A strategy to assess trends in the ecological integrity of New Zealand's marine ecosystems

Prepared for Department of Conservation

December 2011

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NIWA Client Report No:	HAM2011-140
Report date:	December 2011
NIWA Project:	DOC12203

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Executive summary

The challenge for basic and applied ecologists in the next decade will be to ensure that ecological principles are used to improve the nation's programme to protect and manage water resources (Karr 1991).

Two decades after Karr's paper introducing the concept of biological integrity to aquatic environmental research, we are starting to think about the application of this concept in New Zealand's extensive and diverse marine ecosystems.

This report addresses the issue of ecological integrity in marine ecosystems. Ecological integrity is a broad and overarching concept that we are using as a principle to develop a cost-effective monitoring programme relevant to all marine habitats. Given this scope, the monitoring programme would not be expected to provide the detail needed to identify all changes in marine ecosystems, but would provide a currently unavailable description of the status of our marine ecosystems and highlight where and when further information is needed.

Marine habitats are highly diverse and this project is a broad-brush approach encompassing a wide range of pelagic, soft-sediment and rocky habitats. We focus on current and future Marine Protected Areas, and to a lesser extent protected species, within the 12NM limit. This mainly encompasses intertidal to shelf break habitats, although in some parts of New Zealand the 12NM limit includes slope and deep-water habitats.

Our approach is to review the concept of integrity in relation to both the work already done for the Department of Conservation (DOC) in terrestrial and freshwater ecosystems as well as the application of the concept in a marine context internationally.

We then identify potential monitoring variables and consider the development of tools linking different measurements into a context that can be used to define integrity status.

We consider monitoring data available from New Zealand's marine ecosystems, but it should be noted that to date there is no national monitoring of the status of our marine environment. This is a different situation to our terrestrial and freshwater ecosystems, requiring that we are both innovative, building on a limited knowledge base, but also that we develop this programme as an iterative process.

This report frames up the development of a cost-effective monitoring programme and identifies key gaps or areas where techniques need to be developed and tested.

Background

This report was developed in response to a TOR from the Department of Conservation "Ecological Integrity in the Marine Environment: a review (4 May 2011)".

The objectives of this project were:

1. To review the concept and applicability of "Ecological Integrity" (as defined by Lee et al. 2005) for New Zealand's marine environment, with a focus on monitoring marine sites and species managed by the Department of Conservation[1].

2. To review indicators of marine ecological integrity that have been identified internationally and to identify potential indicators that may be relevant to New Zealand's marine ecosystems.

3. To identify a potential suite of key marine ecological indicators for DOC, review any past or on-going testing or research on these indicators and identify a priority list of those that require further research, testing and ground-truthing.

Project outcome
Improved understanding and reporting of the status of New Zealand's marine environment and the role of conservation management in restoring or maintaining ecological integrity.

Deliverables
A published report and/or scientific paper(s) on the concept of ecological integrity in New Zealand's marine environment, which includes identification of a suite of potential indicators that may require further testing and research.

A seminar to DOC staff on the review, with discussions on future research, including testing and ground-truthing requirements.

[1] Marine Reserves (Marine Reserves Act 1971), Marine Mammal Sanctuaries (Marine Mammals Protection Act 1978), marine mammals (Marine Mammals Protection Act 1978), seabirds (Wildlife Act 1953) and marine species included in schedule 7a of the Wildlife Act 1953.

1 Part 1: Ecological integrity in a marine ecosystems context

1.1 The purpose of 'Integrity monitoring' for DOC

Initiating the process of assessment of ecological integrity for DOC in terrestrial systems, Lee et al. (2005) discussed a number of issues concerning monitoring to assess ecological integrity at the national scale. They emphasise the importance of defining a comprehensive, verifiable picture of New Zealand's biodiversity, in the broadest context, for (i) both national and international assessments, (ii) checks on the efficacy of conservation management and (iii) knowledge advancement. They note that in many countries, such national monitoring is based on a combination of techniques including inventory, definition of status and trends, and surveillance monitoring of biodiversity. In our experience of long-term ecological monitoring in New Zealand's marine ecosystems, this is an important aspect of wise resource management and a knowledge base that grows in value with time. Lee et al. (2005) noted that monitoring requires long-term commitment, adequate resources, and organisational stability. Also important in any monitoring programme is a basis of scientific rigour.

Defining the status and trends of ecological integrity at both national and regional scales should support informed decision making on the allocation of DOC's resources; assess the effectiveness of management and policy; and act as an early warning system (Allen et al. 2009). Allen et al. (2009) identified three reasons why new monitoring is needed to define ecological integrity in terrestrial ecosystems. Namely, existing information is historical; or lacks sufficient detail; or is spatially incomplete. For marine coastal ecosystems, new monitoring is needed for these reasons, but more fundamentally, it is needed because there is no nationally coherent programme of assessment of the status and trends of marine ecosystems. There are a few good examples of how to monitor New Zealand's coastal marine environment (Hewitt et al. 2009, Hewitt & Thrush 2007). However, in comparison to the investment in terrestrial and freshwater ecosystems, coastal ecosystems have been undervalued nationally, resulting in sparse information.

Ecological integrity definitions and the associated indicators are hierarchically nested and nationally focused. Schallenberg et al. (2011) and Lee et al. (2005) provide a sound conceptual basis for defining ecological integrity in terrestrial and freshwater ecosystems and they deconstruct the concept of ecological integrity to define specific quantifiable indicators. However, they fall short of reconstructing the information gained from these measurements into an overall assessment of the status and trends of ecosystems with regard to ecological integrity. Nevertheless, Schallenberg et al. (2011) refer to the development of a multi-metric index of integrity, potentially based on boosted regression tree approaches, although they do not develop this index.

1.2 Ecological integrity – what is it?

Ecological integrity is a holistic term that seeks to capture our sense of nature, its functionality and self-maintenance. Thus, it is dependent on human values and our perceptions of nature, which in our society are wide-ranging. For DOC, ecological integrity has been defined in the context of terrestrial and freshwater ecosystems (excluding estuaries), with both Lee et al. (2005) and Schallenberg et al. (2011) reviewing the ecological literature and policy context. This work will not be repeated here. Instead, our aim is to build on the work for terrestrial and freshwater ecosystems presented to DOC and support an integrative management framework balanced across ecosystems. However, it is important to note that marine ecosystems are different both in terms of some key aspects of their ecology and ecosystem function, as well as our ability to sample and apparent willingness (as a nation) to do so.

Lee et al. (2005), referring to a report from The United States National Academy of Sciences (2000), defined biological integrity as: "*The capacity to support and maintain a balanced, integrated, adaptive biological system having the full range of elements (genes, species, and assemblages) and processes (mutation, demography, biotic interactions, nutrient and energy dynamics, and metapopulation processes) expected in the natural habitat of a region (Karr 1996).*"

The National Parks of Canada definition of ecological integrity was also presented by Lee et al. (2005) as: "*a condition that is determined to be characteristic of its natural region, and likely to persist, including abiotic components, and the composition and abundance of native species and biological communities, rates of change, and supporting processes. (Canada National Parks Act 2000)*". Lee et al. (2005) consider that this is the best descriptive term. However, summarising Lee et al. (2005), Allen et al. (2009) defined ecological integrity as "*the full potential of indigenous biotic and abiotic features, and natural processes, functioning in sustainable communities, habitats and landscapes*". Major components of ecological integrity are defined as: indigenous dominance (to maintain natural character); species occupancy (to avoid extinction); and ecosystem representation (to maintain a full range).

Considering freshwater ecosystems Schallenberg et al. (2011), citing De Leo & Levin (1997), imply that integrity encompasses a sense of completeness in terms of ecosystem structure and function. They went on to define Ecological Integrity as "*The degree to which the physical, chemical and biological components (including composition, structure and process) of an ecosystem and their relationships are present, functioning and maintained close to a reference condition reflecting negligible or minimal anthropogenic impacts".*

Figure 1: The four Pillars of Integrity identified by Lee et al. (2005) and Schallenberg et al. (2011).

These definitions present some serious problems in terms of operationalising ecological integrity in the context of marine ecosystems. Firstly, quantifying all non-indigenous biota is exceedingly difficult, and determining all as a threat is not justified; especially given the age and connectivity of the world's oceans and the long evolutionary history of many marine species. However, it is worth noting that about half of New Zealand's marine species are endemic (Gordon et al. 2010). Globally, this is an unusually high level of endemism (Costello et al. 2010). Defining the functioning of sustainable communities is ambiguous as economic and ecological sustainability are often confused in marine ecosystems. Finally, all ecosystems are dynamic (see discussion below on baselines). The concept of ecological integrity refers to the necessity to safeguard the self-organising capacity of ecosystems (Burkhard et al. 2011). This emphasises the role of complex system dynamics in ecological functioning and the maintenance of resilience. This does imply an important focus on functional performance including resilience, which in turn implies that we need to consider functional extinction as distinct from species extinction. To paraphrase the Justice Potter Stewart (US Supreme Court) who remarked in a 1964 case that "*I shall not today attempt further to define [obscenity]... and perhaps I never could succeed in intelligibly doing so. But I know it when I see it*." Ecological integrity is a high-level concept perhaps better understood by its absence rather than its presence. Despite the difficulty of defining the term rigorously, it is possible to define more specific components and rigorously quantify these (Schallenberg et al. 2011).

1.3 International context

Before developing a strategy for New Zealand's marine ecosystems, it is appropriate to consider major international initiatives in terms of monitoring the status and trends of marine ecosystems. As we have already indicated much of the scientific literature moves very quickly from the general idea of ecological integrity to specific indices. In this section, we will focus on the broader issues.

Considering the broader scientific literature relevant to marine ecosystems, we note that there is concern over the poor definition of ecological integrity (Callicott et al. 1999) and that it is frequently used without clear definition or with no definition at all (e.g., Henriques et al. 2008, Munari & Mistri 2007). Ecological integrity is used with the sentiment of preserving the 'important' aspects or attributes of ecosystems, but is not always either conceptually or operationally defined. Larkin (1996) commented on the diversity of meanings for both ecosystem health and ecosystem integrity, and, in the context of ecosystem-based management, critically noted that these terms are not readily translated into operational language for resource management. The broad definition of 'ecological integrity' means that it spans multiple spatial and temporal scales which complicate its effective implementation, parameterisation and subsequent interpretation (Borja et al. 2008a, Nunneri et al. 2007). Inevitably, there are different indices for different types of species in different habitat spaces (e.g., phytoplankton, zooplankton, benthos algae, fish), however, there are few methodologies integrating all elements into a single evaluation of a watershed (Borja et al. 2008b, Diaz et al. 2004).

In Europe, in response to the water framework directive (WFD), there has been an explosion of index development and revision. Specifically in a marine context, this has led to the recent development by the European Union (EU) of the Marine Strategy Framework Directive (MSFD). These directives include the need to consider ecological integrity within its aims, although this is mainly as an overarching principle that is operationalised as "good environmental status" (http://ec.europa.eu/environment/water/marine/ges.htm; http://ec.europa.eu/environment/indicators/index_en.htm). "*Good environmental status means the environmental status of marine waters where these provide ecologically diverse and dynamic oceans and seas which are clean, healthy and productive within their intrinsic conditions, and the use of the marine environment is at a level that is sustainable, thus safeguarding the potential for uses and activities by current and future generations, i.e., the structure, functions and processes of the constituent marine ecosystems, together with the associated physiographic, geographic, geological and climatic factors, allow those ecosystems to function fully and to maintain their resilience to human-induced environmental change. Marine species and habitats are protected, human-induced decline of biodiversity is prevented and diverse biological components function in balance*".

The MSFD is the first legislative instrument in the EU for marine biodiversity and contains a regulatory objective that "biodiversity is maintained by 2020", as the cornerstone for achieving good environmental status. The MSFD enshrines in a legislative framework the ecosystem approach to the management of human activities that impact on the marine environment by integrating the concepts of environmental protection and sustainable use. This approach is generally consistent with policy aspirations and strategy documents in New Zealand as well as the underpinning research we have previously conducted in collaboration with DOC in the Coasts and Oceans OBI.

Criteria and methodological standards on Good Environmental Status (GES) of marine waters have been defined by the EU (Article 9 (3) of the MSFD; http://ec.europa.eu/environment/water/marine/ges.htm). Specifically, 11 descriptors of good environmental status are laid down in Annex I of the Directive.

"*Namely:*

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

It is clear that guidance will need to be sought to operationalise the monitoring the status and trends in these descriptors. A recent publication (Rice et al. in press), summarizes the conclusions of an international group of experts assembled to operationalise the MSFD with regard to seafloor habitats. This group concluded that eight attributes of the seafloor should be monitored:

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

Within the EU, implementation is left to the individual countries, but this technical advisory group recommended three steps using the best available information:

- identify the ecological structures and functions of particular importance to a given ecosystem, using emerging methods for integrated ecosystem assessments

- review the human activities likely to occur in the area of concern, and based on the past and present levels of pressures associated with these activities, identify the ones most likely to pose a threat of degrading environmental status, and
- for the ecosystem components and pressures identified as being of greatest importance for a particular area, identify candidate indicators using established criteria.

This was considered to be most appropriately implemented via the use of a risk-based approach that encompassed:

- the intensity or severity of the impact at the specific site being affected
- the spatial extent of the impact relative to the availability of the habitat type affected
- the sensitivity and vulnerability versus the resilience of the area to the impact
- the ability of the area to recover from harm and the rate of such recovery
- the extent to which ecosystem functions may be altered by the impact, and
- where relevant, the timing and duration of the impact relative to the times when the area serves particular functions in the ecosystem.

The EU recognises that a major challenge in the implementation of its marine strategy is the necessity of scientific knowledge. This problem is also substantive for New Zealand if we are to do more than token work. Collaboration between research and policy is essential, and science must provide the knowledge upon which integrated management can build the tools for assessing progress towards good environmental status.

With a broader trans-Atlantic perspective of assessing the quality of seafloor habitats (Diaz et al. 2004) recommend adopting a framework for ecological assessment that:

- Delineates, or classifies, regions of habitat that can be quantitatively defined (in time or in space) according to their physical, chemical and biological character.
- Identifies clear relationships between anthropogenic disturbance and key ecological attributes of the target habitat and/or species.
- Assesses and monitors the status of ecosystem performance relative to recent historical system states and suitable reference sites.
- Incorporates predictive models and other theoretical approaches.
- Is of high relevance, communicable, robust from an analytical and statistical standpoint, and be able to meet environmental legislative criteria.

Again, these are high-level recommendations that require detailed thinking to enable operationalisation. Nevertheless, while these frameworks are more extensive than that required for Marine Protected Area (MPA) monitoring, they do provide some useful information for designing a network for monitoring ecology integrity in MPAs.

1.4 Specific indicators and their attributes (in an ideal world)

General requirements for an ecological indicator are:

- Easily measured.
- Sensitive to stressor on the system.
- Respond to stress in a predictable manner.
- Signify impending negative change so that they can be averted by management intervention.
- Integrate across key gradients in the system.
- Under reasonable sampling scenarios do not behave erratically.
- Relate to management goals.
- Be scientifically defensible.

These criteria (see, Dale & Beyeler 2001, Norris & Hawkins 2000) are rarely met completely by any index. Most of the marine indices focus on benthic macroinvertebrate communities because there is a long history documenting the relevance to the rest of the ecosystem and comparative ease of sampling and interpretation of data (Borja et al. 2008b, Gray 1981).

There is a bewildering plethora of indices for application in coastal marine ecosystems. Martínez-Crego et al. (2010) analysed the strengths and weaknesses of 90 published biotic indices purported to assess the status of coastal waters and identified a number of consistent issues with applicability (due to both practical and conceptual difficulties). This included failure of indices based on community structure attributes to show decline linked to stressor and poor relevance to ecological integrity specifically for indices based on attributes at the sub-individual level (e.g., multi-biomarkers). Earlier Diaz et al. (2004) evaluated 64 separate indices of seafloor habitat quality and also noted serious problems. In response to this growth in index production one of the strongest proponents for indices in the EU (Angel Borja) notes "*There is a need to identify the best ecological integrity measure from the masses which have been generated in the last decade. There is a need for minimum criteria for index validation and work on achieving uniform assessment across varying sales. There is a need to integrate indices over different ecosystem elements*" (Borja et al. 2009a). Diaz et al. (2004) concluded that *"The fact that so many indices of aquatic habitats quality have emerged over 20 years indicates there is little acceptability of any specific metric by environmental managers or scientists."*

Most indices fail on their narrow applicability and their reliance on the definition of reference conditions. To some extent, this can be overcome by combining different but complementary characteristics. A recent special session on the on use of indices in ecological integrity assessments, held by the Coastal and Estuarine Research Federation (Borja et al. 2009b) identified four major challenges:

- reduce the array of indices by identifying the index approaches that are most widely successful
- establish minimum criteria for index validation

- intercalibrate methods to achieve uniform assessment scales across geographies and habitats, and
- integrate indices across ecosystem elements.

Most of these indices were developed in Europe and North America for application in estuarine or coastal soft-sediment habitats where major stressors in the region are high levels of contamination and eutrophication. Many of these indices involve allocating macrobenthic species to groups reflecting different responses to stressors thus requiring local knowledge and expert judgement. For example, the European index called AMBI (Borja et al. 2000) recognises five distinct ecological groups of benthic species whose abundances are supposed to vary predictably with levels of organic enrichment (from first-order opportunists that proliferate in over-enriched anoxic sediments to specialist carnivores that only occur in unpolluted conditions). Similarly, the calculation of the B-IBI (developed in the USA; Weisberg et al. 1997) is based on percentages of "pollution sensitive taxa" and "pollution indicative taxa" as well as overall macrofaunal abundance, richness and diversity. Unfortunately, an assessment of the applicability of the overseas indices AMBI and B-IBI on sandflat macrofaunal communities in the Auckland Region showed their marked lack of sensitivity to New Zealand stress gradients (van Houte-Howes & Lohrer 2010). This was probably because organic enrichment is not currently the dominant stressor in these habitats. While eutrophication and sediment contamination may well be more general problems in the future for New Zealand, sedimentation and fishing are the main and current threats to ecological integrity. More specifically, New Zealand MPA benefits focus on reducing the multifaceted ecosystem and biodiversity impacts of fishing (Thrush & Dayton 2002, Thrush & Dayton 2010).

Recently, a number of indicators have been developed to assess the impact of fishing. Many focus on traditional fisheries management, while some have broader ecosystem implications relevant to assessing ecological integrity. In particular, the application of biological traits analysis has both facilitated comparisons across communities and made important links between changes in community structure and function (Bremner et al. 2006a, Bremner et al. 2006b). For example, filter-feeding, attached epifaunal organisms and large organisms, in general, tend to show negative correlations with trawling intensity, whereas small infauna and scavengers tend to become more abundant (Tillin et al. 2006). Over the long term, a common pattern emerges of the loss of epifauna and large and long-lived organisms such as burrowing urchins, large bivalves, sea pens, and reef-building sabellid polychaetes (Robinson & Frid 2008). de Juan et al. (2009), see also de Juan et al. (2007), developed a multivariate approach to assessing trends in benthic communities based on the abundance and density of large epifaunal organisms. This approach utilised a combination of biological traits that relate to size, age, rarity and vulnerability to trawling disturbance, with trials indicating the approach was effective in both sandy and muddy sediments.

Modelling approaches have also been advocated as a way of developing indicators, as opposed to interpreting their broader significance. One approach is to develop indicators based on data already being gathered for traditional fisheries management. Link et al. (2002) investigated a range of abiotic, biotic, and human metrics for the northeast U.S. continental shelf ecosystem, a comparatively data-rich ecosystem. Their analysis offers a note of caution in the definition and interpretation of a minimal suite of indicators. They emphasised the need for a diverse array of indicators to characterise ecosystem status, highlighting that such indicators cannot easily be treated as analogues of the indicators used in single-species fisheries management. They also called for the development of mechanistic or analytical models of key ecosystem processes.

One of the problems with many of the indices currently available is that they focus on structure and deal less explicitly with function. To some extent this failure can be met by studies that employ functional trait approaches and thus focus more on what organisms do rather that what they are. However, at present there is a gap in terms of what we can easily and routinely measure with regards to the function, and what these values mean (Birchenough et al. 2012). Directly measuring function is possible but in many cases is more likely to be supported as a research question rather than a routine long-term monitoring tool. In specific circumstances metrics are available that provide insight into function that cannot be derived from the community data. Landscape scale information on the connectivity of habitats, patch structure or the degree of bare space, provide broader scale parameterisation and provide insight into function (Bartel 2000, Bostrom et al. 2006, Garrabou et al. 1998). Linking information on species richness and the degree of variability in community composition (β-diversity) can also be used to infer functional relationships, ecological connectivity and resilience (Hewitt et al. 2005, Thrush et al. 2006, Thrush et al. 2008, Thrush et al. 2010).

Other forms of indirect inference on function are derived from trophic and population structure based indices and degree of bioengineering. Production/biomass or population size structure information can be used to make insightful inferences about the role of species in energy flow through food webs and community dynamics. Bioengineering species fundamentally influence the architectural and functional complexity of the seafloor. These include emergent organisms that modify flow and provide settlement sites and refugia for predators or prey; predators digging into the substratum in search of food (e.g., rays, birds, fish, crabs, etc.) and organisms creating tubes, burrows, mounds, and other manipulations of the sediment. The activities of these benthic marine organisms significantly influence the nature and rate of biogeochemical processes that sustain the biosphere (Lohrer et al. 2004, Thrush & Dayton 2002, Thrush & Dayton 2010). Microbial species in the sediments drive nutrient and carbon cycling, but this is strongly facilitated by the movement, burrowing, and feeding of infauna and epifauna. Bioturbation indices, such as the measurement of the apparent redox discontinuity layer (aRPD), can also be used, particularly in muddier sediments, to provide insight into functioning related to nutrient recycling and oxygen flux across the sediment water interface (Birchenough et al. 2012, Solan et al. 2004, Teal et al. 2010). The strong link between function and ecological integrity suggests that we need to focus on incorporating function and ecosystem performance information into the development of monitoring tools for ecological integrity.

1.5 The New Zealand marine environment context:

New Zealand’s Marine Protected Areas (MPA) Policy is intended to guide the development of a comprehensive and representative network of MPAs. There are over 30 no-take marine protected areas established in New Zealand waters, with MPA applications recently approved in both the Sub-Antarctic and West Coast marine bioregions. With these new reserves, protection is given to over 10% of New Zealand’s territorial sea. However, most of this area (99%) is surrounding isolated offshore island groups (Auckland, Kermadec, Sub-Antarctic Island groups). Many of New Zealand’s MPA applications were supported by community groups, and often represent perceived high value habitats for recreational purposes, with an over-emphasis on rocky reef habitats relative to soft sediment habitats.

Monitoring of New Zealand MPAs to date has not been designed to assess ecological integrity, previous monitoring has served a variety of purposes and perhaps the most important was to demonstrate the value of MPAs in terms of the recovery of exploited populations and consequent community change. Consequently, from an integrity perspective monitoring has been inconsistent, with high variation in monitoring frequency and collection methods (Table 1). As the perceived goal to society of an MPA is often seen as recovery from overfishing, the majority of monitoring surveys have been focussed on key species that are predicted to increase in abundance following protection such as lobster, blue cod, and other recreationally important reef fish (Table 1; McCrone 2001). Monitoring surveys from 1999-2008 represent this bias toward a key species approach, with two-thirds of monitoring surveys looking at either rock lobster or subtidal reef fish, and many community-based surveys of intertidal and subtidal reefs still focussed on other key species such as paua and kina. Only eight of over 300 monitoring surveys in 1999-2008 monitored soft sediment communities, of these, only one survey was specifically designed to monitor threats to the MPA (in that case, human disturbance impacts). Other monitoring surveys were not designed to address threats or future changes to marine communities beyond changes due to release from fishing pressure.

While most MPAs have baseline habitat surveys, there is little consistency in habitat detail and georeferencing, ranging from descriptive language in older MPA applications, to detailed habitat maps common to most recently designated MPAs. Three MPAs lack formal baseline habitat information, and only three of the MPAs existing in 2008 had been subject to follow up habitat monitoring surveys, all in the Nelson-Marlborough region. More habitat detail is generally given to rocky reef areas in MPAs, with numerous types of kelp forest and rock platform often included in a list of habitats, while soft sediment habitats are rarely described to a similar level of description.

Standard operating procedures have been drafted by the Department of Conservation to avoid historical lack of consistency in collection methods. Standard sampling procedures have been provided for underwater visual counts for reef fish, reef macroinvertebrates, and reef macroalgae; baited underwater video; transect surveys of intertidal soft sediment and rocky reef communities; and potting for rock lobster and blue cod (McCrone, DOC unpublished internal report).

Table 1: Monitoring statistics for New Zealand MPA surveys by the Department of Conservation for the period 1999 to 2008, delimited by habitat and survey type.

	% of total monitoring surveys (1999-2008)	Total number of surveys (1999-2008)	Primary methods[1]
Hard substrate - estuarine and marine (intertidal to 30m)			
Habitat	0.01	2	UVC, BUV, DUV, side scan sonar & drop camera
Subtidal reef fish	0.32	102	UVC, BUV
Subtidal reef benthic communities and key species (e.g., paua, kina)	0.21	68	UVC, Photoquadrats
Subtidal reef rock lobster	0.32	101	UVC, Pot
Subtidal reef blue cod	0.08	24	UVC, Pot
Intertidal reef communities	0.03	11	Transect, Photoquadrats
Soft substrate - estuarine and marine (intertidal to 30m)			
Subtidal estuarine fish	0.00	1	BUV/DUV
Intertidal soft sediment benthic communities	0.01	2	Transect, Photoquadrats
Subtidal soft sediment benthic communities	0.02	5	UVC, Photoquadrats
Other			
Monitoring impacts of human activity	0.01	2	Transect , UVC, BUV, Photoquadrats, drop camera, other

1: UVC – Underwater Visual Count, BUV- Baited Underwater Video, DUV – Drift Underwater video.

Considering other monitoring of marine ecosystems in New Zealand, there is considerable ecological information collected on NZ's coastal ecosystems, although this is dwarfed by the extent of our coastal domain. Information is collected by various agencies other than DOC, for example, NIWA, Cawthron, various Universities, MAF BNZ, MAF Fisheries, Maritime NZ, although most is collected and held by regional and district councils. Most information is not collected as part of a regular time series and the lack of a national perspective means that the types of data collected are not standardised, and the space and time scales over which it is collected vary. An effort to address the lack of a national perspective is recently being undertaken by the MAF Fisheries funded project "ZBD2010-42: Development of a National Marine Environment Monitoring Programme (MEMP)". This project seeks to collate high level sampling details of all data collected in the marine environment (biological, chemical and physical from estuaries to the edge of the EEZ) and determine which sites and variables could contribute to regional and national reporting of the State of the Environment. In order for variables to contribute to such a scheme, they must not only mean something, but also be collected in a sufficiently standard fashion as to allow comparison between places.

1.6 The importance, and problem, of baselines

For any indicator to be interpreted it is important that measured values can be judged against some reference value. In ecology, this often requires the identification of baseline conditions or an agreed level of natural disturbance to which the valued ecological state is resilient. Unlike at least some New Zealand riverine ecosystems (Schallenberg et al. 2011), there is no meaningful ecologically defensible classification system that can provide a reference. The environmental classification approach has failed in marine systems due to importance of biophysical interactions that drive emergent community patterns at scales relevant to management and conservation. But, how do we develop baselines at this late date? Intense disturbance selects for species with appropriate responses. As a result, small, mobile species and rapid colonists dominate benthic communities, and we lose track of natural biodiversity. Equally importantly, ecosystems are dynamic and may not respond to disturbance in a simple monotonic fashion, but instead exhibit threshold-type responses (de Young et al. 2008). This means that benchmarks and baselines must be carefully considered. Thus, ecological insight is currently our best option to assess the risk of threshold-type responses and ecological ratchets (Duarte et al. 2009). For marine systems baselines can be derived from time-series and in the case when quality environmental data is available it is reasonably easy to detect trends and deviations from previously monitored states (Hewitt et al. 2001, Hewitt & Thrush 2009, Hewitt & Thrush 2010). However, this technique requires a commitment to gathering meaningful ecological time series. Another approach applied in the management of exploited marine fisheries is the definition of some sustainable population level, although this definition is reliant on fisheries theory.

Information on status and trends of endangered species are regularly reported due to New Zealand's commitments to international treaties, though in the marine environment, these species are mostly charismatic megafauna such as marine mammals and seabirds. Recent archaeological and ecological research has provided further estimates of historical abundance of these higher trophic groups prior to Maori and European colonisation (Jackson et al. in press, Pinkerton in press). While these estimates have high uncertainty, they nevertheless suggest New Zealand coastal ecosystems supported orders of magnitude higher abundance of large megafauna than are currently present. However, it is unlikely that MPAs at currently implemented sizes will result in large increases in population size of the wide-ranging marine species (Dayton et al. 2000). Instead, management of threats and monitoring of population trends and responses to threats at regional or national scales are more relevant than MPA scale monitoring for these species.

Other ways of estimating historical baselines include the development of individual indicator models that link changes in indicator response to specific stressors. However, this requires some careful thinking to extrapolate into a multiple stressor context. These approaches could be developed in an ecological framework, but this has yet to be done. Nevertheless, defining some threshold of indicator response to elicit specific management action is important for any effective monitoring programme. After all, one of the main reasons we need to monitor environmental change is to overcome the potential for sliding baselines where personal or societal environmental values are gradually degraded and the full extent of environmental change is not recognised. Until a suitable baseline can be built from appropriate ecological integrity monitoring, a number of other approaches are used to infer change:

- differences in the variability of specific response variables, within and between MPAs
- variability outside of a pre-specified range (e.g., 1 standard deviation)
- expert opinion and local knowledge (see below).

Importantly, in the context of MPA monitoring, we should expect to see change as the system recovers its integrity (depending on the time since the MPA was created and its history of exploitation and the state of other stressors to the system).

1.7 The importance of resilience - the potential for rapid change in baselines

Loss of ecological resilience results in regime shifts that exhibit drastic broad-scale changes in species composition and function (de Young et al. 2008). Regime shifts are described by thresholds, step-trends, criticality, rapid transitions or tipping points, reflecting major changes in the functionality of ecological systems implying that detection of change is not just a matter of statistical significance. Ecological resilience can be considered as the ability of a system to maintain its identity in the face of both internal and external drivers (Cumming et al. 2005). Note that this definition overlaps strongly with that of ecological integrity. Ecological resilience represents an insurance against potentially adverse changes in the delivery of ecosystem goods and services. Thus, resilience is not only a property of an ecological system but is an important ecological service, offering insurance against loss of valued functions. Unfortunately, perspectives on values, states and trends are easily biased by shifting baselines that plague ecological comparisons when information on ecosystem history is limited (Dayton,1989, Dayton, et al. 1998, Duarte et al. 2009, Leppakoski 1975).

Resiliency is the key to conserving ecological integrity via the ability of the system to cope with inevitable changes. If ecological resilience is threatened through disturbance of the landscape and loss of biodiversity, MPA's should offer some potential to restoring resilience, although defining the extent to which MPAs may contribute will be context specific, emphasising the need for appropriate monitoring. Concurring with Schallenberg et al. (2011), we think the focus of tying resilience to ecological integrity should be on capturing the dynamics of ecosystems rather than trying to provide some static benchmark. This is not to suggest that some specific measures, which contribute to the overall measure of ecological integrity, cannot be judged against specific benchmarks or limits.

With regard to resilience, it is always important to define the "of what" and "to what" questions (Carpenter et al. 2001). Research to date has focussed on these questions, however specific techniques to indicate the potential for change in resilience in natural ecosystems are only just beginning to emerge and few have received any empirical testing (Thrush et al. 2009). We have had some success in demonstrating that these largely conceptual developments can lead to forewarning signals in real world monitoring data (Hewitt & Thrush 2010), again using long-term monitoring data. But a more experimental approach may prove useful and informative, if not as an actual monitoring tool, then at least to inform the interpretation of more survey-based approaches.

Based on the above discussions we need to consider six features of ecological integrity, each of which contribute to aspects of nativeness, pristineness, diversity and resilience (Figure 2).

Figure 2: General features of marine ecosystems to include in marine ecological integrity monitoring.

2 Part 2: Developing an Ecological Integrity Monitoring Strategy for marine ecosystems

The implementation of an ecological integrity monitoring programme for the marine environment has a number of significant advantages for marine conservation and broader ecosystem-based management. First, we would develop a time series of broad-scale data from MPAs around New Zealand and this would substantively expand our knowledge base. Second, we would have information to improve the management of MPAs. Third, through the development of novel societal education initiatives associated with the integrity monitoring, we would increase societal valuation of marine conservation and marine ecosystems. Fourth, this monitoring would provide data, knowledge and insight to allow for national assessments of the status and trends in marine ecosystems. Rigorous wisely designed monitoring programmes gain value with the length of the time series and the number of sites incorporated. Often the range and extent of the benefits are difficult to initially grasp, but they are illustrated in a New Zealand coastal context by the Auckland Council's monitoring program. This programme, established in 1986, was designed to assess the ecological health of Auckland's harbours and coastal ecosystems, as well as providing direct evidence of the status of these ecosystems (an important role for the regional councils as stewards of these resources). The programme has gone on to inform and influence a range of management actions including the development of risk assessment procedures, improving understanding of cumulative change, distinguishing natural variability from human induced change, up skilling of managers and society, testing of the efficacy of management actions, and informing decision making to result in improved environmental decision making (Figure 3).

This monitoring strategy must be applied broadly across New Zealand for regional to national assessments for coastal marine ecosystems. Within the context of this report we are focused on monitoring the status of ecological integrity in MPAs. This implies that our focus is not on capturing the effects of the wide range of stressors that impact marine ecosystems, particularly in coastal ecosystems. Rather we focus on the restoration of estuarine and coastal ecosystems associated with the removal of fishing pressure and all the concomitant ecosystems effects that flow from that.

We are conscious of the need to balance rigorous scientifically defensible monitoring designs and data interpretation with potential cost (Figure 3). What we propose is a framework that allows for, and encourages, more detailed measurements and supports knowledge generation considered appropriate to management, policy and societal needs. We also stress the importance of building on previous monitoring and maintenance of these time series in developing an appropriate framework for an ecological integrity monitoring strategy.

2.1 Overall structure of an Ecological Integrity Monitoring Strategy for marine ecosystems

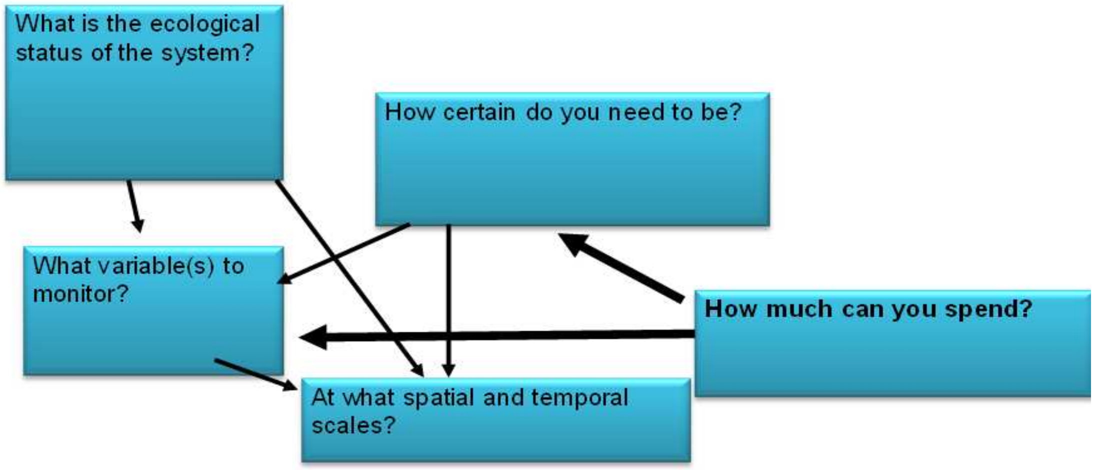

Figure 3: Deciding on how to balance variability, certainty and cost in ecological monitoring programmes.

Variables that underpin ecological integrity must be monitored within the context of a hierarchical framework. This allows us to both incorporate processes and ecosystem features that operate at different space and time scales (variation in habitats, environmental drivers and community components), and define shifts in scale from individual MPAs to regional and national assessments (Figure 4).

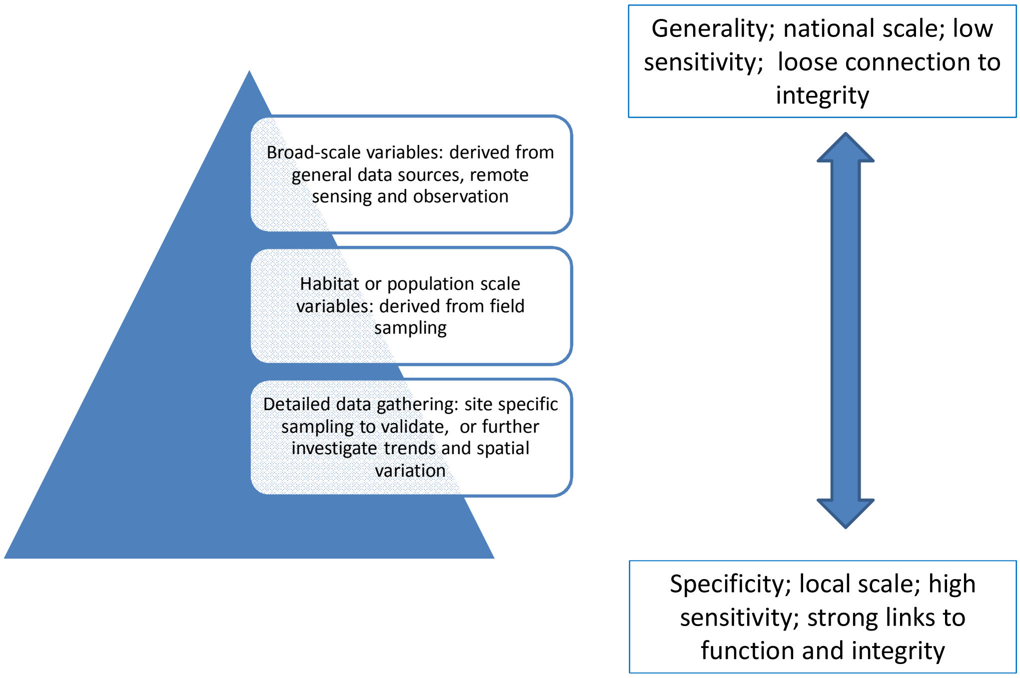

Figure 4: Integrated hierarchy of ecological integrity measures for marine ecosystems.

The first phase of the strategy should be a desktop exercise, using broad-scale information, considering the environmental setting and risks to the specific MPA. This exercise will perform three functions: firstly, to identify categories of environmentally similar MPAs (possibly within bioregions) for future comparisons of their ecological status (see Figure 5); secondly, to identify threats to the integrity and potential recovery of the MPA; and thirdly, to assist in the selection of MPAs for future monitoring. The types of information included in risks are: visitor impacts, specific land-derived contaminants (e.g., sediments, heavy metals), proximity to aquaculture and other in-water engineered structures, and the risk of exposure to fishing pressure. This list of human pressures could also include offences where compliance with policy has been breached. Broad-scale environmental drivers include exposure, depth, slope and location in adjacent seascape. At this stage it is also important to include information on the variety of biogenic habitats (e.g., kelp forests, sponge gardens, bivalve beds), along with the broad-scale environmental setting). All of these features are important both in identifying the most important variables to sample at lower scales and in providing a background against which to interpret change in ecological integrity at a regional level and contribute to national status assessments. Note that this aligns with the proposed to implement the MSFD in the EU.

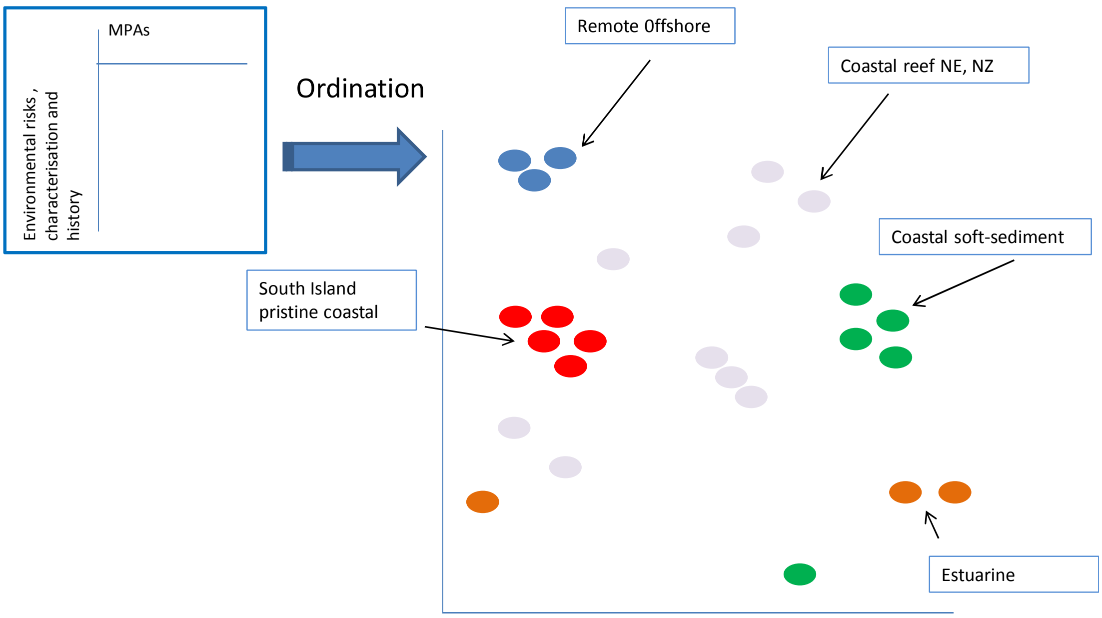

Figure 5: **Preliminary risk and environmental characterisation procedure.**

In order to achieve this assessment, a nationally coherent series of habitat definitions will need to be defined to ensure consistency across sites. Once standardised, numbers of habitats within an MPA and measures within the habitats in a specific MPA, can be compared across MPAs of similar environmental characteristics. These habitats should reflect the diversity and structure of seafloor habitats, but must be simple enough to allow for practical implementation. An illustration of what such a list might look like is given in Table 2. We define habitats as areas that show a broad level of consistency in their physical and chemical structure generated by the interaction of these processes with the resident biology, which is often dominated by particular key species. The implication is we do not define habitats based on features such as sediment or reef, mud or sand alone. Rather we define habitats such as kelp forests, sponge gardens, urchin barrens, *Atrina* beds, or shrimp burrow dominated habitats. Different habitats may be characterised by their apparent homogeneity or heterogeneity. Importantly, habitat types should relate to function and include similar levels of detail for both soft and hard substrates. While this may seem complicated, it can be easily defined by observation. Once developed the habitat definitions should be used to assess the suitability of currently available habitat maps as the basis for integrity assessment and monitoring. We envisage that a suitable standard will be the map developed for Te Whanganui-A-Hei (Cathedral Cove) *Marine* Reserve.

A second phase to this broad-scale assessment is the selection of a sub-set of MPAs that can be surveyed for validation and testing of the monitoring strategy. This should include representative MPAs that cover a range of environmental characteristics to ensure ecological integrity metrics are generalisable. After ecological integrity metrics have been validated, which is expected to be after a period of ~5 years of monitoring, it is likely that some of these MPAs will be included as sentinel sites for long-term monitoring of ecological integrity. Ideally, a rotational strategy of MPA monitoring will occur for non-sentinel sites such that baseline data on habitat maps and ecological integrity for each MPA is accumulated over time, and new data can be compared to sentinel sites to evaluate status and trends in ecological integrity at each MPA. This process is illustrated in Figure 6.

Table 2: Possible list of habitats. Note this is only an illustrative list and needs to be both expanded and have density and patch structure thresholds developed. The potential for habitat classification overlaps also needs to be determined, by utilising present habitat maps. Physical habitat features, such as depth, slope and wave exposure, will also need to be noted during monitoring.

Muddy Sediment Habitats	Sand – Gravel Sediment Habitats	Rocky Habitats
Featureless mud	Featureless sand	Bare rock
Mangrove	Featureless gravel	Algal green turfs
Presence of crab burrows	Heavily bioturbated sands, with numerous spatangoid urchin tracks visible	Algal red turfs
Seagrass bed	Shell fragment dominated sediments	Coralline turf
Uniform muds with no evidence of shell hash	Cockle bed	Coralline crust
Atrina bed	Dog Cockle bed	*Ecklonia* canopy
Oyster reef	*Atrina* bed	*Carpophyllum* canopy
Shrimp burrow fields	Sponge gardens	*Macrocystis* canopy
Metanephrops beds	Rhodolith bed	Encrusting anemones
Seapen fields	Seapen fields	Encrusting sponges
Seawhip fields	Seawhip fields	Bryozoan thicket
Brittle star beds	Brittle star beds	Brittle star beds
Crynoid patches	Bryozoan reef	Black coral thickets
Predator feeding pit patches	Crynoid patches	Crynoid patches
Faunal structures extending >1 or 2 meters above the bed (giant glass sponges; black corals)	Predator feeding pit patches	Urchin barrens
Deep sea vents	Faunal structures extending >1 or 2 meters above the bed (giant glass sponges; black corals)	Faunal structures extending >1 or 2 meters above the bed (giant glass sponges; black corals)
Canyon floors	Seaweed wrack	*Durvillea* belts
Seaweed wrack	Mussel beds	Mussel beds
Tube worms	Tube worms	Paua beds
	Seagrass bed	Seagrass bed

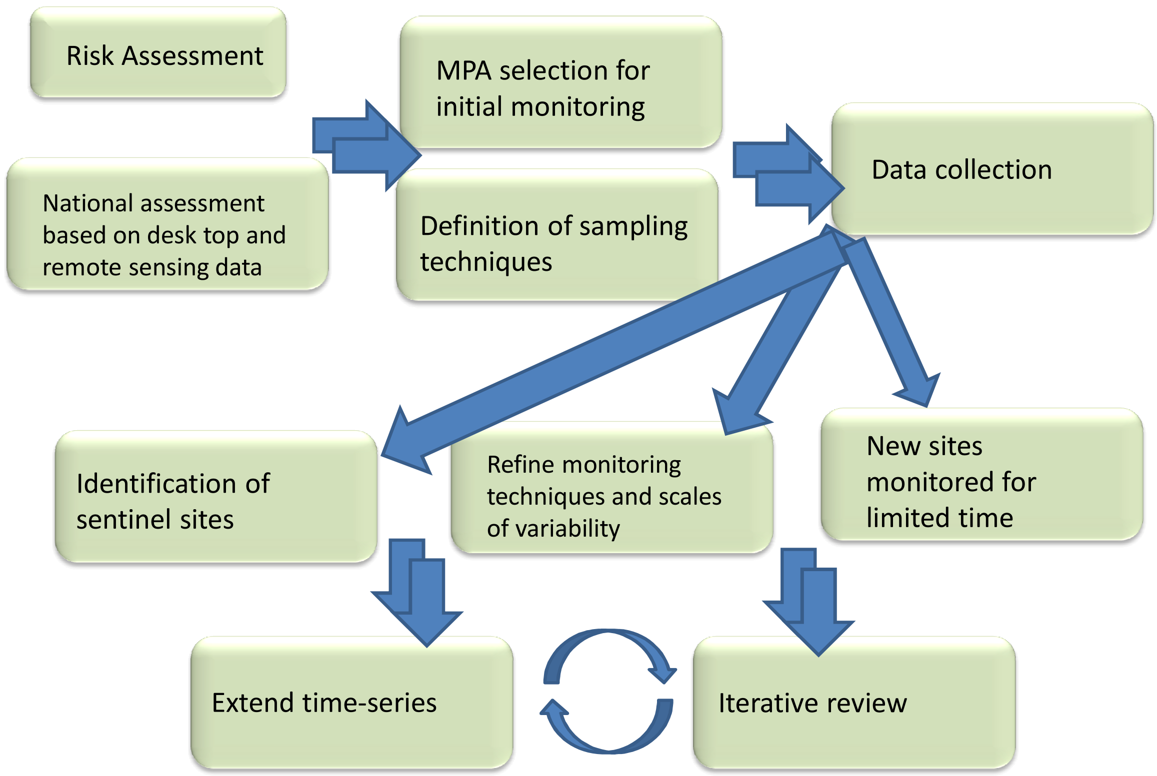

Figure 6: Phases of the monitoring programme (initial 5 years).

Within this second phase, a cost-effective sampling design will need to be considered. This will involve assessing the precision required, the spatial extent of sampling within the MPA and the frequency of sampling required to provide an adequate time series (Figure 3). Statistical criteria are available for assessing many aspects of sampling, although balancing the number of sites with the level of replication often requires expert judgment. As with much of the monitoring that has previously been conducted in MPAs, a contrast between variables inside and outside of the MPA can be very informative in defining the contrasts in ecological status and function generated by protection. However, it is important not to spend too much effort sampling multiple control sites due to the expense of sampling multiple habitats or response variables. Multiple controls often confound the identification of impacts as they inevitably add spatial variance to outside of MPA samples that cannot be mirrored within the reserve (see Hewitt et al. 2001) for further discussion and some real world examples). Control sites should be chosen carefully to match both the environmental conditions encountered in the MPA and the level of habitat heterogeneity. In some cases a reasonable match may not be possible or the protection offered by the MPA may not encompass major stressors to ecological integrity (e.g., sediment loading or urban contaminants). This does not mean that monitoring to assess ecological integrity is not worthwhile as the main strategy is to build up the time-series of data collected within the individual MPAs. Apart from monitoring status and trends in ecological integrity, monitoring can also serve other important societal and educational functions.

A third phase of the strategy is to consider how the data will be used and linked to management actions. For example, a management action could be associated with more intensive sampling, further research, more intensive policing of the MPA, or a report on status in a national context. These actions could be triggered by any selected level of change in the monitored variables and in many cases, it may not necessarily be appropriate to define significance based solely on statistical criteria.

Monitoring programmes are designed with a purpose and thus must link to actions as part of an adaptive management processes (Figure 6); however, this is a critical but often over looked aspect. This element of the monitoring design is beneficial in identifying the need for data security but also the translation of the data into reports of different levels of technical specificity. This provides a vehicle for reporting and reviewing the status of the programme. The programme may require adaptation for a number of reasons, but it should remain scientifically rigorous. Importantly, results from the monitoring should lead to a series of clear management actions, ranging from: further review of data; new sampling; investigation of potential measures to mitigate adverse impacts on ecological integrity; to changes in the size, shape and number of MPAs. Such a process is described in Figure 6, however the details of how specific actions may be triggered by shifts in ecological integrity measures should be decided upon as the programme develops over the next 5 years.

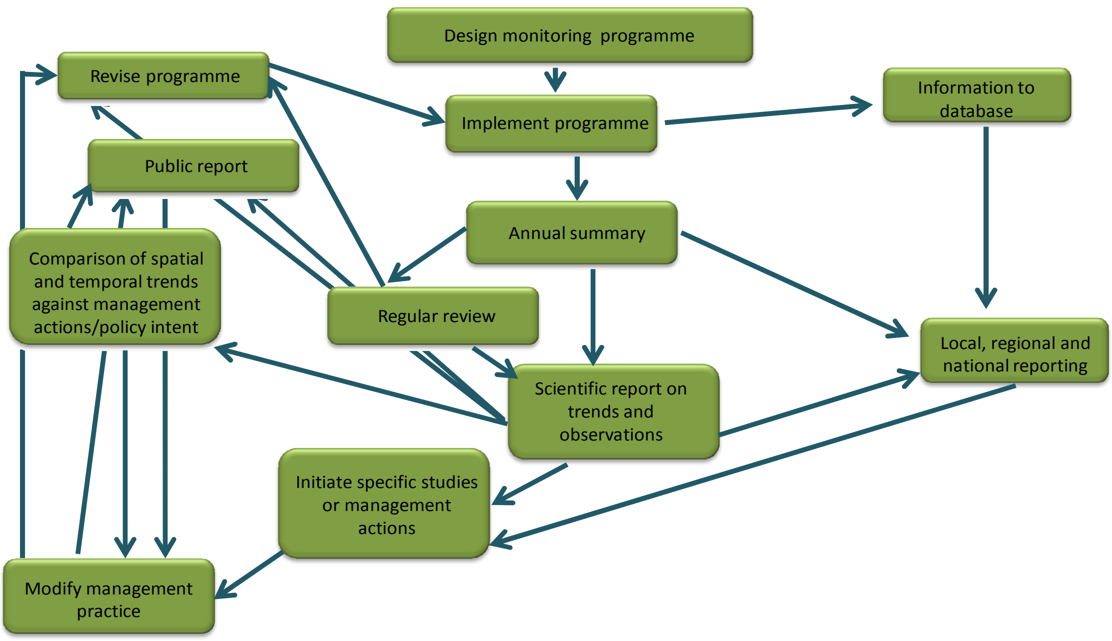

Figure 7: Ecological integrity monitoring for decision-making.

Although focused on development and management in the context of marine conservation, this approach could have much wider application to New Zealand's marine environment. It could be used to inform process such as Integrated Ecosystem Assessments (Levin et al. 2009), which would fit comfortably within a broader ecosystems-based approach to managing our marine ecosystems. This includes: definition of objectives; threats to ecosystems and ecosystem management drivers; development of indicators for ecosystem state; establishment of thresholds for each indicator; risk analyses to evaluate how indicators respond to human and environmental disturbances and the probability that indicators will reach an undesirable state; evaluation of management strategies to predict the effects on the indicators; and monitoring management strategy outcomes (Levin et al. 2009). Other policy and management tools, e.g., marine spatial planning and other strategies within an ecosystem-based management context would also be improved by ensuring fuller assessments of the status of marine ecosystems and better integration of MPAs into the wider range of uses and abuses of the marine environment.

The final phase of the strategy is to consider how the indicators measured will be utilised to present a measure of ecological integrity and be analysed over time to determine trends. This phase will be discussed in more detail in the section "development of a measure of ecological integrity".

2.2 Variables to measure

Integration across a range of parameters avoids too great a focus on the integrity of a single ecosystem aspect. Managers typically need assessments that function across habitat boundaries and over larger geographies on a single assessment scale. Combining indices from different geographies requires that indices be calibrated to the same scale. Some index approaches are more amenable than others to crossing geographical scales.

Schallenberg et al. (2011) identify seven criteria for monitoring ecological integrity in New Zealand freshwater ecosystems:

- Ease of sampling and analysis.
- Potential geographic coverage.
- Relation to ecological integrity components (e.g., nativeness, pristineness).
- Sensitivity to pressure gradients.
- Calibration to reference condition.
- Temporal variability.
- Use nationally and internationally.

We have attempted to work with this framework, although we note significant problems in the definition of reference condition (particularly because, in a broader marine ecosystem-based management perspective, it is the MPAs that will make the most substantive contribution to defining the reference condition).

We propose a hierarchy of monitoring variables ranging from the general to the specific. To match with potential resources we have designed the strategy so that not all levels need to be initially monitored, some can be switched on when decision thresholds are tripped (see below) and the range of monitored variables can be added to as cost-effective technology becomes available.

In order to be able to concurrently identify ecologically meaningful features, sample across the full depth range, and collect data over a sufficient area, we focus on video imagery as the primary tool for collecting data on seafloor and associated organisms. We propose measuring specific habitat features and species abundances, but to more strongly link the structural observations derived from the video (or more traditional sampling) we also propose a biological traits approach. The formulation and analysis of functional groups has a long history in benthic ecology. Functional groups are a consortia of species that share some common attribute that is likely to influence function in a specific way. For example, surface-deposit feeders and suspension feeders should generate strong differences in sediment biogeochemistry irrespective of species-specific attributes. Biological traits analysis provides a more multi-dimensional analysis of species attributes that affect function (Hewitt et al. 2008). For example, filter-feeding, attached epifaunal organisms and large organisms tend to show negative correlations with trawling intensity, while small infauna and scavengers tend to become more abundant (de Juan et al. 2009, Tillin et al. 2008). Such broad-scale description of functional attributes provides insight into how structure and function relationships can change over a range of disturbance intensity. A prototype trait-based approach to assessing how ecosystem functions are modified by stress has been recently proposed for terrestrial plant communities to overcome the context specificity that often dominates community ecology (Suding & Goldstein 2008, Suding et al. 2008). Functional groups and biological trait analysis can be conducted over large space and time-scales and are amenable to investigating the relationships, especially where weak interactions and rare species can influence functional performance across gradients in species composition or environmental factors (Walker et al. 1999). Many ecosystem functions are not the product of species abundance or presence/absence alone. In marine soft-sediments, large organisms are particularly important in influencing processes that affect the fluxes of energy and matter (Thrush & Dayton 2002). These studies provide a useful framework from which to develop a trait-based functional approach to the analysis of video imagery.

We focus our recommendations on monitoring at the MPA scale and both across and within habitats. The habitats we refer to here are those defined in the preceding section (overall strategy)

Table 3 lists a range of monitoring variables that would contribute to an assessment of changes in the integrity status of New Zealand's MPAs. Variables are listed under their core category (naturalness, pristineness, diversity and resilience), although many of these variables would contribute to more than one category.

We do not envisage that the full range of monitoring variables would be sampled in all MPAs, as some variables would be a low risk factor in many. Also we envisage a progressive implementation of data gathering with a base set of core variables that can be sampled easily, supplemented by more detailed sampling if this is required as part of a decision criteria being tripped in the monitoring data (or through other management decision making processes). We focus on a photographic approach to basic sampling, as this allows permanent and geo-referenced recording of information collected along transects. It provides for rigorous and cost-effective analysis of many seafloor features linked to ecological integrity. Below we indicate the type of sampling envisioned and refer to video transects and drop cameras, although Remotely Operated Vehicles or Autonomous Underwater Vehicles could also be used if appropriate. Many of the sampling designs and indices proposed are well developed; while others would need to be developed specifically for this application. Some testing has already been performed as part of our research into the resilience of coastal ecosystems (Hewitt & Thrush 2010, Thrush et al. 2008).

Table 3: Recommended indicator variables for assessing ecological integrity in marine ecosystems and associated measure at different scales of sampling. Note that indicators need not always be measured at each potential scale. We use the 4 categories (nativeness, pristineness, diversity and resilience (Figure 1) to provide concordance with (Lee et al. 2005) and (Schallenberg et al. 2011). Indicators can also be easily categorised using the 6 elements of integrity in Figure 2, in fact for both categorisations (Figure 1 & 2) individual indicators may contribute to more than one category.

Category	Indicator	MPA or broader scale	Across habitat scale	Within habitat scale
		What to measure	What to measure	What to measure
Nativeness				
	Invasive species outbreaks	Occurrence	NA	NA
	Invasive species recognised as major threats	Occurrence	Number of habitats occupied/number of habitats	Abundance initially in monitored video transects; if under high risk, more detailed habitat specific sampling
	Nuisance species outbreaks	Occurrence	Number of habitats occupied/number of habitats	Abundance initially in monitored video transects; if under high risk, more detailed habitat specific sampling
	Presence or spread of invasive habitat-forming species or invasive bioengineers	Occurrence	Number of habitats occupied/number of habitats	Abundance initially in monitored video transects; if under high risk, more detailed habitat specific sampling
	Unusual events not included above (e.g., harmful algal blooms, disease outbreaks, die offs)	Occurrence	NA	NA
	Change to the natural disturbance regime	Storm frequency, temperature anomalies, accidents and spills, sedimentation	NA	NA
	Shore line occupied by native vegetation	Proportion	NA	NA
	Catchment different land use	Proportion	NA	NA

Category	Indicator	MPA or broader scale	Across habitat scale	Within habitat scale
		What to measure	What to measure	What to measure
Nativeness				
	Areas of intensive marine activity in the vicinity of the MPA (e.g., fishing, mining, aquaculture, tourism)	Identification of activity and potential impacts	NA	NA
	Engineered structures in or near MPA	Presence and distance from MPA	Number of habitats occupied/number of habitats	NA
Pristineness				
	Assemblages of marine mammals; sea and shore birds; large predatory fish and invertebrates	Counts and assessment of species viability	NA	NA
	Number of species listed as threatened or at-risk under the New Zealand Threat Classification System	National lists	NA	NA
	Number and status of threatened or at-risk species in region	Regional lists and trends	Use of specific habitats by threatened or at risk species	NA
	Number, areal extent and diversity of habitat types	Number of habitats within MPA	Spatial extent, patch size and diversity derived from video transects	Density of habitat forming species or bio-traces from video transects
	Depth limits on macroalgae or seagrass	NA	Depth limits of plants and species identification from video transects and voucher specimens	NA
	Broad scale oceanographic features	Remote sense data on SST; wave climate; water column primary productivity	NA	NA
	Mixed layer depth	Calculation	NA	NA

Category	Indicator	MPA or broader scale	Across habitat scale	Within habitat scale
		What to measure	What to measure	What to measure
Pristineness				
	Eutrophication status	Bottom water oxygen concentration	Only to be measured in habitats where hypoxia can occur due to a combination of hydrodynamics, geomorphology and organic loading	NA
	Sediment draping of organisms and surfaces	NA	NA	Observation, notes on texture, colour and thickness from potentially affected habitats
	Quantities of marine litter (plastics)	Visual counts and definition of origin	Strandline surveys and video transects surveys	Abundance initially in monitored video transects
	Presence of un-natural underwater noise	Hydrophone records	NA	NA
	Anchoring	Surveys of boat anchoring by size of vessel	NA	NA
	MPA compliance	Number of warnings issues Number of persecutions Time spent on compliance monitoring	NA	NA
	Urban contaminant levels	Measured in sediments	NA	NA
	Resident organism contaminant levels	Measured in selected organisms	NA	NA
Diversity				
	Diversity of visible organisms and traces	NA	Richness across habitats from video transects	Richness within habitats from video transects
	Functional trait diversity of visible organisms and traces	NA	Functional diversity index applied across video transect data	Functional diversity index applied to video transect data
	Compositional variability of visible organisms and traces within habitats	NA	NA	β-diversity along video transects

Category	Indicator	MPA or broader scale	Across habitat scale	Within habitat scale
		What to measure	What to measure	What to measure
Diversity				
	Compositional variability of visible organisms and traces across habitats	NA	β-diversity across habitats based on video transects	NA
	Species richness	NA	NA	Species abundance data from core, grab or visual quadrate or scrape sampling
	Biological traits	NA	NA	Functional diversity index derived from core, grab or visual quadrate or scrape sampling
	Compositional variability in community structure	NA	NA	B-diversity from core, grab or visual quadrate or scrape sampling
	Key species	NA	NA	Counts and size structure from video transects
	Exploited population (fish/shellfish)	Population size/age structure	Current size and density compared to historical reference conditions	Abundance and size structure derived from BRUVs, pots or other surveys
	Fish species diversity	NA	NA	Species richness from BRUVs
Resilience				
	Production/biomass or size ratios	NA	NA	Counts and size/biomass estimates for key species and selected fish from video data and BRUVs
	Food chain length and trophic diversity	NA	Stable isotope analysis to define trophic structure across habitats	Replicate samples of organisms with habitats
	Presence of large and old organisms	NA	Defined from habitat type	Occurrence, abundance estimates from video data, age estimates from size, growth data or isotope methods as appropriate
	Redundancy within functional groups	NA	Occurrence, abundance estimates from video transects	Functional diversity index initially from video transects supported by core, grab or visual quadrat or scrape sampling when appropriate
	Recovery rates	NA	NA	Observation of natural recovery rate per patch size or experimentation

Category	Indicator	MPA or broader scale	Across habitat scale	Within habitat scale
		What to measure	What to measure	What to measure
Resilience				
	Resistance to disturbance	NA	NA	Characterisation of biological traits related to vulnerability/fragility/resistance to disturbance. Derived from species/abundance data derived from video data supplemented by core, grab or visual quadrat or scrape sampling
	Variability in spatial structure of community composition	NA	NA	Species /abundance data derived from video data supplemented by core, grab or visual quadrat or scrape sampling
	Flickering in time series	NA	NA	Species /abundance data derived from video data supplemented by core, grab or visual quadrat or scrape sampling
	Transport vs recycling of energy and matter	NA	NA	Experimental studies, models and isotope data
	Maintenance of feedback processes	NA	NA	Defined by structured equation modelling of key environmental characteristics and species-abundance data

Table 4: Data sources and measurement development for variables recommended for monitoring ecological integrity in marine ecosystems.

Indicator	Comment	Potential data sources	Status of measurement development	Reference
	Broad-scale variables that can be measured at the scale of the MPA or larger. These variables contribute to the assessment of ecological integrity at national, regional and local scales and provide context for comparison across MPAs and habitats and within habitats.			
Invasive species outbreaks	Recording the occurrence of outbreaks is feasible, while identifying and monitoring all invasive species in MPAs (including rare species) would be prohibitive	Occurrence of outbreaks for a select list are monitored regularly but only in specific places, although it is reasonable to assume that highly dense outbreaks of larger organisms would be noticed and reported to MAF BNZ by the public. Invasive species recognised as major threats are also regularly monitored in specific places but only recorded as present/absent Observation by MPA management staff and reporting from public	Methods for monitoring of the select list are specified by MAF BNZ A method to incorporate occurrences reported by general public, where the MPA is not routinely monitored, would need to be developed from presence only statistical techniques	
Nuisance species outbreaks	Some species are naturally present in the system but bloom under specific conditions, such as high nutrients or loss of high or meso-level predators	Occurrences of unusual events are usually publically reported, but no one agency is tasked with collating this information Observation by MPA management staff and reporting from public	As above, a method to incorporate reports would need to be developed from presence only statistical techniques	
Presence or spread of invasive habitat-forming species or invasive bioengineers	Here we would document occurrence of patches of resident habitat-forming invasives such as Pacific Oysters	Occupancy within habitats	Patch size along transects	

Indicator	Comment	Potential data sources	Status of measurement development	Reference
Invasive species recognised as major threats	Presence of species on the BIOSECURITY DANGER LIST	Observation by MPA management staff and reporting from public and observed during sampling	Standard techniques are available for transect sampling but as above a method to incorporate reports would need to be developed from presence only statistical techniques	
Unusual events (harmful algal blooms, disease outbreaks, die offs)	These may occur across a much broader scale that the MPA, but affects its integrity	Occurrence of unusual events are usually publically reported but no one agency is tasked with collating this information Observation by MPA management staff and reporting from public	As above, a method to incorporate reports would need to be developed from presence only statistical techniques	
Change to the natural disturbance regime	This may be reflected by changes in storm frequency, other climate related changes as well as sedimentation events, or changes in the levels of bioturbation and predator-induced habitat disturbance	Marine forecast records, remote sensing data, observation by MPA management staff and reporting from public	A method to incorporate such changes would need to be developed potentially based on number of exceedences outside the previously generated statistical distributions	
Shore line occupied by native vegetation	Occurrence and changes in occupancy of fringing habitat over time	Regional councils; aerial photographs; shore line surveys	Standard techniques	
Proportions of catchments in different land use	Occurrence and changes in occupancy over time	Regional council data bases, NIWA data-bases; CLUES	Standard techniques	
Areas of intensive marine activity in the vicinity of the MPA (e.g., fishing, mining, aquaculture, tourism)	Location relative to MPA, extent and nature of activity	Regional council and Mfish data bases, aerial photographs	Need to develop a standard technique similar to those above	
Engineered structures in or near MPA	Location relative to MPA, extent and nature of structure(s)	Regional council and Mfish data bases, aerial photographs	Need to develop a standard technique similar to those above	

Indicator	Comment	Potential data sources	Status of measurement development	Reference
Assemblages of marine mammals; sea and shore birds; large predatory fish and invertebrates	These organisms often sit high in the food web, play important trophodynamic roles and are often identified by the general public as indicative of a 'complete ecosystem'. The domain of most of these organisms is much larger than an MPA, although some may be transitory while others are resident for certain periods	Protected species monitoring, bird monitoring data sources, Mfish data bases, observation, sampling	Need to develop a standard technique consistent with the invasive species techniques	
Number of species listed as threatened or at-risk under the New Zealand Threat Classification System	To incorporate endangered species monitoring and to assess how endangered species lists change	Endangered species monitoring, bird monitoring data sources, Mfish data bases, observation, sampling	As above	
Number and status of threatened or at-risk species in region	To assess trends in status and define any mitigating factors (such as MPAs in species range)	Protected species monitoring	As above	
Depth limits on macroalgae or seagrass	Plant depth distributions are limited by turbidity	Video sampling	Commonly measured in EU WFD monitoring	(Mangialajo et al. 2008, Mangialajo et al. 2007)
Broad scale oceanographic features	Key variables from remote sensing include SST and chlorophyll, while further development of others (e.g., turbidity) is in process, and will lead to further broad scale metrics of productivity and other oceanographic features.	Remote sensing data and its application is developing and we can expect it to play an increasing role in the future.		
Mixed layer depth	Reflects the depth to which water column productivity is directly connected to seafloor habitats	Mixed layer depths are available at 250m resolution across the EEZ but are probably not correct near the coast due to the algorithims used		

Indicator	Comment	Potential data sources	Status of measurement development	Reference
Quantities of marine litter (plastics)	Adverse effects due to ingestions etc. Plasticisers released, particularly from micro-particles, can also function as hormonal mimics	Observation; community MPA clean up initiatives		
Presence of un-natural underwater noise	There is growing concern associated with noise and EMF from cables and power generation structures in the marine environment. Boating, shipping, mining and certain types of acoustic surveying can also affect the behaviour of marine species	Hydrophone deployment	DOC works on this, could need to tap into NOAA network. Also some specific work on reefs at UOA	
MPA compliance		Compliance monitoring and legal documents		
Resident organism contaminant levels	Sampling of common species with the potential for bioaccumulation	Regional council data bases; Sampling	Standard techniques available	
	Habitat-scale variables, these are all expected to require field observations			
Eutrophication status	This is not expected to be relevant to all locations, but it may be a future issue in some areas	Observation and sampling bottom water oxygen concentrations	Well established techniques. Commonly measured in EU WFD monitoring	
Sediment draping of organisms and surfaces	Impacts on plant photosynthesis and feeding behaviour and efficiency of suspension feeders and deposit feeders	Video imagery	Well established techniques	(Ellis et al. 2004, Hewitt & Pilditch 2004, Lohrer et al. 2006, Norkko et al. 2006, Schwarz et al. 2005, Thrush et al. 2004)
Anchoring	As a direct form of bottom disturbance and an indicator of boating activity in the MPA	Observation Some MPAs are surveyed for anchoring statistics	Need to define a sampling protocol; possible link to NABIS data from aerial surveys (Ministry of Fisheries)	

Indicator	Comment	Potential data sources	Status of measurement development	Reference
Diversity of visible organisms and traces	This would reflect the degree of bioturbation and thus some important elements of ecosystem function in sedimentary habitats but also indicate the presence of certain functional groups representing large burrowing organisms	Video sampling	Categories need to be identified, plenty of expertise available for doing this in coastal and deep-sea habitats in NIWA, DOC and Universities	
Functional trait diversity of visible organisms and traces	Biological traits analysis is well developed in the marine literature, but mostly based on sample x species data matrices. We would need to convert this into matrices derived from video data to reflect traits associated with size, age, vulnerability to disturbance, feeding, modifications of hydrodynamic conditions; changes to sediment biogeochemistry, habitat formation etc.	Video sampling	Needs to be developed but skills and experience available in NIWA	(de Juan et al. 2007, Hewitt et al. 2008)
Compositional variability of visible organisms and traces within habitats	Detailed interpretation of the video imagery to define variability within and between habitats (β-diversity)	Video sampling	Well developed	(Hewitt et al. 2008, Hewitt et al. 2004, Thrush et al. 2006, Thrush et al. 2010)
Compositional variability of visible organisms and traces across habitats	Detailed interpretation of the video imagery to define variability within and between habitats (β-diversity)	Video sampling	Well developed	(Hewitt et al. 2008, Hewitt et al. 2004, Thrush et al. 2006, Thrush et al. 2010)
Exploited population (fish/shellfish)	We expect these to increase in MPAs where protection levels are affected and the region was previously exploited. They can lead to potential trophic cascades affecting broader community and ecosystem dynamics in the MPA	Derived from video or diver counts and BRUVs	Well established techniques but we recommend the BRUVs as developed by Euan Harvey at University of West Australia	http://www.uwa.edu.au/people/euan.harvey http://www.marine.uwa.edu.au/recherche/ (Shears, N.T. & Babcock 2003, Shears, N. T. et al. 2008)
Fish species diversity	Expected to change particularly in soft-sediment habitats as food resources and habitat heterogeneity changes with protection	Derived from BRUVs		

Indicator	Comment	Potential data sources	Status of measurement development	Reference
Production/ biomass and size ratios	We expect populations to contain more large and old individuals and communities to contain more large and long-lived species	Can be derived from field data and national/international data bases	Well developed in fisheries ecosystem-based management literature	(Hiddink et al. 2006, Jennings et al. 1999, Jennings et al. 2002a, Jennings et al. 2002b)
Food chain length and trophic diversity	Energy transfer within ecosystems and the nature of trophic connections have important consequences of dynamics	Sampling of specific food webs and specific components of those food webs and stable isotope analysis	Techniques are well developed although this would be a new application. Expertise in NIWA and UOO. This may be too expensive for regular monitoring, but should be part of a nationally accumulated data-base to verify cheaper and cruder techniques	
Presence of large and old organisms	The presence of such organisms is an indicator of disturbance history, such organisms often play important roles in defining ecosystem structure and function	Video sampling, may need some ground truthing with appropriate age estimates	Techniques available	
Redundancy within functional groups	Information on the range of species that perform one type of function (e.g., deep burrowers) provides information on resilience and efficiency of process as well as indicating individual species that play important functional roles	Video transects with interpretation supported by process studies	For video data functional groups need to be defined – but this is achievable	
Recovery rates	Often taken as an indicator of resilience and can be used to demonstrate the potential for cumulative impacts. Experiments are needed to measure rates effectively. From one perspective the creation of MPAs in impacted areas is this experiment, but smaller scale disturbance recovery experiments could also be informative – albeit expensive	Conduct experiments and interpret time series from MPA monitoring	Theory is developed and some testing is being performed in research programmes	(Thrush & Dayton 2010, Thrush et al. 2008, Thrush et al. 2009, van Nes & Scheffer 2007)

Indicator	Comment	Potential data sources	Status of measurement development	Reference
Resistance to disturbance	Use biological traits analysis to identify the proportion of species with traits related to either vulnerability or resistance to disturbance		Demonstrated in publications particularly with respect to fishing impacts	(de Juan et al. 2007, Hewitt et al. 2008)
Variability in spatial structure of community composition	Variation in community composition as indicated by dispersion in ordination space has been linked to resilience and response to disturbance and stress	Time series data from video transects or time series of traditional sampling (cores, grabs, quadrats)	Demonstrated in a few publications but needs further verification, which would be possible once appropriate time series are available	(Thrush & Dayton 2010)
	Detailed measurements that will require intensive sampling, it is envisioned that this would be used only in specific cases to provide extra data to confirm changes or verify coarse resolution techniques			
Species richness	Determined from samples by species data matrices. Only applied in certain circumstances – for example to ground-truthing more cost-effective and broader scale sampling or investigate a problem identified by monitoring or other activity	Derived from detained traditional core, grab or quadrat sampling.	Well established techniques	
Biological traits	Determined from samples by species data matrices.	Derived from detained traditional core, grab or quadrat sampling.	Well established techniques	
Compositional variability in community structure	Determined from samples by species data matrices.	Derived from detained traditional core, grab or quadrat sampling.	Well established techniques	
Key species	Counts and size structure analysis	Derived from detained traditional core, grab or quadrat sampling.	Well established techniques	
Flickering in time series	Complex system theory predicts that before a system jumps to an alternate state, variability in the time series increases	Time series data from video transects or time series of traditional sampling (cores, grabs, quadrats)	Demonstrated in complex system models, with one empirical example in a marine ecosystem	(Hewitt & Thrush 2010, Thrush & Dayton 2010)

Indicator	Comment	Potential data sources	Status of measurement development	Reference
Transport vs recycling of energy and matter	As we develop better functional indicators we will be able to better define the proportion of reprocessing and recycling of energy and matter within a system versus export and shift into a different state (e.g., anaerobic)	Isotope data and functional indicators	Needs development	
Maintenance of feedback processes	Communities and ecosystems function through interaction webs and in many cases positive feedback processes are present that highlight the potential for small changes in conditions to lead to rapid shifts in ecosystem structure and function	Time series data from video transects or time series of traditional sampling (cores, grabs, quadrats) plus environmental data	Under development for some systems	

2.3 Pathways to the development of an ecological integrity index

Integrating individual variables into an overall assessment of ecological integrity is a major challenge, recognised by both Lee et al. (2005) and Schallenberg et al. (2011). Although beyond the scope of this report, we want to consider possible ways forward. Schallenberg et al. (2011) propose an approach based on the use of regression trees, but do not offer any proposal of what form this may take and we comment no further on this.

There are a number of statistical approaches to linking the different integrity measures together, but as the different measures we propose are sampled over different scales and have different values (weights) in terms of what they mean in terms of ecological integrity, an approach based on solely on statistical considerations would be problematic. We think that an overall ecological integrity indicator should use statistical methods appropriately, but should also incorporate our (hopefully) expanding knowledge of how the different indicators reflect different elements ecological integrity over different space and time scales. The simplest integration would be the calculation of an average value for each of the measured variables, within the four pillars of ecological integrity (nativeness, pristineness, diversity and resilience), which are then summed, thus reaching a highest value of four. However, Bayesian hierarchical modelling approaches may well be more useful for synthesis and interpretation, with their inherent ability to incorporate not only both empirical and modelled variables but also uncertainties at different levels. Regardless of the integrative method used, two constraints occur. All variables must ultimately be rated to have high values in states that we consider reflect increasing ecosystem health or integrity. Variables must also be weighted to indicate the relative importance of their contribution to integrity (Table 5). Here, we use the weight to indicate both positive and negative contributions to ecological integrity.

Thus using a simple additive approach would equate to:

EQN (1) $EI = N + P + D + R$

Where

EQN(2) $N = \text{average}\ (V_1 {*}W_1, V_2{*}W_2, \ldots . V_n{*}W_n)$ and V_{1-n} are variables representing nativeness and w_{1-n} are their weights, and similarly for P (pristineness), D (diversity) and R (Resilience).

In keeping with the hierarchical approach to the assessment of ecological integrity, the first level of assessment will be based at the MPA scale (rather than at the scale of individual habitats within the MPA). This level of assessment is practical because many types of data are only available at this scale e.g., water column primary productivity, presence of apex predators, and because many types of stressors are not confined to individual habitats within MPAs e.g., harmful algal blooms, sediment loadings. This upper level of assessment also provides a means by which MPAs can be compared more reasonably, as a shallow estuarine MPA may not have much in common with a deep offshore island MPA (and some data categories, such as mixed layer depth and macroalgal depth limit, may not be applicable or informative in all reserves). In other words, it is advisable to contrast the ecological integrity of reserves relative to other reserves of the same basic bioregion and type (e.g., “northeast New Zealand coastal reef reserves”). The ordination plot (Figure 5) exemplifies some of the potential MPA categories and how they can be differentiated. This approach can be advanced by the application of the risk assessment and environmental characterisation procedures.

The second level of the hierarchy is aimed at assessing the integrity of individual habitats within reserves. These types of assessments would require data collection. Again, it will be important to assess integrity fairly across widely differing habitat types (e.g., subtidal surf zone sand versus subtidal rocky reef). This information provides the opportunity to produce a habitat-related index of EI (EIH), with EIH for different habitats compared between MPAs or trends in a specific habitat (for example a habitat considered to be at high risk) in a specific MPA monitored over time. This habitat specific information could also be factored into an overall MPA integrity index; in this case weighting by the proportional areal contribution of individual habitats to the MPA would be necessary. This weighting should be applied to each individual variable, which are then treated as per EQN(2) . The approach can be applied in the context of either a one-off assessment that compares integrity across locations or it can be applied in a time-series approach.

We have left out some of the more detailed sampling and experimental approaches to assessing integrity in the overall index, to ensure a cost-effective approach that can be widely applied. However, more detailed analysis may be important both to validate coarser measurements and to gain a more detailed understanding of ecological integrity. For all the variables we have identified we need to consider how they might affect our overall assessment of integrity. This can be thought of as asking whether, for example, losses of apex predators, or habitat structure, or diversity or changes in the disturbance regime all affect our sense of integrity to the same extent (Table 5). This will be important to consider for each variable, relating to how it is measured and how we might expect changes in both the mean and variance of estimates to influence our assessment of ecological integrity.

Table 5: Integrity variables and weightings. Positive weighted scores indicate that the variable considered have a positive effect on ecological integrity.

Broad-scale variables measured at the scale of the MPA or larger. These variables contribute to the assessment of ecological integrity at national, regional and local scales and provide context for comparison across MPAs and habitats of measurements made within habitats.	**Contribution to ecological integrity weighting range: 1: low to 7; high**
Invasive species outbreaks	-5
Nuisance species outbreaks	-5
Occurrence of Invasive species recognised as major threats	-6
Unusual events (Harmful algal blooms, disease outbreaks, die offs)	-7
Change to the natural disturbance regime	-7
Shore line occupied by native vegetation	+3
Proportions of catchments in different land use	-3
Areas of intensive marine activity in the vicinity of the MPA (e.g., fishing, mining, aquaculture, tourism)	-3
Engineered structures in or near MPA	-3
Assemblages of marine mammals; sea and shore birds; large predatory fish and invertebrates	+6

Broad-scale variables measured at the scale of the MPA or larger. These variables contribute to the assessment of ecological integrity at national, regional and local scales and provide context for comparison across MPAs and habitats of measurements made within habitats.	**Contribution to ecological integrity weighting range: 1: low to 7; high**
Number of protected species listed	+3
Number of protected species with declining status	-5
Number, areal extent and diversity of habitat types	+7
Depth limits on macroalgae or seagrass	+5
Broad scale oceanographic climate trending to more stressful conditions	-6
Mixed layer depth increasing	+3
Quantities of marine litter (plastics)	-6
Presence of un-natural underwater noise	Unknown
MPA non-compliance	-5
Resident organism contaminant levels	-4
Habitat-scale variables, these are expected to require field observations	
Eutrophication status	-5
Sediment draping of organisms and surfaces	-6
Anchoring	-5
Diversity of visible organisms and traces	+6
Functional trait diversity of visible organisms and traces	+7
Compositional variability of visible organisms and traces within habitats	+4
Compositional variability of visible organisms and traces across habitats	+7
Exploited population (fish/shellfish) increasing	+6
Fish species diversity	+6
Production/ biomass and size ratios increasing	+6
Food chain length and trophic diversity	+6
Presence of large and old organisms	+7
Redundancy within functional groups	+7
Recovery rates	Unknown
Detailed measurements that will require intensive sampling, it is envisioned that this would be used only in specific cases to provide extra data to confirm changes or verify coarse resolution techniques	
Variability in spatial structure of community composition	+7
Species richness	+7
Biological traits	+7
Compositional variability in community structure	+7
Decrease in key species	-6
Flickering in time series	-7
Transport vs recycling of energy and matter increases	-7
Maintenance of feedback processes	+7
Actual measurements of the location and size of specific habitat patches over time decrease	-7

However, while it is important to be able to summarise all these measures of integrity into an overall estimate of an MPA, it is equally important that any such measure can be decomposed so that we can understand what particular elements of integrity are driving observed changes. This is imperative if we are to understand the implications of change in any index value, link it to management actions and use it for public dissemination.   Here we see a strong role for multivariate techniques. Multivariate techniques are inherently graphical and lend themselves well to visualising factors influencing differences either between locations or at one location over time. These can be employed at any scale within the hierarchy. For example, multivariate analyses of summaries over the six different components of integrity could be used to determine which if any is driving change. This could be followed by an analysis of separate measures at an across and within habitat scale.

While we are attempting to develop some overall synthesising index of integrity for simple reporting and interpretation purposes, we think it is vital to monitor trends and conduct spatial comparisons, on each of the individual variables. This is important as understanding these trends are likely to lead to appropriate management actions, evaluation of their relative success, and to understanding the potential ecological importance of trends in the overall index.

3 Part 3: Summary and recommendations for future development

In this report we have reviewed the concept of ecological integrity as applied to terrestrial and freshwater ecosystems by DOC and considered its application in a marine, and primarily MPA, context. We have reviewed the high-level policy and scientific literature on the application of ecological integrity, and related concepts, in the marine environment internationally. On the basis of this specific analysis and our experience with New Zealand's marine ecosystems we have identified a suite of key marine ecological indicators. However, we have gone beyond this as we consider that, to make the monitoring of ecological integrity of the marine environment cost effective in a New Zealand context, we need some novel and innovative thinking. Thus, we have provided an overarching framework in which ecological integrity monitoring for the marine environment could be developed. This includes identifying the ways in which our list of proposed indicators could be integrated, to develop an effective integrity index, which allows assessment of status and trends in ecological integrity and assists in the effective development of ecosystem-based marine management strategies for our marine ecosystems. While such indicators are likely to vary in composition between terrestrial, freshwater and marine ecosystems, there is a good opportunity to develop integrity indices for different ecosystems in parallel and link them to identify connections from the hilltops to the ocean troughs.

Specifying the details of DOCs progress to monitoring ecological integrity in New Zealand's MPAs is beyond the scope of this report but we make the following general recommendations.

We recommend:

Management actions (procedural)

- Develop the preliminary risk and environmental characterisation procedure and use a multivariate analysis on the environmental characteristics (including those used to define MPA habitat classes in the MPA Implementation policy) to group MPAs with similar characteristics.
- Develop a nationally coherent series of habitat definitions that can ensure consistency across sites.
- Use this list of habitats to refine habitat mapping of existing MPAs, and determine potential problems in the habitat definitions. Following this, some habitat definitions may be altered.
- Assess the fitness-for-purpose and spatial/regional variability in availability of broad-scale data, including the frequency at which this data is updated.
- Conduct a broad-scale assessment of the ecological integrity of MPAs based on available broad-scale data.
- Make a preliminary selection of a subset of MPAs, informed by the preliminary risk and environmental characterisation procedure and practical considerations, to use for developing, method testing and validation of the integrity monitoring programme.

- From this subset, determine a smaller selection of MPAs to act as sentinel sites for long-term monitoring.
- Consider how to include remote MPAs (e.g., Kermadecs, Auckland Islands, future off shore and Ross Sea) in the integrity monitoring programme – perhaps via site specific benefactor support.
- Within the initial 5 year period, establish sampling protocols for all measured variables within the MPAs and adjacent reference sites. This will include defining the number of sites, the size of the sampling units, the required level of replication to ensure precision at appropriate spatial scales and the required level of sampling needed to reasonably estimate richness (of species, habitats, functional groups). This bullet will require investment in some operational capacity (see operational action section below).
- Develop a long-term strategy so that over time data on both ecological integrity and habitat distributions can be collected from all MPAs on a rotational basis.
- Develop data management and decision-support principles to ensure that the data sets are secure and the information used to its maximum potential (see operation actions below).
- Validate and support this process by international scientific peer review.

Operational actions

- Consider the development of a Technical Advisory Group (expert working group of key knowledge holders of New Zealand's marine environment) to advise on nationally coherent series of habitat definitions.
- Development of a cost-effective sampling design for each of the indicators selected (Table 5).
- Implementation of the preliminary desktop exercise considering the environmental setting and risks to the specific MPA.
- Selection of MPAs and initial monitoring sites (including potential sentinel sites.

Capacity building actions

- Ensure that new and relevant advances in ecosystem science, sampling techniques and remote sensing can be included in the monitoring programme as they become available.
- Develop metrics and reporting processes for reporting on ecological integrity and local, regional, national and international levels.
- Development, specification, testing and ground truthing of video sampling techniques to ensure cost-effective monitoring.
- Development of methods to link all measured variables into an overall ecological integrity index and refinement and validation of the proposed method of enabling comparisons between MPAs and between times.

- Identification of trigger levels in monitored variables or summary indices that generate specific management responses.

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