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 NIWA, unpublished results), but it is not possible to use this proposed measure at this moment.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.