and stored in a database ('Biology and Key Information' database) available to those who undertake the formal assessment of sensitivity. Similarly the SMEEF notes that in order to assess the risk to the environment from fishing it is desirable to be able to identify "biological reference points", which may for example, relate to the "role of the species or habitat in the functioning of the ecosystem in which it occurs" (section 3.3, p 15, Ministry of Fisheries 2005). Again context is important and it is necessary to identify the types of key information and data that are specific to benthic habitat, communities, etc, of seamounts so that appropriate measures/indicators are developed. However, while it is possible in theory to identify a large set of key information for assessing the sensitivity of the environment to a form of disturbance, data relevant to seamounts are not always (or indeed often) available. The SMEEF recognises that appropriate information might be limited, but notes that the assessment of risk should proceed nonetheless using the "best available" information, while also indicating that assessments of risk "should be updated periodically to include assessment" of "new information" (section 2.2.3, p 7, Ministry of Fisheries 2005). Hence the following elaboration of key information and indicators/measures takes into account the present availability, and likely future availability, of the type and quality of data collected for habitat, communities and species, and fishing, associated with seamounts in the New Zealand region.

The proposed indicators and measures of the various concepts of risk associated with assessing the impact of fishing on seamounts should be able to be extracted directly, or determined via the use of some simple associated automation script, from the database. Those indicators/measures already available via data stored in SEAMOUNT (v2) and those likely to be added in the near future (1 to 5 years) are italicised in the text below. A complete list of the proposed indicators/measures, including those which might become available in the more distant future (over 5 years), appears in Table 1.

5.1 Vulnerability

Many factors can make an environment and its biota vulnerable to fishing disturbance, and some of these features, which could be used in an assessment of sensitivity, have already been identified by previous risk assessment schemes. These include what the MarLIN sensitivity assessment scheme terms "structural", "functional", "characterising", and "other important" species (Hiscock & Tyler-Walters 2003). That is, species whose population degradation or loss would likely influence the integrity of the community as a whole. Our current knowledge of seamount communities in the New Zealand region means that only a measure of the structural species, and, in a restrictive sense, the presence of species that characterise a particular community/habitat are applicable.

Structural species are those that "provide a distinct habitat that supports an associated community" (Hiscock & Tyler-Walters 2003) (these species are also sometimes functional). Such species for seamounts include the corals, particularly matrix-forming scleractinians (e.g., *Solensmilia variablis, Madrepora oculata*), and sponges. The physical structures formed by both these groups of organisms can provide habitat for rich and diverse communities of other fauna, and the structures are often relatively large and fragile and therefore vulnerable to damage or destruction caused by bottom trawling (Koslow et al. 2001, Clark & Koslow 2007). Thus, the *presence of habitat-forming species (corals and sponges)* on a seamount is a useful indicator of vulnerability.

There are many good quality presence records for corals and sponges from seamounts in the New Zealand region, but most seamounts have not been biologically sampled and no information on the widespread presence/absence of habitat-forming species is available. However, in the future it will be possible, and it is planned, to model the distribution of these taxa on seamounts and so a measure will be available that relates to the predicted occurrence of the number of habitat-forming species (FRST-funded biodiversity project BFBB082). The results of future study might also reveal that other taxa (e.g., brachipods) provide important structural habitat on some seamounts, and data for these species can also be incorporated into such a measure.

The planned analysis of biotic data gathered by NIWA's Seamount Programme will result in the identification of species that characterise particular communities on New Zealand seamounts

(Rowden & Clark, unpublished data). Until that analysis is completed, the only species that can be said with any degree of certainty to be characterising species of a specific community are the bathymodiolid mussels that are obligate inhabitants of hydothermal vent habitats on the seamounts of the Kermadec volcanic arc. Hydrothermal vent habitats can posses a rich and diverse community, and because they are relatively small in area they are typically more isolated than other deep-sea environments. This isolation means deep-sea vent habitats possess communities with relatively high levels of endemism (Wolff 2005). Indeed, the mussel *Gigantidas gladius* (von Cosel & Marshall 2003), and two other unnamed species of bathymodiolid mussel, are to date thought to occur only at vents in New Zealand waters (Smith et al. 2004b). Such qualities make hydrothermal vent habitats and their fauna vulnerable to disturbance by fishing. Thus, the *presence of vent mussels* and *venting* are useful indicators of a seamount's vulnerability to fishing activities.

Vent mussels, along with a number of other marine invertebrate taxa, are currently listed as 'threatened' species under the "New Zealand Threat Classification System" of the Department of Conservation (DoC) (Hitchmough et al. 2007). The appropriateness of the designation procedure for marine species used by DoC is presently under review (and by default the list of 'threatened' species itself). In the future there is potential worth (if the flaws inherent in the present designation scheme are addressed) in including the *presence of threatened species* as an indicator of the vulnerability of seamounts. In the meantime, the related *presence of legally protected species* can act as an indicator of the vulnerability of a seamount to fishing disturbance. Currently, "black" corals of the order Antipatharia and "red" corals (a definition not strictly confined to a specific taxonomic group, but including the stylastrid 'coral' *Errina novaezelandia*) are afforded protection. Again because of recent review and applications for additional species to receive protected status, the number (and identity) of taxa covered by this indicator is likely to change. Therefore, in the future, information included for this indicator will require modification.

Protection is also afforded to species that comprise communities on seamounts by other means. At least two seamounts (Brothers, Rumble III) that are known to possess hydrothermal vent communities are protected from bottom fishing by the 2001 designation of 'protected status' to 19 seamounts in the New Zealand region (Anon 2001, Brodie & Clark 2003). Clearly, if a seamount is protected from fishing then it is no longer vulnerable to disturbance from this activity. As well as this seamount-specific protection from fishing, there are seamounts which are protected via other forms of legal protection e.g., Marine Protected Areas (MPAs), which afford protection to the Kermadec Islands, Auckland Islands, Mayor Island, and Volkner Rocks. Some seamounts in the New Zealand region are also protected by the MPAs of Australia (e.g., Macquarie Island and Lord Howe Island Marine Parks) (see Rowden et al. (2005) for detail). In other instances the purpose that brings about protection might be unrelated to conservation aims but will nonetheless prevent disturbance of the seabed by fishing (e.g. cable corridors). Thus, whether or not a *seamount is legally protected* is a useful measure of its lack of vulnerability to fishing (though not necessarily mining). Should further legal protection status be achieved for seamounts in the New Zealand region through other means in the future, then information on whether or not a seamount falls within such an area can also be used to evaluate vulnerability. For example, since the database was compiled, Benthic Protection Areas (BPAs) have been designated (15 November 2007, Ministry of Fisheries 2007).

Seamounts are also effectively prevented from being fished by technical restrictions. At present fishing is limited to seamounts (or portions of seamounts) above 1200 m water depth by the difficulties of deploying fishing gear onto a small seamount target below this depth (in contrast to slope areas where gear can be deployed more successfully). Thus, the many seamounts (about 400, see Rowden et al. (2005)) which have a *peak below 1200 m* are also not vulnerable to fishing, while those with a *proportion of seamount below 1200 m* are only partially vulnerable to such a disturbance.

Should fishing on seamounts become possible at greater depths in the future, then clearly the limits of this measure will need to be adjusted accordingly.

The degree to which a seamount is vulnerable to fishing is in part related to its distance from a major fishing port. The cost of fuel (and other costs associated with time spent at sea) is likely to deter vessels from fishing seamounts that are far from their home port. Thus, the *distance from the continental shelf*, a measure in version 1 of the SEAMOUNT database, can be used as a proxy measure of the vulnerability of a seamount to fishing.

Seamounts will be particularly vulnerable if they are a specific target of the fishery. An index to describe the relative importance of a seamount to the fishery has already been devised, which includes measures of fishing effort (number of years fished, number of tows) and catch (total catch over time of the target species). This index is called the *Fishing Importance Index* (or FII) (Clark & O'Driscoll 2003). Thus, the FII is a measure of the vulnerability of a seamount to fishing disturbance, under the assumption that what has been a target for fishing will continue to be a target. In the future, as the means to assign fishing effort to individual seamounts improves, and the index is calculated for those smaller seamounts as yet unfished, the utility of this measure to assess vulnerability will be significantly enhanced.

5.2 Recoverability

After a disturbance event, or a succession of such events, a species, community, or habitat may never recover, recover quickly (days to years), or take a great deal of time (decades to hundreds of years) to obtain its prior status (Hall 1994). With disturbance caused by fishing in shallow/shelf waters, estimates of community recovery range from days to decades, depending on a number of variables, including the type of species present, which will in part be related to the type of substrate on which the fishing has taken place (see for example studies reported in a review by Kaiser et al. (2002)). The information available suggests, for example, that communities of sand habitat will take less time to recover than those of mud or hard substrate with emergent structural fauna (Collie et al. 2000, Dernie et al. 2003, Kaiser et al. 2006). Thus, the type of substrate, or proportion of different substrate types on a seamount, could potentially be a useful indicator and measure of recoverability. However, the results of the previously cited works indicate that the relationship between substrate type and recovery is not straightforward and therefore the general usefulness of the aforementioned indicator/measure is open to question. As already noted (see Section 3), information for substrates on seamounts is relatively scarce so as yet no such indicator or measure can be developed.

Estimates of the time it takes for an individual benthic organism, species population, or community to recover from fishing are not generally available for the deep-sea, let alone for seamounts (see Kaiser et al. (2006) for meta-analysis of intertidal and shelf habitats). However, some New Zealand region-specific information is available for some key structural species that occur on seamounts. Recent research on octocorals indicates that radial growth rates are in the order of 0.18 mm/yr and that it might take an individual colony over 40 years to grow to maturity following damage or death (Tracey et al. 2007). Thus, the *presence of generally long-lived corals* on a seamount (in the context of the SEAMOUNT (v2) database, data in the field 'structure-forming corals') can act as a useful indicator of recoverability of a seamount community (e.g., "very low/none = Partial recovery is only likely to occur after about 10 years and full recovery may take over 25 years or never occur", MarLIN website). It is possible with the wholesale removal of structural species such as corals (or ecosystem engineers *sensu* Jones et al. (1994)) from a habitat that conditions will change to the extent that these organisms are highly unlikely to successfully recolonise to any great degree, and the community that eventually develops will be different from the one that was present before the disturbance (a so-called alternate or alternative stable state or community, see Knowlton (2004)).

More information is needed on the growth and regeneration rates of key structural and functional/characterising species on seamounts, and for the communities to which they belong. With the recent initiation of the monitoring of recovery on Morgue seamount (Graveyard seamount complex, Chatham Rise – MFish project ENV2005/16) following the cessation of fishing in 2001, there is the future possibility of being able to assess how long individual species, and also communities and habitats, of one seamount take to recover from a relatively moderate-heavy (an estimated 800 tows over 7 years of fishing) amount of fishing (Clark & O'Driscoll 2003). These data can then be used to make more robust assessments of the recoverability of seamount biota from fishing, which can then be used to devise further indicators and measures.

The ability of a population of a species or a community to recover may be dependent upon the reproductive capacity/dispersal capability of the species itself. Among the reproductive strategies adopted by species that inhabit seamounts are those that are dependent upon producing large numbers of eggs/larvae which can be widely dispersed by currents, or those that rely upon a relatively small number of offspring that are expected to disperse only short distances from the originating adults. The limited information available for benthic invertebrates of seamounts suggests that the latter strategy predominates (Parker & Tunnicliffe 1994). The consequence of a limited dispersal capability is likely to mean that populations of species will take a relatively long time to recover after a disturbance that removes a significant proportion of the adults (Reed et al. 2000). Therefore the proportion of species in a community that have limited dispersal capability could act as a useful measure of recoverability. However, at present there is next to no information about the reproductive biology and ecology of species found on seamounts in the New Zealand region. Generalised information about reproductive strategies has been collated (D. Stevens et al. NIWA, unpublished results), but it is not possible to use this proposed measure at this moment.

The limited data on reproductive strategies suggest that limited dispersal predominates among macroinvertebrate species that inhabit seamounts with associated Taylor Caps (Parker & Tunnicliffe 1994). These closed circulation features, which are a product of seamount morphology, depth, and strength of the surrounding oceanographic regimes/water flow, are likely to affect the dispersal and recruitment capability of species (see also Section 3). Therefore, the *probability of Taylor Cap formation* (two measures in database) could be used as a proxy indicator of limited dispersal capability of benthic invertebrates, and thus recoverability.

The degree of genetic isolation is also a potentially useful indicator/measure of a population's ability to recover from a disturbance. The more genetically isolated (low genetic variation), the longer the time it will theoretically take for the population to recover from a disturbance (see introduction of Johannessen & Andre (2006)). Unfortunately, at present there is very little information concerning population genetics of macroinvertebrates found on seamounts in the New Zealand region (but see for example Smith et al. (2004a)), and therefore measures of genetic isolation are not currently practical. However, the amount of genetic information for macroinvertebrates, particularly corals in the region, is beginning to increase (Miller et al. 2006).

The ability of a habitat or community (and its component species) to recover from a disturbance is in part dependent upon the extent of the disturbance. For fishing, the magnitude (including intensity), frequency, and duration of the disturbance are important (Kaiser 1998). These factors have been taken into consideration in the development of the *Fishing Effects Index* (FEI) for seamounts in the New Zealand region (O'Driscoll & Clark 2005) The FEI combines data for total distance of trawls and the number of directions towed (standardised for seamount area), and therefore captures in a single metric a measure of the potential for habitat/communities/species to recover, for those seamounts in the region already known to have been fished. In the future, as the means to assign fishing effort to individual seamounts improves, and the index is calculated for those smaller seamounts as yet unfished, the utility of this measure to assess recoverability will be significantly enhanced.

5.3 Representativeness

Characterising an environment, habitat, and its associated biota in the sublittoral component of the marine realm is not a straightforward task, but one that has often been attempted using a variety of methods. The methods used range from those based upon qualitative appreciations of direct (and often very limited) observation to those which analyse quantitative data and sometimes employ some form of prediction or model, although attempts have been made to standardise approaches (Madden et al. 2005). In lieu of obtaining sufficient biological data, marine environments/habitats are often spatially characterised by a number of physical variables that are known (or are thought likely) to influence the type of biotic community that would/could develop at a particular location (Roff & Taylor 2000). The number of these types of classifications that exist for the marine environment throughout the world is growing as their practical utility is realised, in particular for helping to select suitable locations for marine protected areas (Roff et al. 2003). Such classifications allow for a measure of the degree of representativeness of a particular environment/habitat, and by inference its associated community. These classifications are designed to operate on a single or sometimes variety of spatial scales, and their use in management initiatives should be appropriate to these scales.

For the New Zealand region there are two relatively robust, general classifications. The first classifies physical/oceanographic variables that describe the marine environment from the sea surface to the seafloor using data scaled to a common resolution of 1 km^2 (but based on data collected at smaller and larger scales) (Snelder et al. 2006). The second classifies physical/oceanographic (and one biological variable) variables as they relate to individual seamounts (about half the known seamounts of the region) (Rowden et al. 2005). These classifications have been used to identify self-similar areas/units. The first, known as the Marine Environment Classification (MEC), can operate at a number of classification levels (from 2 to 290) which differentiate the environmental space into smaller and smaller units, while the second identifies 12 seamount groupings which exhibit some geographical affinity. The MEC was validated using biological data (but not tested), but while it appears to be reasonably good for representing the environments potentially occupied by different types of fish assemblages, it does not appear to be particularly good at representing those potentially occupied by different assemblages of benthic invertebrates (see figure 6 in Snelder et al. 2006). The seamount classification has not been validated nor tested using biological data, so at present it is uncertain whether the groupings identified correspond to seamounts possessing distinct benthic assemblages. Nonetheless, despite the limitations and uncertainty surrounding these two classifications, they can, at present, either separately or together (e.g., a seamount group within an MEC class), be used to measure the *degree of seamount representativeness*. In the future, both the MEC and the seamount classification will be improved and, hopefully, proved by testing to be useful for defining representativeness. A demersal fish-optimised MEC has already been developed (but not implemented) (Leathwick et al. 2006), and a benthic-optimised MEC is currently being constructed under MFish project BEN2006/01. A second seamount classification will be undertaken (using data in SEAMOUNT v2) as part of NIWA's Seamount Programme. This seamount classification will also be tested using comparable data from 40 seamounts in New Zealand waters (within the current lifetime of NIWA's Seamount Programme).

Macroinvertebrate data from 40 seamounts sampled as part of NIWA's Seamount programme are the best available data for identifying the types of communities present on seamounts in the region.

However, these data could only be used to directly derive a measure of a community's representativeness, both within and between seamounts, for a very small proportion of seamounts. It is highly unlikely that sufficient data will be collected from seamounts in the region to define such a community measure in the near future. In the absence of such data, it might be possible to gain suitable proxies using surrogate information on the different substrates and sedimentary conditions found on seamounts (Post et al. 2006). However, as noted earlier (Section 3), information for sediment types on the scale of a seamount (let alone within a seamount) is relatively scarce, making measures of representativeness based upon substrate type unattainable. However, with the increasing amount of multibeam data that is being collected for the region (e.g., as part of FRST-funded programmes at NIWA, Ocean Survey 20/20), and the recent work on relationships between backscatter data and seabed substrates (Le Gonidec et al. 2003) and habitat (Durand et al. 2006), and macroinvertebrate diversity (Rowden et al. unpublished results) it may be feasible in the future to use backscatter derived substrate data as some measure of representativeness.

6. DEVELOPING A RISK ASSESSMENT METHOD

The SMEEF, as part of its implantation plan, calls for the development of a risk assessment methodology. While the SMEEF provides an outline of the sort of overall process envisaged for assessing "the risks of adverse effects of fishing" (see figure 2, p 5, Ministry of Fisheries 2005), it does not indicate what sort of process is imagined for prioritising "species and habitats for development of environmental standards" (section 4.1, p 20, Ministry of Fisheries 2005). Therefore, to help develop a preliminary risk assessment method/model for seamounts (which will be advanced further under MFish project ENV2005/16 and the NIWA Seamount Programme), after a brief background on ecological risk assessment, the appropriateness of a number of different methods/models is reviewed below.

6.1 Conducting ecological risk assessment

Risk assessment is a technique that is relatively common in fisheries management to reduce the risk of undesirable events (Francis & Shotton 1997). Ecological risk assessment is beginning to become a component of risk management where, for example, responsibilities extend to managing fisheries at an ecosystem-level (Link et al. 2002). There are many methods by which ecological risk can be assessed. Some are designed to be **general assessment frameworks**, which can be used with or without minor modification to accommodate a specific risk (e.g., U.S. Environmental Protection Agency framework for ecological risk assessment, EPA (1992)). Such frameworks are the basis for 'expert' decision-making, albeit sometimes using quantitative data and a Delphic process (a structured process that aims to reduce subjectivity and remove 'group-think'). New Zealand is reasonably well advanced in applying the theory of risk management, with the production of a general standard: Australian/New Zealand Standard for Risk Assessment (Australian/New Zealand Standards, 1999). Despite this standard and its successful implementation in a marine fishery context (shellfish farming in Tasmania, Crawford (2003)), and its most recent adaptation for assessing relative risk of different fisheries in a New Zealand context (Campbell & Gallagher 2007), it is wise to consider other risk assessment methods. It has been suggested that the "best ecological risk assessments are the ones that are appropriate for the specific risk management needs of the individual site", i.e., it is important to select the 'right tool for the job' (Sorensen et al. 2004). Some risk assessment methods involve the calculation of specifically developed indices (e.g., for sensitivity and vulnerability). Zacharias & Gregr (2004) used such indices and demonstrated how they can be used to identify and spatially map areas off the west coast of Canada in which whales are vulnerable to disturbance. Such indices can be calculated by different means, including the use of so-called **fuzzy logic expert systems** (e.g., Cheung et al. 2004) which have, for example, been used to identify the "intrinsic" vulnerability of seamount fish to fishing (Morato et al. 2004). The latter method allowed for measures of uncertainty about the assessment to be made, an advantage which can also be incorporated into other models of risk assessment (e.g., the Relative Risk Model and the use of Monte Carlo analysis, Hayes & Landis (2004)). More recently, Ramsey & Veltman (2005) used two **qualitative modelling** approaches to predict the effects of predator control on the fledgling success of kokako. The first approach used loop analysis to predict the direction of change in species abundances; the second, fuzzy web interactions, based on fuzzy set theory, was used to predict the magnitude of change in species abundances. The authors found these tools, when combined, were suitable for predicting the effects of perturbations in complex ecological communities and suggested that such predictions can be incorporated into a risk assessment process. Most recently of all, Hiddink et al. (2007), in order to address some of the perceived disadvantages of some of the above methodologies, developed a relatively simple **sensitivity model** for determining two measures for this concept of risk which they believe offers the most robust means to assess the ecological effect of bottom trawling on benthic habitat and communities.

6.2 Choosing an Ecological Risk Assessment tool/method

6.2.1 General assessment framework

The Australian/New Zealand Standard for Risk Assessment (Australian/New Zealand Standards, 1999) sets out a general framework which is applicable to a wide range of industries and activities (Figure 3). This process generally involves identifying, analysing, evaluating, and treating risks. It can involve environmental, social, and economic aspects. "Risk" is defined as "the likelihood of an undesired event occurring as a result of some behaviour or action (including no action)" (Hayes 1997). This type of general assessment framework is very similar to the one erected and used by MarLIN to assess ecological sensitivity (Figure 4), but without the 'treatment' phase – i.e., the part of the process that involves management decision and action. In the context of MarLIN, that part of the process is undertaken by environmental management agencies. In the SMEEF context, the assessment of risk to 'habitats and species' informs the treatment/action phase that is represented by fisheries management initiatives that are part of the overall SMEEF framework (figure 2, section 2.1, p 5, Ministry of Fisheries 2005).

The analysis/evaluation component of the Australian/New Zealand Standard involves an assessment of the *likelihood* and *consequences* of an event having an impact. Broad categories or levels of likelihood and consequences can be erected appropriate to the degree to which a qualitative assessment can be confidently made. For example, the consequences of fishing activity for the benthos could be considered as: 1 - Insignificant, 2 - Minor, 3 -Moderate or 4 - Major; while likelihood categories described by the standard are: A - Almost certain, B - Likely, C - Moderate, D - Unlikely and E - Rare. Appropriate measures and indicators of risk are used to assess consequence and likelihood, and combined into a qualitative risk analysis matrix, which ranks levels of risk from Low, through Moderate and High to Extreme (Table 3). This type of qualitative risk assessment matrix is also used by the MarLIN scheme (Table 4), where overall level of risk equates to level of sensitivity (see also earlier explanation, Section 4) whereas likelihood and consequence equate to intolerance (or vulnerability) and recoverability, respectively.

The procedure and measures/indicators by which intolerance and recoverability are assessed by the MarLIN sensitivity scheme broadly described by Hiscock & Tyler-Walters (2006) and provided in detail on the MarLIN website. These documents also assign definitions to the various categories (i.e., levels) of sensitivity (i.e., risk), e.g., "Very high" sensitivity is indicated by the following scenario: The habitat or species is very adversely affected by an external factor arising from human activities or natural events (either killed/destroyed, 'high' intolerance) and is expected to recover only over a prolonged period of time, i.e., over 25 years or not at all (recoverability is 'very low' or 'none'). The habitat or species is adversely affected by an external factor arising from human activities or natural events (damaged, 'intermediate' intolerance) but is not expected to recover at all (recoverability is 'none').

Campbell & Gallagher (2007) published a method for assessing the risk of the effects of fishing on the environment that also uses the general assessment framework. They present what they call a "semi-quantitative risk analysis model" specifically for the management of New Zealand fisheries. The method is based directly on the general framework approach of the Australian/New Zealand Standard risk management. Five ecological categories associated with the effects of fishing are identified – non-target species, biodiversity, habitat, trophic interactions, protected species – for each of which the likelihood and consequence of fishing disturbance is evaluated. The method adopts the likelihood categories as proposed by the Australian/New Zealand Standard. Specific indicators/measures and definitions, to support assessment against five levels of consequence ('insignificant' to 'significant'), are provided in the methodological outline. The various indicators/measures include what the authors call "thresholds" (e.g., "minor consequence: reductions in protected species population abundances are less than 1%"), which are in part derived from legislative and policy obligations. The assessment process involves both scientific experts and stakeholders (using questionnaires and working groups) and threshold values can be adjusted during this consultation. Subsequently a risk matrix is constructed as per the risk assessment standard, but the method also details a risk ranking procedure which is directed at prioritising science-related actions (e.g., "High risk: possible increases to scientific activities required"). This step is part of the 'treatment' phase of risk assessment, and is probably incorporated by Campbell & Gallagher (2007) because they appear to favour a close integration of science and management. While the authors illustrate the potential of their adaptation of the general assessment framework using the orange roughy fishery, the method "does not evaluate ecosystem risk directly", but rather *relative* risk posed by different fisheries. Nonetheless, Campbell & Gallagher (2007) expressed the hope it will at least help environmental managers prioritise their actions concerning various ecological issues that result from particular types of commercial fishing.

Such a general risk assessment method can incorporate quantitative data, but the process is largely qualitative and involves some subjective assessment by 'experts'. Nonetheless, the output from such a process is a readily understandable assessment of risk. The simplest means to aid the evaluation of this risk by environmental managers is to produce a visual representation of the results of the risk analysis. This can, for example, be achieved by mapping the risk values as a GIS data layer over other layers that might display for instance the geographic and areal distribution of a habitat or habitats of interest. The sensitivity assessments undertaken by MarLIN to date have been illustrated in this manner to good effect (e.g., for hydrocarbon contamination, see figures 16 and 17 in Tyler-Walters & Hiscock (2005)). In the context of present interest in assessing ecological risk to seamounts from fishing, it is easy to envisage the almost wholesale transferability of the MarLIN approach (including linkage to a facilitating database such as SEAMOUNT), which could culminate in the production of a map of the New Zealand region displaying sensitivity values for individual seamounts.

The advantages of a general risk assessment framework method such as represented by the Australia/New Zealand Standard and illustrated in practice by the MarLIN scheme are: (1) it is a relatively simple procedure, (2) commensurate with the quality of data currently available, (3) experience of other similar schemes can be used, and (4) allows for a degree of comparability with results from other schemes. The disadvantages are: (1) it involves 'expert' opinion and therefore subjectivity, (2) it is relatively inflexible, and (3) it does not incorporate a means to deal with uncertainty.

6.2.2 Fuzzy logic expert systems

The ability to classify or mathematically calculate ecological impacts in the marine environment hinges on the quality and quantity of available data. As mentioned in earlier sections, acquiring lots of robust data in the marine environment is costly and time-consuming. Fuzzy set theory, developed by Zadah (1965), provides a way of processing imprecise information and incorporating expert knowledge into a classification scheme. Fuzzy logic takes variables for which there is only 'expert' knowledge, or limited quantitative information, and allows the setting of fuzzy thresholds or boundaries between true or false rules. Conventional Boolean sets classify variables as either true or false; fuzzy logic classifies variables through a graduation of membership (Salski 1992, Silvert 1997).

Fuzzy set theory classifies variables according to their fuzzy membership functions. For example, Figure 5 from Cheung at al (2004) illustrates the output fuzzy sets for a single life history characteristic (age at first maturity) in their analysis of intrinsic vulnerability of seamount fishes to fishing. Using known relationships between age at first maturity and intrinsic vulnerability the authors transformed this attribute into a linguistic category. A fish at age 4 years, for example, has 0.5 membership in both medium and high intrinsic vulnerability categories. A fish at age 5.5 years has 0.3 membership in the high category and 0.7 in the very high category. At age 7 years or greater a fish has 1 (100%) membership in the very high intrinsic vulnerability category. These membership functions need to be determined by the available 'expert' knowledge using published data or data collated, specifically for the purpose, in an appropriate database such as SEAMOUNT (v2).

To create a fuzzy logic model a number of stages have to be followed. First, it is necessary to determine the model structure, the input and output variables, and the linguistics terms to be used, i.e., high, moderate, low risk. Then it is important to formulate the knowledge base; that is, where will the data come from; what expert knowledge is available; and which publications and databases can be used to inform the expert decision making. Every variable in the database then needs to be weighted or ranked in order of importance to the risk assessment, and fuzzy sets for every variable need to be defined, as in Figure 5. Next the fuzzy logic processing methods to be used are chosen, and once processing begins the model will need to be calibrated (tweak fuzzy sets) and then finally validated (Cheung et al. 2004).

Fuzzy logic processing methods generally follow the IF-THEN form. That is, IF A THEN B, where A is the premise and B the conclusion which may lead to other rules (Cheung et al. 2004). For example, using a variable from SEAMOUNT (v2) we might say;

IF *depth at base* = < 600 m THEN Risk is **High**

IF *depth at base* = 601–1200 m THEN Risk is **Moderate**

IF *depth at base* = > 1201 m THEN Risk is **Low**

More variables or rules are added during the processing to produce a range of conclusions, and ultimately a single point output for all rules, such as HIGH RISK. For example;

IF *depth at base* = < 600 m, AND *structural species* = present, AND *distance to shelf* = < 100 km, THEN Risk is **HIGH.**

While some degree of subjectivity is unavoidable in fuzzy set theory, the resulting outputs, be they linguistic or numerical, are calculated from the predetermined memberships in the fuzzy sets and the weighting assigned to each variable (see Cheung et al. (2004) for details of mathematical functions).

The advantages of using a fuzzy logic expert system are: (1) the linguistic categories can be defined in whatever way is most suitable, (2) it allows incorporation of information from a wide range of sources, (3) the predictive ability of the system can be increased all the time, (4) it can work for variables with different data availability, (5) it can adapt to new information from either qualitative expert knowledge or quantitative studies, and (6) once the data are compiled in a central database, various interpretive reports can be generated for difference purposes. The disadvantages of fuzzy logic expert systems are: (1) it can be difficult to estimate or agree on membership functions where this relies on expert knowledge, and (2) there are many ways of interpreting fuzzy rules, combining the outputs of fuzzy rules, and defuzzifying the output.

6.2.3 Qualitative modelling

The dynamics of marine ecosystems are driven by a complex series of interactions, including those between species, between species and their environment, and human impacts on the marine environment. However, there is a lack of quantitative knowledge of the interactions between many of these components, and the complexity of the interactions makes the overall effect of one component on others unclear. This makes qualitative modelling techniques attractive for understanding complex marine ecosystems, and potentially a means by which to assess the risk that a disturbance such as fishing has for habitats, communities, and species.

Food webs are one example of a qualitative modelling technique used in ecological research (Whipple et al. 2000). One way to represent a food web is as a signed community matrix with entries of (-, 0, +) representing the signs of species interactions. This type of representation emphasises the structure of interactions, instead of the often unknown quantitative values of interactions. Levins (1974) initiated a qualitative modelling technique known as loop analysis for analysing the stability of communities represented by signed community matrices, and for predicting the direction of changes in abundance of species to perturbations to the community. However, for large matrices (more than five species) or highly connected communities loop analysis techniques can be cumbersome and the results difficult to interpret. Loop analysis has been reformulated in terms of matrix algebra, and the interpretation of the results simplified with additional techniques (Dambacher et al. 2002, 2003a, 2003b).

A simple plankton community model is used here to illustrate the qualitative analyses techniques involved (Stone 1990, Dambacher et al. 2002). The signed digraph representing the food web and interactions for this community is show in Figure 6. Table 4 shows the associated signed community matrix with positive interactions represented by 1, negative interactions by -1, and neutral or ambiguous interactions by 0. From the signed community matrix the so-called adjoint matrix and weighted predictions matrix can be calculated, these two additional matrices being the key to a qualitative understanding of the community.

The adjoint matrix gives the predicted direction of response of quantities to sustained positive changes in other quantities (Table 5). The entries are the totals of all feedback cycles that led from one quantity to another, taking into account their sign. Entries can be negative, but in the plankton model none are. Entries of zero can occur when there an equal number of positive and negative feedback cycles going from one quantity to another, in which case the direction of a response will depend strongly on the actual value of interactions. This is the case with the response of zooplankton, phytoplankton, and nutrients to an increase in the abundance of bacteria. In contrast, an increase in the abundance of nutrients is predicted to give an increase in all other quantities.

The weighted predictions matrix has entries that are between 0 and 1 (Table 6). It measures the extent to which the predictions from the adjoint matrix are determinate: a value close to zero indicates that the predicted direction of change is very dependent on the quantitative value of interactions, while a value of one indicates that the predicted direction of change is independent of the quantitative value of interactions. Simulation studies indicate that entries that are greater than or equal to 0.5 give about 90% probability of obtaining the correct direction of change. For the plankton model example, most of the entries are greater than or equal to 0.5. In particular, the responses to a nutrient increase are positive, independent of the quantitative value of interactions. In ecological systems it is rare that actual interaction values between community members or ecosystem components are known, and in some instances they are exceedingly difficult to measure (Dambacher et al. 2003a). The composition or structure, however, may be well known and can be encompassed by a qualitative approach, and thus all that is needed is an understanding of the direction of interactions between ecosystem components.

Qualitative modelling is not an end in itself; rather it can use imprecise information to generate testable hypotheses about ecosystem responses to disturbance. In other words, qualitative modelling should be seen as a precursor to environmental risk assessment methods, highlighting the essential model components, the direction of their interactions, and the strengths or weaknesses of supporting data (Dambacher et al. 2003a, Ramsey & Veltman 2005). The advantages of qualitative modelling are: (1) that quantitative values need not be known, (2) biological groupings can be specific (the coral species *Solenosmilia variabilis*) or general (corals), and (3) and the model can incorporate non-biological variables, such as fishing pressure, or management decisions. The disadvantages are that: (1) the model predicts only the direction of changes, not the magnitude, and (2) the analysis assumes the community starts in an equilibrium state to which a perturbation is applied.

6.2.4 Sensitivity model specific to seabed habitat and fishing

Hiddink et al. (2007) have produced a novel method that models sensitivity of seabed habitat to disturbance by bottom trawling. They term the component concepts of risk resistance and resilience, which are synonymous with the previously discussed terms vulnerability and recoverability. The method relies upon information for the recovery time of production or biomass of benthic invertebrate communities which is predicted using a size-based model (Duplisea et al. 2002) that incorporates the effect of natural disturbance (see related work by Hiddink et al. 2006a, 2006b, 2006c). The method has been applied to data from the North Sea (at a scale of 9 km^2) and the measures of sensitivity have been mapped to provide environmental managers with what would appear to be a very robust tool with which to assess the risk of the adverse ecological effects of fishing. The method allows managers to "predict the implications of changing patterns of human impact on seabed habitats when establishing spatial management plans" (Hiddink et al. 2007). The authors claim that their method can be readily developed and applied to other situations and areas.

The advantages of this method are: (1) it is a quantitative and objective technique, not reliant on qualitative scoring by 'expert' opinion, (2) the model has been validated, (3) results can be used to make quantitative predictions of the effects of different management scenarios for fishing, and (4) data for most of the model parameters are available. The disadvantages are: (1) that not all data necessary for parameterising the model are currently collated at the scale of a seamount (e.g., benthic macroinvertebrate biomass – but some presumably available from bycatch records).

7. CONCLUSIONS

The addition of 41 new data fields makes the SEAMOUNT (v2) database a significant improvement on its predecessor. This development has been possible through the collaborative nature of the pooling of resources provided by the present MFish project and NIWA's Seamounts Programme. Although there is still a need to further advance the SEAMOUNT database, the potential utility of the database as a management tool is now considerably enhanced, in particular to support the type of ecological risk assessment envisioned by MFish's SMEEF. Data and information contained within the database can be used to provide indicators and measures of risk (some as suggested by this report) to habitats, communities, and species potentially threatened by fishing activities on seamounts.

A number of methods available for the task of risk assessment were reviewed as part of this project, and each was shown to have particular advantages and disadvantages, with arguably no one method emerging as a 'clear winner'. Overall, it is thought worthwhile to continue the preliminary evaluation of these and other methods (particularly those of Hiddink et al. (2007), Campbell & Gallagher (2007), and the recently published CSIRO-AFMA report) before deciding on which particular ecological risk assessment method should be developed further for application to New Zealand seamount management.

8. RECOMMENDATIONS

SEAMOUNT database

- Add more data fields (e.g., benthic biomass estimates using bycatch data, seamount morphology derived from automated approach, threatened species, connectivity indices, dissolved oxygen) as part of existing research projects where possible or new projects if required.
- Improve interactive capability through, for example, inbuilt GIS web-map functionality.
- Enhance linkage with other databases under NIWA's marine databases initiatives (including direct access to species occurrence data for seamounts).
- Determine a strategy for ensuring that the database is maintained in the long term and regularly updated.
- Develop a new project to implement the above four recommendations.

Ecological Risk Assessment method

- Hold a workshop to further explore the usefulness of the proposed indicators/measures of risk and how to best progress the preliminary development of an appropriate ecological risk assessment method for seamounts. Such an exploration, when attempting to determine which particular indicators/measures and method should be adopted, will need to be fully aware of the sorts of data currently available in the SEAMOUNT (v2) database (and likely to be available), and guide which sorts of data could be added without too much additional effort (e.g., benthic biomass). This workshop should involve governmental and industry stakeholders, as well as scientists from a range of disciplines and countries to evaluate different approaches.

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Table 1: Summary of indicators and measures which could be useful for assessing ecological risk (i.e., sensitivity) to seamounts posed by bottom trawling.

Now (data available via SEAMOUNT v2)	Near future (1-5 years)	Distant future (>5 years)
Vulnerability		
Presence of habitat-forming species (corals and sponges)	Predicted occurrence of habitat-forming species (corals & sponges)	Predicted occurrence of habitat-forming species (corals & sponges & other taxa)
Presence of vent mussels and venting	Occurrence of characterising species of seamount communities & community types	Predicted occurrence of characterising species of seamount communities & community types
	Presence of threatened species (from list in Hitchmough et al. 2007)	Presence of threatened species (updated/amended list)
	Presence of legally protected species (revised list)	Presence of legally protected species (updated/amended list)
Seamount is legally protected (designation as of 2007)	Seamount is legally protected (updated designation)	Seamount is legally protected (updated designation)
Seamount peak below 1200 m	Seamount peak below ???? m (adjust with technological advances)	Seamount peak below ???? m (adjust with technological advances)
Proportion of seamount below 1200 m	Proportion of seamount below ???? m (adjust with technological advances)	Proportion of seamount below ???? m (adjust with technological advances)
Distance to continental shelf	Distance to major fishing port	
Fishing Importance Index	Fishing Importance Index (improved version II)	Fishing Importance Index (improved version III)
Recoverability		
	General type of substrate	General type of substrate (improved data)
	Proportion of different types of substrate	Predicted proportions of different types of substrate
Presence of generally long-lived corals	Growth rates of structural species present	Growth rates of functional species present
	Indicative recovery rates of some types of seamount communities (spatially restricted)	Indicative recovery rates of many different types of seamount community (unrestricted)
	Proportion of species in a community that have limited dispersal capability (qualitative indication)	Proportion of species in a community that have limited dispersal capability (quantitative measure)
Probability of Taylor Cap formation (2 measures)	Probability of Taylor Cap formation (validated models)	Probability of Taylor Cap formation (validated & tested models)
	Degree of genetic isolation exhibited by a population of a species (few key species)	Degree of genetic isolation exhibited by a population of a species (large no. of key species)
Fishing Effects Index	Fishing Effects Index (improved version II)	Fishing Effects Index (improved version III)

Table 1 (Continued)		
Now (data available via SEAMOUNT v2)	Near future (1-5 years)	Distant future (>5 years)
Representativeness		
Degree of environmental representativeness (proportion of seamounts in same MEC class – level 20)	Degree of environmental representativeness (proportion of seamounts in demersal fish and/or benthic optimised MEC class – level ??)	Degree of environmental representativeness (proportion of seamounts in demersal fish and/or benthic optimised MEC class – level ?? - tested)
Degree of environmental representativeness (proportion of seamounts in same MEC class – level 33)	Degree of environmental representativeness (proportion of seamounts in demersal fish and/or benthic optimised MEC class – level ??)	Degree of environmental representativeness (proportion of seamounts in demersal fish and/or benthic optimised MEC class – level ?? - tested)
Degree of seamount representativeness (proportion of seamounts in seamount class)	Degree of seamount representativeness (proportion of seamounts in seamount class – tested)	Degree of seamount representativeness (proportion of seamounts in seamount class – improved & tested version)
	Proportion of different types of substrate	Predicted proportions of different types of substrate
	Predicted levels of biodiversity (based on predicted substrate heterogeneity) – limited spatially	Predicted levels of biodiversity (based on predicted substrate heterogeneity) - widespread

Table 2: The Australian/New Zealand Standard consequence versus likelihood risk assessment matrix. Redrawn from Australian/New Zealand Standards (1999).

	Consequence			
	1	*2*	*3*	*4*
Likelihood	***Insignificant***	***Minor***	***Moderate***	***Major***
A - Almost certain	High	High	Extreme	Extreme
B - Likely	Moderate	High	High	Extreme
C - Moderate	Low	Moderate	High	Extreme
D - Unlikely	Low	Low	Moderate	High
E - Rare	Low	Low	Moderate	High

Table 3: The MarLIN sensitivity assessment matrix. Redrawn from Hiscock & Tyler-Walters (2006). Note that in the context of the concepts of risk used in this report, 'intolerance' equates to 'vulnerability'. Tolerant* and 'Not sensitive*' are indicated for species that might benefit by change in a factor.

	Recoverability						
	None	***Very Low*** >25 yr	***Low*** >10-25 yr	***Moderate*** >5-10 yr	***High*** 1-5 yr	***Very high*** <1 yr	***Immediate*** <1 week
Intolerance							
High	Very High	Very High	High	Moderate	Moderate	Low	Very Low
Intermediate	Very High	High	High	Moderate	Low	Low	Very Low
Low	High	Moderate	Moderate	Low	Low	Very Low	Not sensitive
Tolerant	Not sensitive	Not sensitive	Not sensitive	Not sensitive	Not sensitive	Not sensitive	Not sensitive
Tolerant*	Not sensitive*	Not sensitive*	Not sensitive*	Not sensitive*	Not sensitive*	Not sensitive*	Not sensitive*
Not relevant	Not relevant	Not relevant	Not relevant	Not relevant	Not relevant	Not relevant	Not relevant

Table 4: The signed community matrix associated with the signed digraph in Figure 6, where R= Protozoa, B= Bacteria, Z= Zooplankton, P= Phytoplankton, N= Nutrients. For a given column entry read down the rows to find the direction of an interaction. For example, bacteria have a positive interaction with protozoa, and negative interaction with nutrients. After Dambacher et al. (2002)

	R	B	Z	P	N
R	-1	1	0	0	0
B	-1	-1	0	1	1
Z	0	0	-1	1	0
P	0	0	-1	-1	1
N	1	-1	1	-1	-1

Table 5: The adjoint matrix of the signed community matrix, where R= Protozoa, B= Bacteria, Z= Zooplankton, P= Phytoplankton, N= Nutrients. A sustained positive increase to the quantity denoted in a column label has a predicted effect on other quantities denoted by the row entries below it. After Dambacher et al. (2002)

	R	B	Z	P	N
R	5	2	2	1	3
B	1	2	2	1	3
Z	2	0	4	2	2
P	2	0	0	2	2
N	4	0	4	0	4

Table 6: The weighted prediction matrix for the signed community matrix, where R= Protozoa, B= Bacteria, Z= Zooplankton, P= Phytoplankton, N= Nutrients. Entries with a value less than 0.5 are in bold, and indicate that the response prediction given by the adjoint matrix is indeterminate. After Dambacher et al. (2002)

	R	B	Z	P	N
R	0.7	0.5	0.5	**0.3**	1
B	**0.1**	0.5	0.5	**0.3**	1
Z	1	**0**	0.5	0.5	1
P	1	**0**	**0**	0.5	1
N	1	**0**	0.7	**0**	1

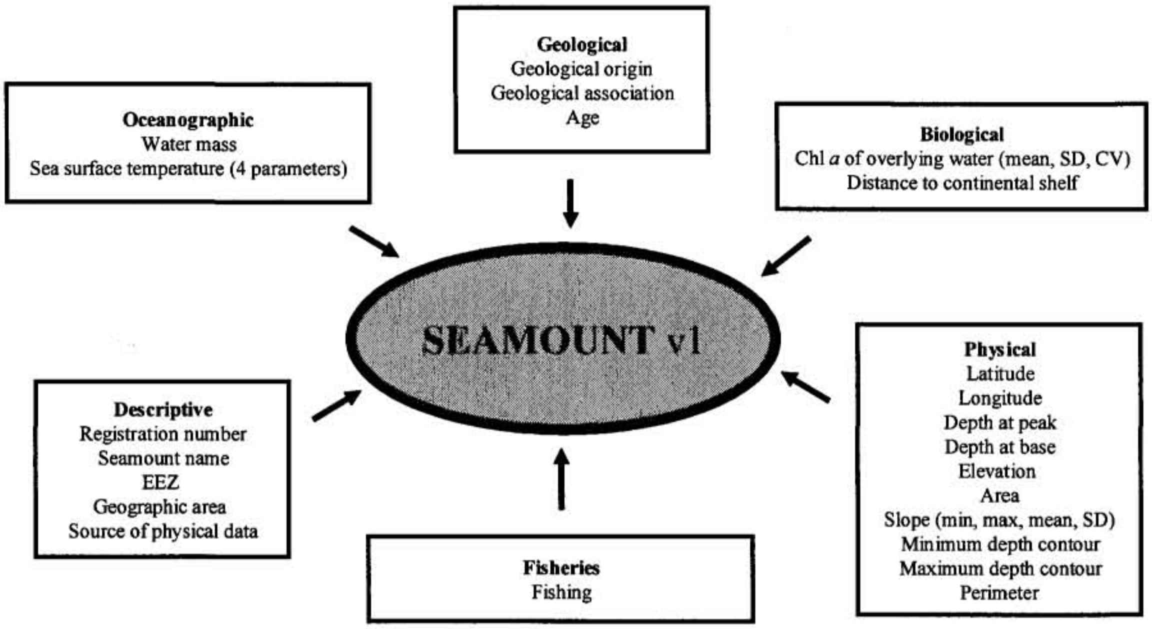

Figure 1: Schematic diagram showing the 31 data fields of the SEAMOUNT (v1) database.

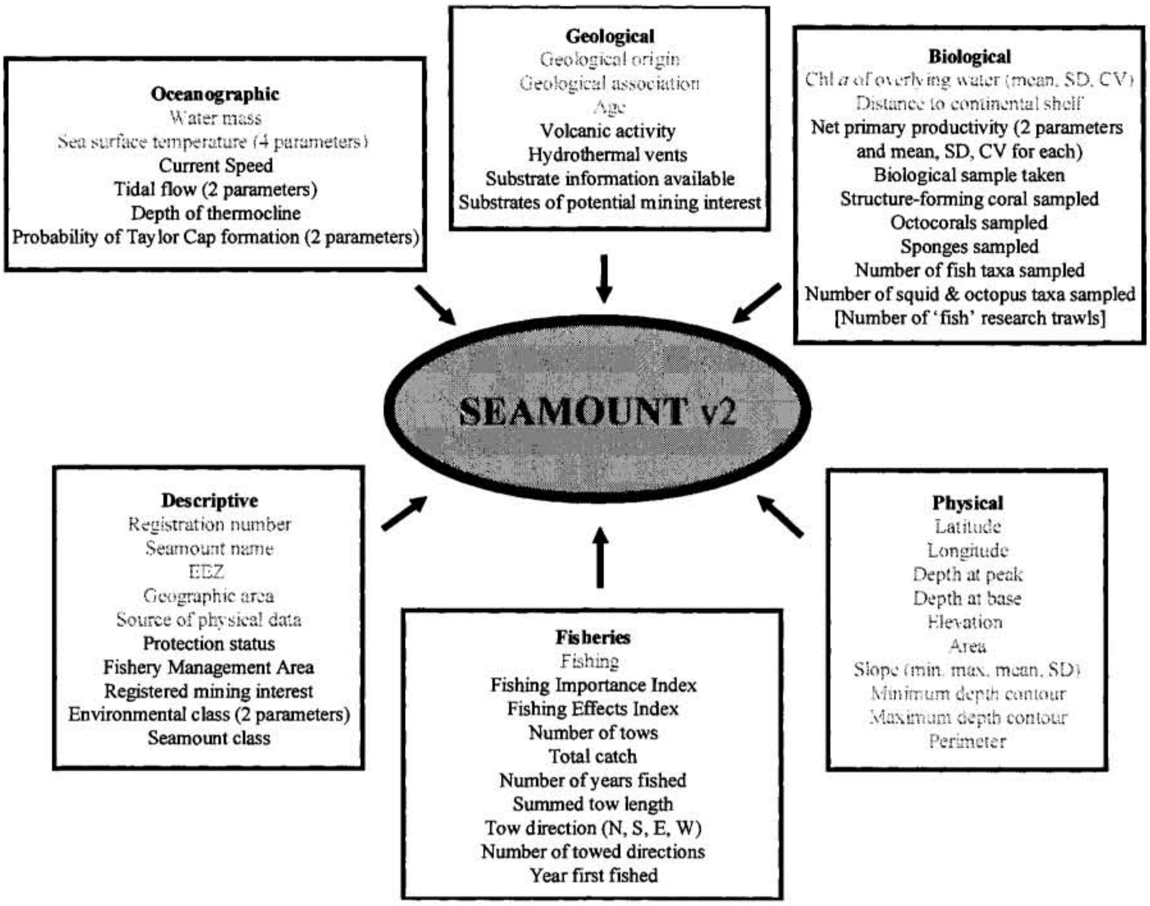

Figure 2: Schematic diagram showing the 72 data fields of the SEAMOUNT (v2) database (the v1 fields shown in grey text, the additional 41 fields shown in black text).

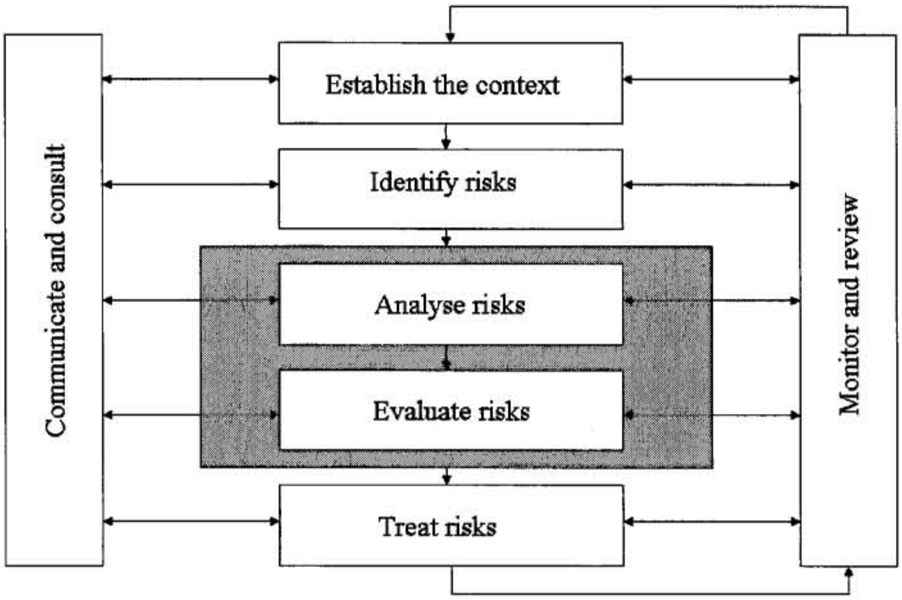

Figure 3: The Australian/New Zealand Standard framework for risk assessment. Redrawn from Australian/New Zealand Standards (1999).

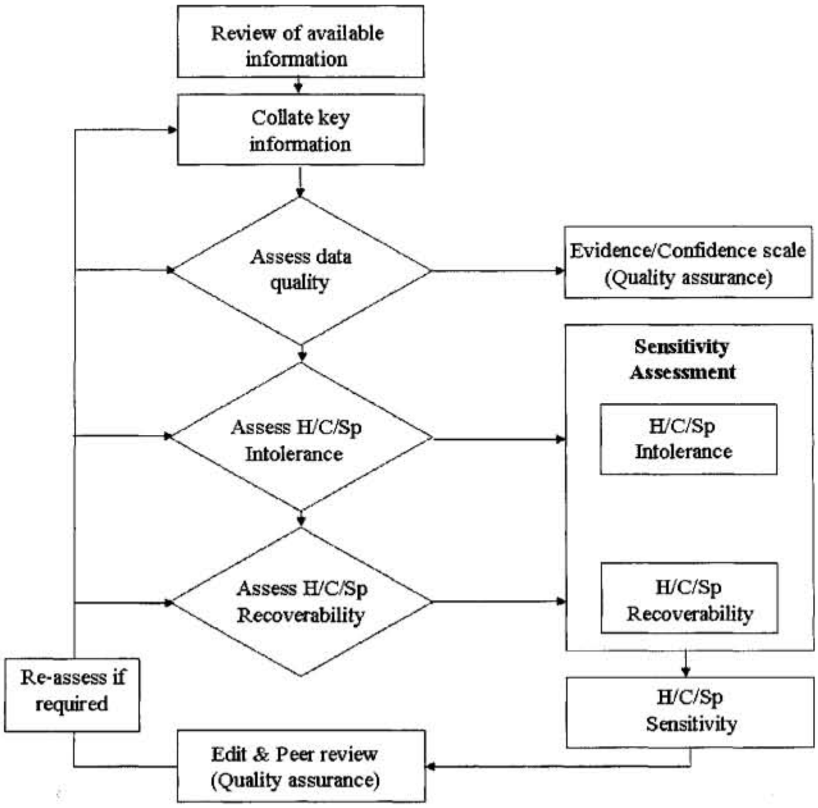

Figure 4: The MarLIN framework for assessing sensitivity (i.e., level of risk). Redrawn from Hiscock & Tyler-Walters (2006). H = Habitat, C = Community, Sp = Species. Note that in the context of the concepts of risk used in this report, 'intolerance' equates to 'vulnerability'.

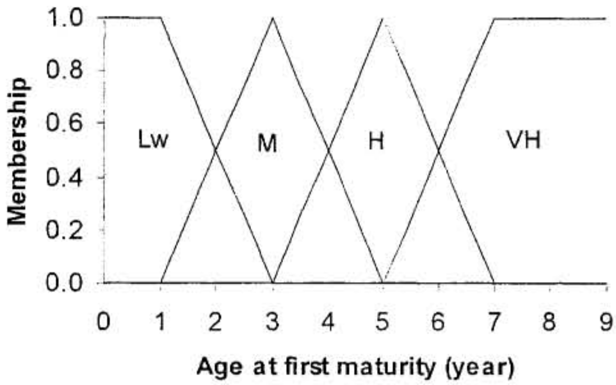

Figure 5: Fuzzy set defining age at first maturity, with "degree of belief" = membership in a fuzzy set. After Cheung et al. (2004). Lw = low, M = medium, H = high, VH = very high membership.

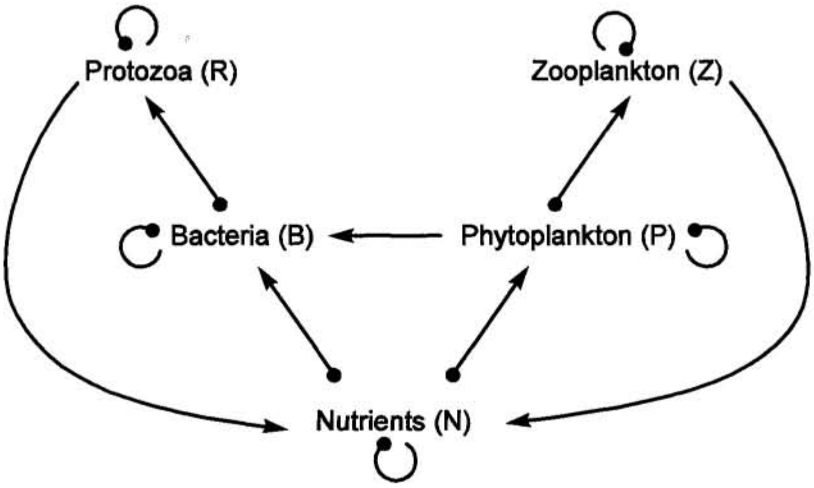

Figure 6: Signed digraph (signed directed graph) for a plankton community model. A line ending in an arrow indicates that the model component it started on has a positive interaction with the model component it ends on (e.g., an increase in phytoplankton abundance leads to more bacteria); a filled circle indicates a negative interaction (e.g., more phytoplankton led to a reduction in nutrients). A line with an arrow at one end and a filled circle at the other indicates a predator-prey relationship (e.g., the protozoa-bacteria interaction). A line that finishes where it starts, with a filled circle at the finish, represents density-dependent negative feedback (e.g., zooplankton). After Dambacher et al. (2002).

Appendix I: Summary of the SEAMOUNT (v2) database fields, brief description, units, and number of records. Note these fields are regularly being added to, and this represents the available data as of May 2007.

Data field	Description	Units	No. records
reg_no	Unique identifier	Number	756
area_code	NIWA ocean area	Name	755
EEZ	Inside/outside EEZ	Yes/No	756
FMA	Fisheries Management Area	FMA Number	450
fished	Occurrence of bottom fishing	Yes/No	727
latitude	Latitude	Decimal degree	755
longitude	Longitude	Decimal degree	755
depth_top	Depth at peak	m from sea surface	746
depth_base	Depth at base	m from sea surface	720
elevation	Elevation of seamount	m	721
name	Seamount name	Name	307
source	Source of locality data	Vessel/trip/person, regional bathymetry	744
min_cont	Minimum depth contour	m	610
max_cont	Maximum depth contour	m	610
area_km2	Approximate area at base	sq.km	633
age	Geological age of seamount	MYr	125
assoc	Geological association	Margin, oceanic etc	645
origin	Geological origin	Formation type	653
volcanic_activity	Level of volcanic activity	Active/inactive	325
hydrothermal_activity	Active hydrothermal activity	Yes/No	30
mining	Substrates of potential mining interest	Yes	81
substrate	Substrate information available	Yes	122
dist_shelf	Distance from continental shelf	km	710
surf_water	Surface water mass	Subtropical etc	754
chl_a	Mean chlorophyll a		710
chlor_mu	Surface chlorophyll concentrations	mg/m^3	755
chlor_sd	Standard deviation of mean chlorophyll a		755
chlor_cv	Coefficient of variation of mean chlorophyll a		755
c2_npp_mu	Carbon-based model of mean primary production		755
c2_npp_sd	Standard deviation of mean modelled primary production		755
c2_npp_cv	Coefficient of variation of mean modelled primary production		755
vgpm_npp_mu	Vertically generalised production model of mean net primary		754

	production		
vgpm_npp_sd	Standard deviation of VGPM mean net primary production		754
vgpm_npp_cv	Coefiicient of variation of VGPM mean net primary production		754
biol_samp	Biology sampled	Yes	232
matrix_coral	Structure forming corals present	Yes	56
sponge	Sponges present	Yes	42
protect_status	Protection status	Closed	24
total_num_tows	Total number of bottom tows	Number	708
total_catch	Total catch of fish in bottom tows	tonnes	708
total_years_fished	Total number of years fished	Number	708
FII_all	Fishing Importance Index	Relative value	708
Appendix I continued			
year_first_fished	Year first fished	Year	181
dist_towed	Sum of tow lengths on seamount	km	708
n_directions	Number of tow directions	1-4	708
direction_N	Number of tows in North direction (315-045°)	Number	708
direction_E	Number of tows in East direction (045-135°)	Number	708
direction_S	Number of tows in South direction (135-225°)	Number	708
direction_W	Number of tows in West direction (225-315°)	Number	708
FEI	Fishing Effects Index	Relative value	708
annual_amp_SST	Annual amplitude of sea surface temperature	Degrees C	693
winter_SST	Wintertime sea surface temperature	Degrees C	693
summer_SST_anomaly	Summertime sea surface temperature anomoly	Degrees C	693
spatial_SST_gradient	Spatial sea surface temperature gradient	Decimal	693
curr_speed	Mean current speed	m/second	753
depth_thermocline	Depth of thermocline	m	745
MEC_20	Marine Environmental Classification	20 class	677
MEC_33	Marine Environmental Classification	33 class	672
ann_mean_semi_diur_tide	Mean semi-diurnal tidal flow	m	756
diurnal_tide	Mean diurnal tidal flow	m	756
prob_cap_diurnal	Probability of Taylor Cap formation in mean tidal flow	Decimal	525
prob_cap_meanflow	Probability of Taylor Cap formation in mean current flow	Decimal	525
min_slope	Minimum slope	degrees	530
max_slope	Maximum slope	degrees	530
mean_slope	Mean slope	degrees	528
sd_slope	Standard deviation of mean slope	degrees	527

calc_area	Calculated area from basal polygon	Sq.km	541
perimeter	Perimeter distance of base	km	541
No_fish_research	Number of fish taxa from research surveys	Number	45
No_squid_oct_research	Number of squid taxa from research trawl surveys	Number	42
No_trawls_research	Number of research trawls on seamount	Number	45
comments	Comments	Anything	29