Headline indicators for the New Zealand ocean

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Executive Summary

Headline indicators are used in New Zealand to measure and disseminate information on progress towards sustainable management of the environment, including human impacts on, and the state of, the New Zealand ocean domain, defined here as the area outside territorial waters but within the New Zealand Exclusive Economic Zone (EEZ). Headline indicators form an important part of the 5-yearly "State of the New Zealand Environment" reporting co-ordinated by the Ministry for the Environment. Recent research (both funded by Ministry of Fisheries, and within the FRST Coasts & Oceans OBI) provides an opportunity to improve the usefulness of state of the environment indicators for the New Zealand ocean. We describe and evaluate over 35 candidate indicators of the *pressure* on the New Zealand ocean, the *state/impact* of ocean ecosystems, and the *response* of institutions, policy and society to promote sustainability. From these candidate indicators, we recommend that 16 indicators (4 pressure, 8 state, 5 response) be considered further in the run up to 2012 state of the environment reporting (see table below; grey panels are already used as headline indicators in New Zealand). Substantial but tractable research, and multi-agency input and evaluation, will be required to determine which of the proposed new indicators may be useful, and to develop these for use.

Type	Indicator	Description
Pressure	Total fishing removals	Total landings by weight and category of marine biota
	Commercial trawling footprint	Area of seabed trawled in fishing year (trawl footprint).
	Area Trawled Index, ATI	Weighted average proportion of BOMEC (Benthic – optimised MEC) regions trawled in fishing year
	Biomass Trawled Index, BTI	Proportion of demersal fish biomass in trawled areas, combined by relative ecological importance
State / Impact	Stock Status Index, SSI	Status of QMS stocks against management targets, combined by relative ecological importance
	Threat status of marine species	Ecological status of marine species (DOC classification)
	Marine Trophic Index, MTI	Average trophic level of catches taken by bottom trawling, based on research and commercial trawls
	Oceanographic change observed by satellites	Characteristic oceanographic modes of variability based on satellite temperature, colour and altimetry
	System-level Catch-Per-Unit-Effort	Catch-Per-Unit-Effort for all species taken by bottom trawling, corrected for changes in effort
	Demersal Fish Diversity, DDI	Multi-index measure of diversity in demersal fish taken by commercial bottom trawling
	Feeding type ratios	Relative change in functional groups (type of feeding) taken by commercial trawling
	Species distributions	Change in spatial distribution of fish species between years
Response	Total area with some form of protection	Proportion of the oceanic EEZ with some protection
	Overfishing Index, OFI	Proportion of QMS stocks where overfishing is currently known to be occurring
	State of Knowledge Index, SKI	Assessment of state of knowledge of status of stocks in the QMS, combined by relative importance
	Area Protected Index, API	Weighted average proportion of each ecologically distinct (BOMEC) subregion protected from bottom trawling
	Biomass Protected Index, BPI	Weighted average proportion of finfish (and potentially squid) biomass in areas protected from fishing

1 Introduction

Indicators are used around the world to measure and disseminate information on progress towards the sustainable management of human impacts on marine ecosystems (OECD 1993, 1998; Garcia & Staples 2000). As for other aspects of ecological sustainability "if you cannot measure it, it does not count" (Hanson 2003). Headline indicators try to reduce the multidimensional complexity of measuring progress towards sustainability to a level where they can be understood by policy makers, the general public, and other stakeholders with a non-technical background (Patterson 2002). As in other nations such as Australia, Canada, USA and Sweden (Griffith 1997; Vandermeulen 1998; Ward 2000), New Zealand reports headline indicators via "State of the Environment" reporting (MfE 2007). In New Zealand, this reporting occurs every 5 years and the Ministry for the Environment is primarily responsible. Headline indicators sacrifice specificity for generality, and Rice & Rivard (2007) note that they are designed for audit ("how are we doing?") rather than control ("what should we do in the future?"). They aim to provide evidence of the effectiveness of current management practice, and show whether there is a need for a change in policy or its implementation. If action is required, more specific indictors and analysis are needed to infer causality and determine what the appropriate action should be (Rice 2000; Link 2005; Rice & Rivard 2007).

A huge number (>300) of marine ecosystem indicators are in use or proposed around the world (Cury et al. 2005; Rochet & Rice 2005; Rice 2003), with consensus that a suite of indicators is needed to measure progress towards the sustainable management of the impact of human activities on marine ecosystems (Cury & Christensen 2005; Rice & Rochet 2005). In this report we assess which indicators are likely to be most feasible and useful as headline indicators for assessing the sustainability of human impacts on the New Zealand ocean. Although much of the focus of this report is on fishing, marine ecosystem indicators should also capture the effects of human actions such as ocean mining and climate change. We define the New Zealand ocean as being the area outside the 12 nautical-mile (19.3 km) territorial limit but inside the New Zealand Exclusive Economic Zone (EEZ) (Figure 1). New Zealand has the 5th largest (EEZ) zone in the world, extended in 2008 from 4.4 m km^2 (oceanic part 4.17 m km^2) to 6.1 m km^2 (UN 2008) (oceanic part 5.87 m km^2).

Here, we are primarily concerned with ecological (rather than economic or social) sustainability. Ecological sustainability is taken to mean that the structure, function and resilience of ecosystems are maintained so that they continue to provide ecosystem services in the future under likely conditions of environmental variability and change. Where directly impinging on ecological sustainability, we also consider those economic and societal aspects that are likely to impact the ability of New Zealand to maintain and improve marine ecological sustainability. However, the legislative and policy context of management of the New Zealand ocean is too complex to be considered in detail here, with 25 Acts of New Zealand parliament, 15 Government strategies and major policies, and 34 international multilateral agreements that relate to oceans management in New Zealand (Willis et al. 2002). We note that an overarching government oceans policy for New Zealand is absent.

How should we select headline indicators for the New Zealand ocean? International working groups on indicators for the marine environment (e.g. Cury & Christensen 2005) recommend that we should aim for a suite of indicators which are sensitive to changes in a variety of factors including: (1) oceanographic and climate conditions; (2) low and middle trophic levels (plankton, pelagic and benthic invertebrates); (3) upper trophic levels (fishes); (4) marine predators (seabirds, marine mammals); (5) aggregate indicators working across trophic levels. The total number of marine headline indicators needs to be no more than about 6 to 10 or else uptake by policy makers and the public is likely to be reduced. There is good consensus on the attributes desirable in indicators. Criteria used to assess the utility of indicators include policy relevance, timeliness/cost of production, accuracy and precision, scientific validity, sensitivity, responsiveness, consensual basis, formal (legal) foundation, specificity, and geographical scope (Garcia & Staples 2000; Rice & Rochet 2005). Here, the evaluation of indicators is based on six criteria used by the New Zealand Ministry for the Environment (Table 1, MfE 2007). In addition to satisfying these criteria of course, any proposed indicators would need to withstand scrutiny by New Zealand stakeholders before they could be adopted into the New Zealand reporting framework.

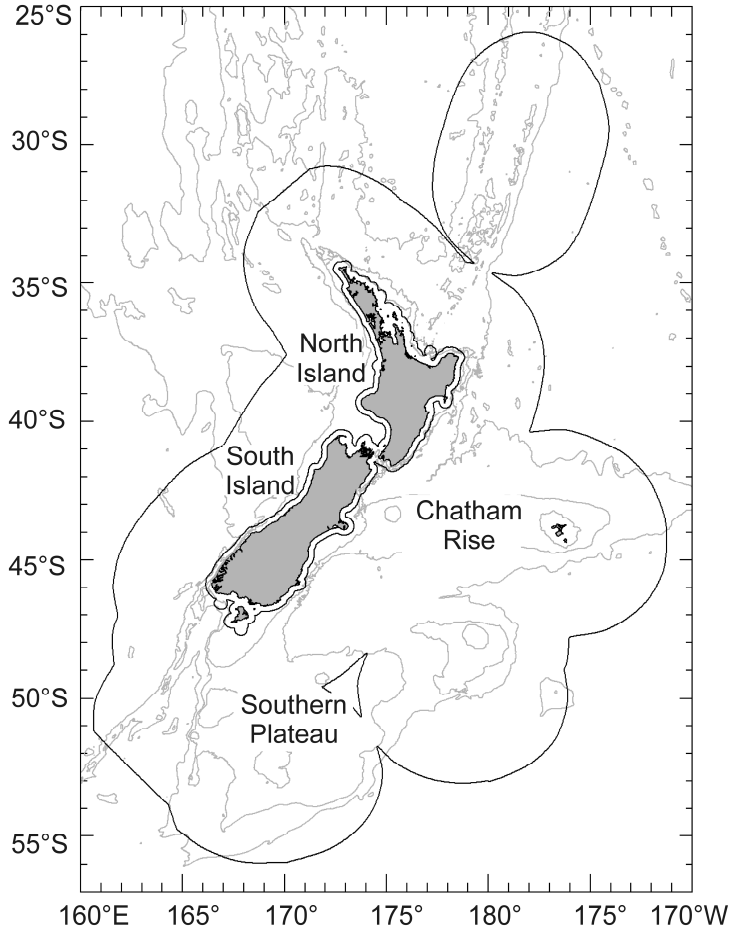

Figure 1. Study area: the New Zealand EEZ (before changes in 2008). Also shown is the bathymetry (depth contours at 250 m, 1000 m and 3000 m), 12 nm territorial limit around the New Zealand mainland, Chatham Rise, and Southern Plateau.

Table 1. Criteria for assessing indicators (based on MfE 2002).

	Criteria	**Description**
1	Nationally significant	Does the indicator give information at the scale of the New Zealand EEZ?
2	Relevant	Is the indicator measuring something of importance in terms of assessing progress towards sustainability?
3	Credible	Are the underlying data, methodology and assumptions scientifically robust? Does the indicator stand up to scientific scrutiny as unambiguously measuring progress towards sustainability?
4	Interpretable	Will non-technical stakeholders be able to interpret what the indicator is showing? Are historical data available to allow the indicator to be put into a medium-term context?
5	Cost-effective	Are the data required available in a timely fashion? Is it likely that data will continue to be collected in the medium to long term? How much additional data/research is required to develop the indicator?
6	Internationally comparable	Have similar indicators been used overseas so that New Zealand performance can be benchmarked against international experience?

2 Candidate indicators

2.1 DPSIR framework

The Driver-Pressure-State-Impact-Response (DPSIR) framework used in New Zealand for state of the environment reporting (MfE 2007) groups indicators in terms of whether they give information on *drivers* of change (social, demographic, economic developments), the *pressure* on the system exerted by human activities, the present *state* of the environment including trends in the current state, *impacts* (effects on environmental or human health), or the *response* of management to promoting sustainability (OECD 1993; Garcia & Staples 2000; MfE 2007). Here, we do not consider drivers explicitly and group the other types of indicator as pressure, state/impact, and response. We are less concerned with why pressures have come, and more concerned with measuring their intensity and effect. We combine state and impact because separating impacts from state requires causality to be determined – we need to know what has caused a change in state to determine if it is an impact of human activity or part of natural variability – and this is outside the scope of the present report.

Pressures include *inter alia* the individual and combined effects of fishing, climate variability and change, pollution and extractive industries. Worldwide, fishing has had profound impacts on marine ecosystem state and function (Halpern et al. 2008; Pauly et al. 1998a; Jackson et al. 2001), and pressure due to fishing in New Zealand is also high. In the New Zealand EEZ, more than 55% of cells of size 25 km^2 shallower than 1600 m were contacted by bottom trawling between 1989–2005 (Baird et al. 2009). Many deep water fish species targeted by commercial fishing are thought to have had their spawning biomass depleted by 30–80% since commercialised fishing started in the 1970s, (Ministry of Fisheries 2009). Some threats to the sustainability of New Zealand marine ecosystems due to fishing have been mitigated by including 82 fish and invertebrate species (as of 2008) in the New Zealand Quota Management System (QMS), including management of fishing capacity and explicit limitation of fishing mortality for QMS species (Mace 2001; Aranda & Christensen 2009). However, fishing at near Maximum Sustainable Yield (MSY) levels as is the target in New Zealand does not necessarily protect overall ecosystem state or function (ICES 2005), with potential for chronic, cumulative degradation of the marine food-web (Cury & Christensen 2005; Jennings et al. 2002; Jackson et al. 2001; Branch 2009), also called ecosystem erosion or ecosystem overfishing (Murawski 2000; Coll et al. 2008). As Duplisea & Castonguay (2006) state: "if we are to eventually define community or ecosystem sustainability…it will most likely come about through combining various indicators of the fish community". Indicators are hence needed that can detect the effects of fishing on ecosystem state and function over the long-term.

Environmental drivers can impact ecosystems as at least as strongly as fishing (Mackinson et al. 2009; Frank et al. 2007; Schiermeier 2004), and can act synergistically with fishing (Winder & Schindler, 2004; Kirby et al. 2009). Oceanographic state and variability are likely to become increasingly important drivers of marine ecosystem change in New Zealand in the medium to long term as global climate change continues (Willis et al. 2007; Polunin 2008). Effects may be manifested through *inter alia* warming of ocean waters affecting species biology and ecology (O'Connor et al. 2007; Perry et al. 2005), regime shifts (large-scale and persistent changes in ocean circulation and vertical water column structure, Mullan et al. 2001), increased likelihood of invasive species (Willis et al. 2007), increasing ocean acidification (Fabry et al. 2008; Cooley & Doney 2009), and effects across multiple trophic levels due to timing of productivity (Sydeman & Bograd 2009).

State / impact indicators for the marine ecosystem summarise information on the health of organisms which live in or otherwise depend on the marine environment, interactions between organisms (both predator-prey and indirect, such as interference and behaviour modification), interactions between biota and the physical environment (including the dependence of organisms on habitat), and the overall viability of communities of organisms within the physicochemical environment. Measuring state of the marine environment also implies consideration of the time derivative of its present state: how are things changing? A single evaluation of an indicator of ecosystem health is far less valuable than a time series of such measurements as the latter gives historical contextual information on the indicator, including its variability, the current rate and direction of change, whether this is

accelerating, and whether cycles or oscillation is apparent.

Response indicators track the extent to which institutional, policy and societal actions act to promote sustainable development. Indicators of institutional state have been used as part of a "Genuine Progress Index" approach around the world to show progress towards sustainability (Hanson 2003; OECD 2003). Conceptually, the state and capability of the fishing industry, fishing management system, and fisheries research/knowledge base are fundamental parts of the picture of how well New Zealand is progressing towards sustainable management of its ocean.

2.2 Pressure Indicators

2.2.1 Total trawl effort

Habitat alteration is a core OECD environmental indicator (OECD 2003). The greatest habitat alteration in the New Zealand ocean is likely to be by bottom trawling by commercial fishing vessels which is known to be particularly destructive to some benthic communities (United Nations 2006; FAO 1995, 2003). Total area trawled was reported as part of the New Zealand state of the environment reporting in 2007 (MfE 2007). In a recent update of 1 million records for trawl effort based on Trawl Catch Effort Processing Returns (TCEPR) since 1989 (Baird et al. 2009), each trawl track has been converted to a polygon equivalent to the estimated area swept by the fishing gear and overlaid on 25 km^2 grid-cells, giving a relative representation of the total trawl effort (Figure 2). TCEPR have been required to be used by large (>28 m) New Zealand fishing vessels to report the location of all trawling effort since 1989, and nearly half (ca. 46%) of smaller vessels (<28 m) have also used this method of reporting since the mid 1990s (Baird et al. 2009). Before this time, and for the other smaller vessels (<28 m), the location of trawling is not known accurately as it was only reported in terms of New Zealand Fishery Statistical Area or using a start position. Since 2007/8, vessels <28 m in length are obliged to report the start location of trawls. Although this change is welcome, the fact that vessels still do not need to give the location that the trawl finished means that the position of all commercial bottom trawling in the New Zealand EEZ is not known, and fishing effort data remain frustratingly incomplete. This issue predominantly affects territorial (coastal) waters with depths <200 m as this is where most bottom trawling by smaller vessels occurs, so that the data summarised by Baird et al. (2009) are suitable for developing an indicator of total trawl effort in the New Zealand ocean domain.

Trawl effort can be reported either as number of grid-cells visited each year (irrespective of the number of times visited or the proportion of the grid-cell trawled), "cumulative area trawled" (defined as the total area swept by trawls irrespective of whether trawls overlay on previous trawls that year), or the annual "trawl footprint" (defined as the area of seabed trawled at least once in a given year (Baird et al. 2009). The seabed area trawled in most grid-cells is quite low (less than 1 km^2 trawled in more than 50% cells: Baird & Wood 2009), so measuring the number of grid-cells visited would overestimate the benthic modification due to trawling. Note that the effect of fishing disturbance on benthic organisms and communities) depends on substrate type, frequency of trawl contact, gear type, and the community affected, so that measuring trawl effort indicates the degree of modification to the benthic environment and organisms due to fishing, rather than the *impact* or consequence of this action on benthic communities. The recovery of benthic ecosystems after trawling is likely to take more than a year (National Research Council 2002); once an area of seabed has been trawled in a given year, more trawls in the same area are likely to cause less change. This means that the modification of seafloor environment by trawling is likely to be better represented by the trawl footprint than the cumulative area trawled. The trawl footprint based on data in Baird et al. (2009) is likely to underestimate pressure due to trawling for two reasons. First, the area of seabed with ecosystems adversely affected by trawling will be greater than the trawl footprint because sediment plumes/displaced material is likely to have impacts outside the area actually swept by the trawl. Second, the positions of about half of all trawls carried out by vessels smaller than 28 m are not recorded, reported or included in the analysis of Baird et al. (2009).

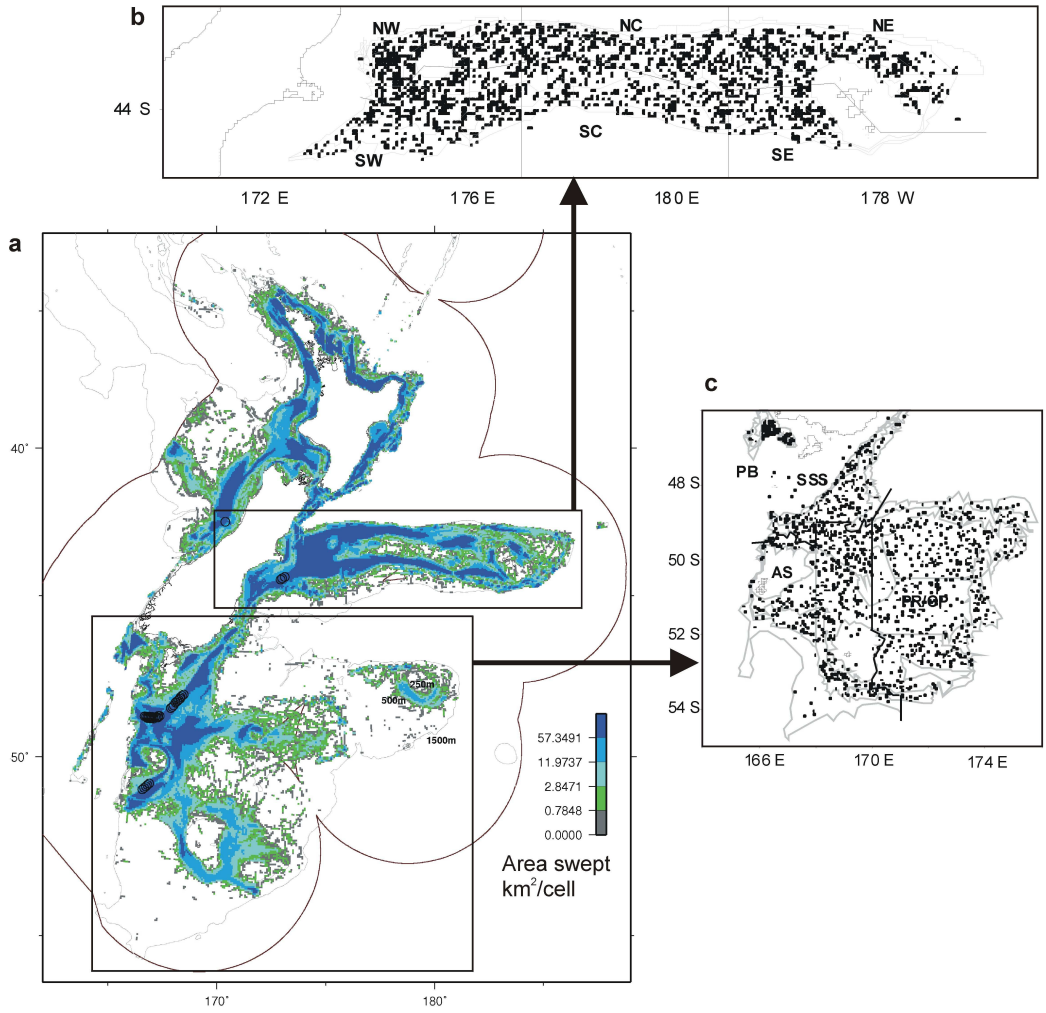

Figure 2. a: Fishing trawl effort for the New Zealand EEZ (Baird et al. 2009). In addition to commercial trawling, research trawls have been repeatedly carried out in two of the most intensively fished parts of the New Zealand oceanic EEZ. **b**: Chatham Rise trawl survey series (1992–2007, Stevens & O'Driscoll 2007). **c**: Southern Plateau trawl survey series (1989–1993, 2000–2005: Tuck et al. 2009).

2.2.2 Area Trawled Index

Information on commercial trawl effort that is highly resolved spatially also allows us to summarise habitat alteration divided by key ecosystem, an OECD core pressure indicator (OECD 2003). This is important because impacts of trawling will vary enormously depending on factors such as substrate type and depth, and biological factors such as species present (National Research Council 2002). The Benthic Optimised Marine Environment Classification (BOMEC, Leathwick et al. 2009) was developed specifically to identify New Zealand "bioregions" that can be considered to be ecologically distinct (Figure 3). BOMEC was developed by combining data on the benthic community (made up of 126 demersal fish species, and 7 groups of invertebrates: asteroids, bryozoa, foraminifera, octocorals, polychaetes, scleractinian corals, sponges), and environmental data, including sediment type using Generalised Dissimilarity Analysis (Leathwick et al. 2009). BOMEC is restricted to sampling depths (less than 3000 m, Leathwick et al. 2009), and provides delineation of 15 bioregions at the scale of the EEZ. The choice of 15 bioregions is essentially arbitrary, being chosen to provide a broad scale classification of the EEZ, and does not imply any level of statistical significance in differences between regions (Leathwick et al. 2009).

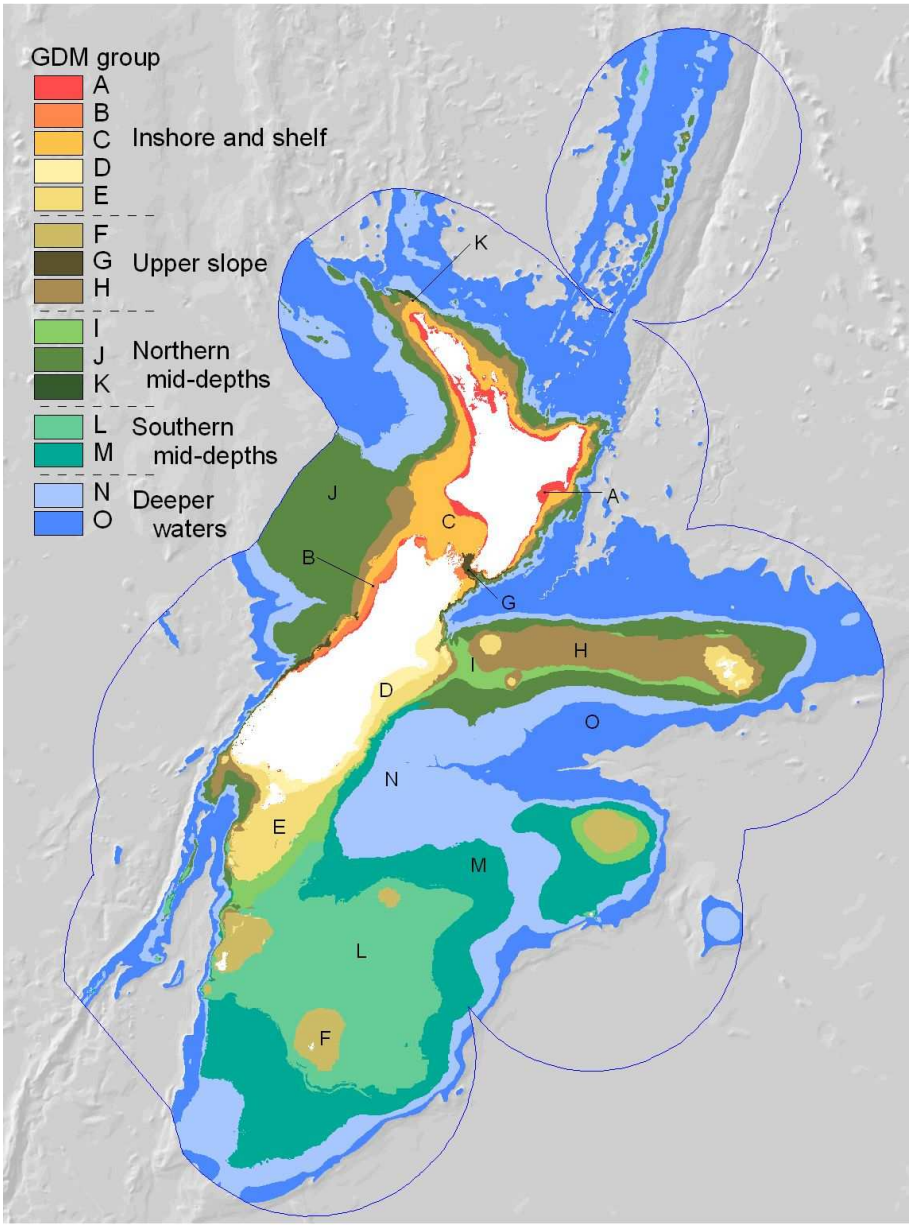

Figure 3. Benthic-optimised Marine Environment Classification for New Zealand (BOMEC, 15 groups; courtesy of Leathwick et al. 2009).

The overall pressure on the bentho-demersal ocean ecosystem of New Zealand can be measured as the Area Trawled Index (ATI), defined as the weighted average of the proportion of each BOMEC bioregion that is trawled in a given year (Equation 1).

Area Trawled Index (ATI)

$$ATI = \frac{\sum_{i=1}^{n}\left(\alpha_i \cdot \frac{A_i^{trawled}}{A_i}\right)}{\sum_{i=1}^{n}\alpha_i} \qquad [1]$$

Where $A_i^{trawled}$ is the area of bioregion i trawled in a given New Zealand fishing year which runs from 1 October to 30 September (henceforth "year"), A_i is the total area of bioregion i, and α_i is a weighting factor that accounts for some bioregions being more ecologically important that others. Although a case may be made for bioregions that support higher biomass or diversity of demersal fish and/or benthic invertebrates being more important ecologically than those with less known secondary production, the ecological basis for this is not well developed as yet, and a default position in the first

instance would be to assume that all bioregions are equally important ecologically and set α_i=1 for all i.

2.2.3 Biomass Trawled Index

Maps of the relative spatial abundance of 126 species of demersal fish in the New Zealand EEZ have been estimated by fitting a large database (nearly 17 000) catch records from research trawls to environmental characteristics including depth, temperature, bottom currents, and primary productivity using the multivariate method of Boosted Regression Trees (Leathwick et al. 2006a,b). Because it is based on trawl data, these predicted relative abundance maps are restricted to areas shallower than 1950 m (Leathwick et al 2006b) and it is not known to what extent these species occur at deeper depths.

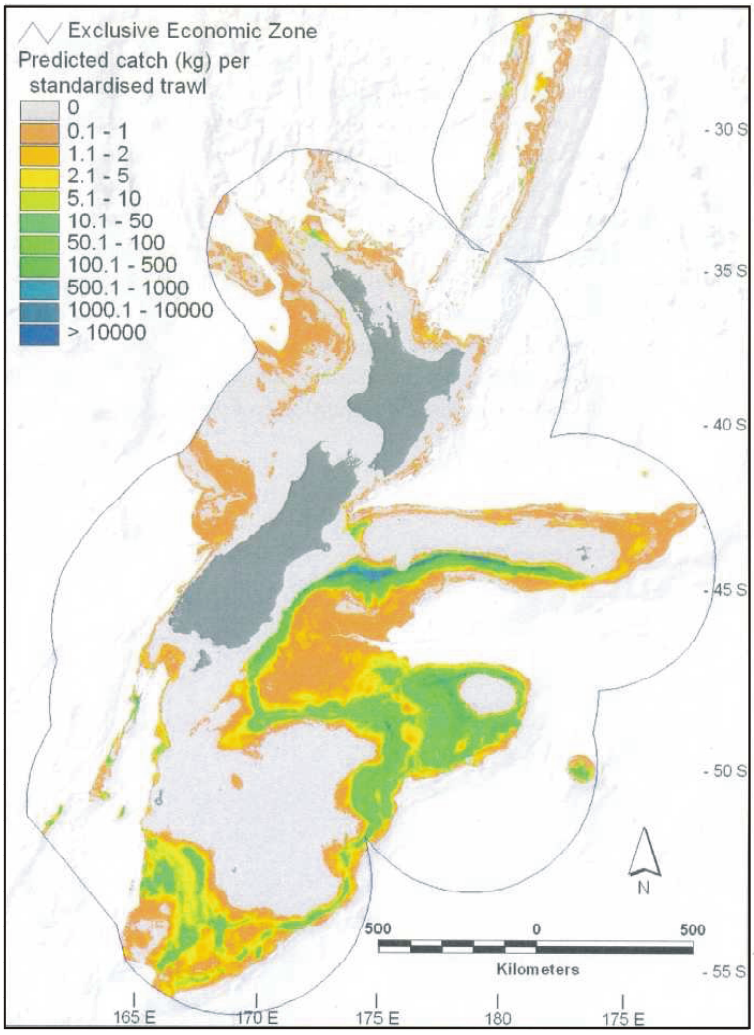

Figure 4. Example of predicted relative spatial distribution of black oreo dory (*Allocyttus niger*) for New Zealand EEZ (Leathwick et al 2006a).

An indicator of pressure due to fisheries trawling on the New Zealand demersal fish community could be calculated as a weighted average of the proportion of biomass of each fish species that is in an area which is trawled in a given year (Equation 2), called the "Biomass Trawled Index", BTI.

Biomass Trawled Index (BTI)

$$BTI = \frac{\sum_{i=1}^{n}\left(\beta_i \cdot \frac{B_i^{trawled}}{B_i}\right)}{\sum_{i=1}^{n}\beta_i} \qquad [2]$$

Where $B_i^{trawled}$ is the predicted biomass of demersal fish species *i* in a trawled area (Leathwick et al. 2006a), B_i is the total predicted biomass of species *i*, and β_i is a weighting factor that accounts for some species being more ecologically important that others. The relative ecological importance of species is not well known. Research to investigate whether it is possible to assign a relative ecological importance to species is urgent and important as it is needed to combine species-specific information into aggregate ecological indicators, both here and for other indicators described below.

Setting β_i=1 would assume all species were equally important to the overall functioning of the ecosystem. This is unlikely to be appropriate as more abundant and productive species are likely to be more important to the overall functioning of the ecosystem than less abundant and less productive species. However lower total abundance combined with biological and life history characteristics (such as lower productivity, later maturation, and fewer offspring) are likely to imply lower resilience. Trophodynamic connectivity and the particular topological characteristics of the food-web are also likely to affect the relative ecological importance of species to maintaining the structure and function of the food web. For example, it is well established that, in general, highly connected species are disproportionately important to the resilience of the food-web than more peripherally connected species (Albert et al. 2000; Dunne et al. 2002; Sole & Montoya 2001). In New Zealand, information on species connectivity is only beginning to become available following recent diet (Dunn et al. 2009), tracer (Nodder pers. com.), and food-web modelling studies (Pinkerton 2008). In summary, understanding the relative ecological importance of fishes in the New Zealand EEZ is important but poorly known at present. Here, we propose as an interim measure to assume that β_i varies monotonically with total secondary production of a given species, net of respiration. We suggest that secondary productivities should be 4^{th} root transformed to give greater importance to species with higher biomass and productivity, while also recognising the importance of species diversity within the fish assemblage (Figure 5).

It is then necessary to estimate total secondary production (i.e. the annual biomass increment in the absence of natural and fishing mortality) for all species of finfish in the New Zealand EEZ. Where quantitative data is available (e.g. from stock modelling) this should be used. This includes about 6 species (hoki, hake, ling, oreos, orange roughy, southern blue whiting). For the ~50 QMS species of finfish with no stock models, we suggest that Total Allowable Catch (TAC) values from the latest Ministry of Fisheries plenary report (Ministry of Fisheries 2009) could be used to provide preliminary "order-of-magnitude" estimates of secondary production. Under an MSY-management framework as is used in New Zealand (Mace 2001), yield and net secondary productivities of stocks are assumed to be closely related. We note that there are important reservations associated with this approach: (1) some TAC values are based on poor knowledge and are sometimes little better than educated guesses; (2) some TAC values are never reached indicating that they may overestimate,actual biomass or production of species; (3) there are political and management reasons why TACs might not correlated with species' secondary production; and (4) variations in natural mortality between species will alter the relationship between yield under MSY-based management and secondary production. Given such issues, alternative approaches to using TAC as a proxy for secondary production should be sought in time for state of the environment reporting in 2012. Nevertheless, provided that TACs are chosen carefully (for example, excluding near-zero TACs where stock rebuilding is taking place), TAC values are likely to provide a summary of the best available information of the relative productivities of most species within the QMS.

Given that TACs do not exist for non-QMS species, BTI could be calculated only for QMS species. Alternatively, and preferred since there are many non-QMS species and these are likely to be important for ecosystem function, alternative methods could be used to estimate secondary production of non-QMS species. This requires two pieces of information per species: biomass and productivity. The best available data to allow biomass to be estimated for non-QMS species in oceanic waters are the two scientific trawl surveys on the Chatham Rise and Southern Plateau. For many non-coastal species, these two surveys may encompass the majority of their distributions (Figure 2), and could be used to estimate order-of-magnitude biomass values as these surveys encompass the major distibutions of most species outside the coastal waters (Figure 2)biomasses if catchabilities can be assumed. Approximate productivities for these species could be estimated based on growth rates and other

biological information such as length-weight measurements available in the literature (e.g. Fishbase, Froese & Pauly 2000). Again, such approaches will be approximate. Because the non-QMS species are generally lower in biomass and productivity than QMS species, greater uncertainties in these data may not undermine the utility of BTI as an indicator of fishing pressure on the ecosystem. The sensitivity of these assumptions and data on BTI should be tested before this indicator is used.

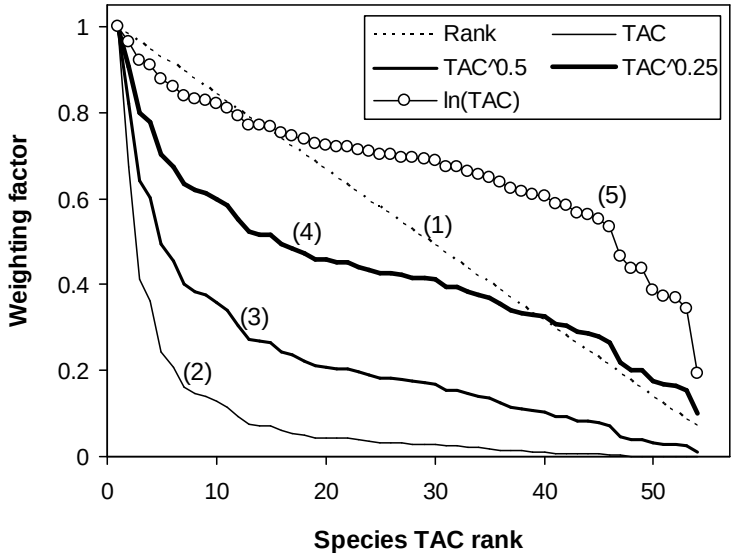

Figure 5. Weighting factors that could be used to combine information on species according to their ecological importance. Here, 57 fish and squid species caught in the New Zealand EEZ are ranked in order of their Total Allowable Catch (TAC) in the 07/08 fishing year (Ministry of Fisheries 2009) and five alternative weighting factors are shown: (1) inverse rank of TAC (=n+1-r_i where n is the total number of species and r_i is the rank of the ith species from largest to smallest TAC); (2): proportional to TAC; (3): proportional to the square root of TAC; (4): proportional to the 4th root of TAC; (5): proportional to the natural log of TAC.

2.2.4 Total fishing removals

Total fishing removal gives a clear indication of the pressure of fishing on the marine ecosystem system and is a core OECD indicator (OECD 2003). For the New Zealand ocean domain, the most appropriate indicator is likely to be total commercial catch by weight from fish caught both inside and outside the Quota Management System (MfE 2009a). Catch histories by fisheries sector are also available (MfE 2009b), divided as: (1) middle-depth species (hoki, hake, and ling); (2) deepwater trawl species (orange roughy, oreos, deep water Macrourids); (3) cephalopods; (4) mackerels; (5) small pelagics (southern blue whiting, pilchards, and mullets); (6) sharks, rays and skates; (7) marine invertebrates except cephalopods (scampi, oysters, and scallops); (8) highly migratory species (tunas, swordfish, and ray's bream); (9) species caught by bottom line species. Catch data disaggregated by group, or preferably species, can also be used to generate aggregated indicator of the level of removals such as the Marine Trophic Index (Section 2.3.12).

2.2.5 Fishery bycatch and discards

Although often used synonymously, bycatch refers to the mortality of non-target species by fishing, while discards are material not retained on board, including offal from target and non-target species. Bycatch and discard rates (as a proportion of total catch) have been used in parallel with measurements of the landings of target species as indicators of the pressure on the marine ecosystem due to fishing (e.g. FAO 1995, 2003; Zhang et al. 2009; Pitcher et al. 2008). An indicator based on bycatch is especially relevant to measuring the pressure on ecosystems due to fishing in regions where controlling gear type is an important part of managing the ecosystem impacts of the fishery, for example in European waters (European Union Common Fisheries Policy 2002). Although New Zealand relies more on the QMS than gear type to manage fisheries mortality, the proportion of bycatch to landed catch is still relevant here as impacts on non-QMS species are poorly recorded (if at all) in New Zealand and fishing may have important implications for the ecological viability of these

species. As all catches (QMS and non-QMS) species are included in the total fishing removals data used presently in New Zealand state of the environment reporting (Section 2.2.4), an indicator showing the catch of non-QMS vs QMS species is effectively an indicator of the ability of the Zealand management system to adequately manage fisheries mortality. This is hence a *response* indicator and is discussed in Section 2.4.3.

The rationale behind assuming improved sustainability due to lower discards rather than lower bycatch is less clear. In New Zealand, retaining caught material onboard may actually lead to greater impacts of fishing on the ecosystem than discarding some or all bycatch at sea, as retention would prevent scavengers feeding on discarded material. For example, the major food of ling (*Genypterus blacodes*) on the Chatham Rise (New Zealand) was recently found to be heads and tails of jack mackerel (*Trachurus* spp), which were considered to have been discarded by commercial fishing vessels (Dunn et al. 2009). Although discarding will change food-web structure, it is not clear that the effect will be more adverse than retaining all material, and we do not consider an indicator based on discards to be useful at this time.

2.2.6 Fisheries Pressure Index

Net primary productivity (NPP) is the amount of organic matter produced by the growth of phytoplankton after accounting for their respiration. NPP is fundamental to the functioning of marine ecosystems as it represents the energy entering the base of marine food webs and sets the carrying capacity of marine ecosystems (e.g. Ware & Thomson, 2005; Murphy et al. 2001) and imposes a fundamental upper limit on fisheries removals. The proportion of total NPP needed to support fisheries removals can be estimated as the Fisheries Pressure Index (FPI), Equation 3 (Knight & Jiang 2009).

Fisheries Pressure Index (FPI)

$$FPI = \frac{\sum_{i=1}^{n} \left[W_i \cdot Y_i \cdot \varepsilon^{1-TL_i} \right]}{NPP} \qquad [3]$$

Where, W is the wet-weight to carbon factor for fisheries landings (often taken as c. 0.1 gC gWW^{-1} for fish and squid, Vinogradov 1953), ε is the mean net transfer efficiency between trophic levels (see below), Y_i is the wet-weight catch of species i in the year of interest (gWW y^{-1}) and TL is the trophic level of species i, and NPP is the annual net primary productivity (gC y^{-1}). There are n species in the catch. A FPI>1 suggests that there is not enough primary production to support fisheries removals. The difference between FPI and 1 represents the amount of primary production available to support marine predators such as seabirds and marine mammals.

Two problems arise with implementing FPI. First, annual primary productivity rates are imperfectly known, and second, FPI is very sensitive to changes in the net transfer efficiency parameter, ε. The high spatial and temporal variability of NPP means that ship-based sampling cannot adequately observe carrying capacity at basin scales, and, instead, remotely-sensed data from Earth-observing satellite sensors are typically used to estimate NPP. Many alternative NPP models are available (e.g., Antoine & Morel, 1996a,b; Behrenfeld & Falkowski, 1997a, b; Westberry et al. 2008), and there are significant quantitative differences (>factor of 2) between these methods (Campbell et al. 2002). Estimating NPP from satellite data in the New Zealand EEZ is challenging because high-nitrate low chlorophyll (HNLC) conditions exist in New Zealand subantarctic waters off South Island (Boyd et al. 1999; Murphy et al. 2001) and satellite methods perform least well in this type of water (Campbell et al. 2002; Carr et al. 2006). Variations between three leading approaches to estimating NPP from satellite data in the New Zealand EEZ are of the order of ±20% (Pinkerton 2009), and it is not yet known whether these estimates bracket the true value (Schwarz et al. 2008).

More importantly probably, FPI is also very sensitive to changes in ε. The value of ε is often taken as 10% (Pauly & Christensen 1995; Knight & Jiang 2009), but values between 2–27% have been reported (Jarre-Teichmann et al. 1998; Christensen & Pauly 1993; Wolff 1994; Wolff et al. 1996; Pauly & Christensen 1995). For New Zealand offshore regions, Bradford-Grieve et al. (2003) gave an average value of ε of 23% from a trophic model of the Southern Plateau and data in Pinkerton (2008)

leads to an estimate of ε of 9% for the Chatham Rise. The mean trophic level of the New Zealand offshore catch is around 4.2 and that a change in ε between only 9% and 11% would change the required primary production by +42% and -27% respectively. In fact, ε is known much more poorly than this, so that uncertainties in FPI limit its usefulness.

The uncertainty in ε, and questions about its theoretical basis, also limit the utility of the "L-index" (Libralato et al. 2008), which aims to measure the potential ecosystem consequences of the loss of secondary production due to fishing. A meta-analysis of ecological models representing 51 exploited ecosystems allowed Libralato et al. (2008) to derive the relationship between the L-index and the probability of an ecosystem being sustainably fished. However, in contrast with comments in Libralato et al. (2008), this report argues that because the primary productivity that was ultimately required to support fishery removals is not measured directly, the L-index is also likely to be sensitive to ε, and consequently the usefulness of the L-index is likely to be compromised.

2.2.7 Fishing in Balance Indicator

The Fishing in Balance Indicator (FIB) (Pauly et al. 2000; Christensen 2000) is similar to FPI but has much lower sensitivity to uncertainties in ε by measuring the change in the proportion of primary production needed to support fisheries removals relative to a reference year. Higher values of FIB hence imply greater pressure on the ecosystem. FIB is often defined as Equation 4 (e.g. Christensen 2000), but in fact this is an approximation to the estimate of primary production needed to sustain fisheries because a number of different species at different trophic levels are involved and their requirements in terms of primary productivity do not scale according to the mean trophic level. A more rigorous but more data intensive definition of FIB is Equation 5.

Fishing in Balance (FIB)

$$FIB = \log_{10}\left[\frac{\varepsilon^{-MTI(y)} \cdot \sum_{i=1}^{n} Y_i(y)}{\varepsilon^{-MTI(y_0)} \cdot \sum_{i=1}^{n} Y_i(y_0)}\right] \quad [4]$$

Corrected Fishing in Balance (cFIB)

$$cFIB(y) = \log_{10}\left\{\frac{\sum_{i=1}^{n}\left[W_i \cdot Y_i(y) \cdot \varepsilon^{-TL_i(y)}\right]}{\sum_{i=1}^{n}\left[W_i \cdot Y_i(y_0) \cdot \varepsilon^{-TL_i(y_0)}\right]}\right\} \quad [5]$$

Where y is the year of interest and y_0 is a reference year (usually assumed to be the start of the time series for which data exists), MTI is the mean trophic level of the catch (see also Section 2.3.12), and other symbols are as Equation 3. Historical time series of fisheries catches (for QMS species at least) are available for the New Zealand EEZ (Section 2.2.4) and reasonable estimates of trophic levels for New Zealand fish are available (Appendix 1), so producing the FIB indicator for the New Zealand ocean is likely to be feasible. Because the FIB indicator is based on changes from a reference year, it is not sensitive to ε. Between 1990 and 2008 the New Zealand FIB changed between approximately -0.14 and 0.17 (data not shown). A range of ε between 9 and 23% would lead to changes in FIB of only about 0.01.

2.3 State/Impact indicators

2.3.1 Large-scale indicators of climate state

The state of the New Zealand climate has important effects on marine ecosystems, and many indicators of climate (and oceanographic) state of the New Zealand EEZ have been brought together to inform fisheries management (Dunn et al. 2007; Hurst et al. 2008). Key climate indicators of potential relevance for monitoring pressure on the marine ecosystem include Kidson regimes (Kidson 2000), Trenberth pressure indices (Trenberth 1976), and the Southern Oscillation Index (SOI). The SOI is the

normalized mean sea surface pressure difference between Tahiti and Darwin (Australia) and is related to the strength of the trade winds in the Southern Hemisphere tropical Pacific (Mullan 1995). SOI is an indicator of El Niño-La Niña oscillation, is correlated with rainfall, wind and temperature in New Zealand, and may be linked to recruitment strength in some demersal fish species (Dunn et al. 2007). Kidson regimes (Kidson 2000) relate to the occurrence of 12 different characteristic types of weather pattern over New Zealand. Trenberth indices (Trenberth 1976) are the difference in mean sea level pressure between pairs of New Zealand weather stations, from which time series of zonal and meridional winds can be estimated, starting in 1973. It should be possible to combine these climate indices into a single, multivariate climate indicator, with which to assess change in climate state for state of the environment reporting.

2.3.2 In situ monitoring of oceanographic state

Long time series of oceanic observations in the New Zealand are sparse, but notably include the expendable bathythermograph series across the Tasman Sea (Sutton et al. 2005), monitoring of SST at 8 mainland New Zealand coastal sites some from 1977, a network of sub-surface drifters (Argo: Roemmich & Gilson 2009), and bimonthly monitoring of ocean acidity along a transect off the Otago shelf (Kim Currie, pers. com.). These, and other in situ data potentially applicable for monitoring the state of New Zealand ocean will be summarised in work under the New Zealand Biodiversity Marine Environmental Monitoring Programme (MEMP, Livingston 2009). At present however, we cannot assess the potential utility of these data for state of the environment reporting.

2.3.3 Satellite ocean observations

Earth-observing satellite measurements include sea-surface temperature (SST, Uddstrom & Oien 1999), ocean colour (OC, Murphy et al. 2001; Pinkerton et al. 2005), and sea-surface height (SSH, Laing et al. 1998). Relatively long time series of consistent information are now available from many of these remote observations: >36 years for SST (1973–present), >12 years for OC (1997–present), and >17 years for SSH (1992–present). Statistical techniques such as rotated empirical orthogonality function analysis (EOF) and principal components have become standard methods for the extraction of characteristic spatio-temporal patterns from such time-series of meteorological and oceanographic measurements (Preisendorfer 1988; Emery & Thomson 1997). EOF analysis of satellite OC data over the north-east New Zealand shelf (Richardson et al. 2002; Kennan & Pinkerton 2008) has been completed, and the analysis is being extended to the EEZ-scale (Kennan, pers. com.). As has been carried out elsewhere (e.g. Polovina & Howell 2005), combined and/or separate EOF analyses of these satellite datasets should be used to provide an oceanographic baseline against which to develop an index of oceanographic change in the New Zealand EEZ, and also potentially acting as an indicator of climate-driven regime shift (Brierley & Kingsford 2009).

2.3.4 Phytoplankton and primary production

As noted in Section 2.3.3, more than 12 years (1997–present) of satellite measurements of ocean colour are available and are routinely used to estimate chlorophyll concentration (*chl-a*) as a proxy for phytoplankton biomass in the New Zealand EEZ (Murphy et al. 2001), with some validation (Pinkerton et al. 2005). *Chl-a* sets a fundamental limit on the carrying capacity of ocean ecosystems. Reporting overall trends in *chl-a* in the New Zealand EEZ, and summarising changes in the characteristic spatial and temporal patterns of *chl-a* are likely be useful indicators of the state of the foundation of the oceanic food web. As noted in Section 2.2.6, methods to estimate net primary productivity (NPP) in the New Zealand EEZ are available, but there are considerable differences between methods, and none has yet been validated. The most promising candidate NPP model (Behrenfeld & Falkowski 1997b), has been used to investigate variability and trends in NPP over the New Zealand EEZ (Pinkerton 2007), and this approach may succeed the use of *chl-a* as a proxy for ecosystem carrying capacity in the future.

2.3.5 Middle trophic level indicators

Middle trophic level organisms in the New Zealand ocean include pelagic crustaceans like copepods, shrimps and prawns, gelatinous zooplankton (jellyfish, salps), larval and juvenile fish (ichthyoplankton), cephalopods (squid and octopus), and small pelagic fishes, especially more than 21

species of myctophids (McClatchie et al. 2005; O'Driscoll et al. 2009a; Hall et al. 2008). The key role of these middle-trophic level species in ocean ecology is well known (e.g. Banse 1995; Marine Zooplankton Colloquium 2, 2001; Smetacek et al. 2004), and they form the basis of the diet of many commercially-important New Zealand fish species (Dunn et al. 2009). These species are likely to be affected both by fishing reducing top-down predation control, and by climate-driven changes in lower trophic food-web components (Frank et al. 2007; Richardson 2008).

Few data exist for middle trophic level organisms in the New Zealand ocean and we cannot as yet monitor their state. New Zealand acquired a Continuous Plankton Recorder (CPR) in 2008 and this has been deployed twice to date as a start of a time series of zooplankton monitoring over the Chatham Rise. In other parts of the world, long time-series of measurements of the zooplankton community by the Continuous Plankton Recorder (CPR) has demonstrated regime shifts (Beaugrand et al. 2002; Aebischer et al. 1990), and been recommended as an effective way of monitoring the state of pelagic ecosystems (Beaugrand 2005). In due course these data could provide an indicator of change in the zooplankton community in the New Zealand ocean. Research is also underway to investigate whether multifrequency acoustic backscatter data taken from research vessels during the annual surveys of fish on the Chatham Rise and Southern Plateau (Figure 2) can be used to derive indices of abundance of mesopelagic fish in these regions (O'Driscoll et al. 2009b), but results are not available at present.

2.3.6 Fish stocks: absolute biomass

The proportion by which the biomasses of predatory fishes have been reduced by fishing is a clear indicator of the state of the marine ecosystem. An indicator showing the total estimated biomass of fish in the New Zealand EEZ would be a valuable indicator of the state of the system, but the efficacy of this may be limited by availability of information. Quantitative estimates of the spawning stock biomass only exist for a small number of the most important New Zealand species (Ministry of Fisheries 2009). Stocks assessed using quantitative models include: hoki (*Macruronus novaezelandiae*), hake (*Merluccius australis*), ling (*Genypterus blacodes*), oreo (*Allocyttus niger*, *Neocyttus rhomboidalis*, *Pseudocyttus maculatus*), southern blue whiting (*Micromesistius australis*) and (to some extent) orange roughy (*Hoplostethus atlanticus*). Together, these species make up almost half (46%) of the total allowable catch of finfish in the New Zealand EEZ (Ministry of Fisheries 2009), and may make up the majority of the demersal fish biomass in some regions (e.g. 75% over the Chatham Rise, Pinkerton 2008). Quantitative stock assessments are attempted for these species each year and significant effort has been made to determine their catch histories since industrialised fishing began. Consequently, the absolute change in biomass that has occurred since fishing began for key selected species of fish could be reported to indicate of ecosystem state.

2.3.7 Fish stocks vs management targets

An indicator that summarises current stock state against management targets will be useful, as this would take into account the significance of a given reduction in biomass against the particular characteristics of a species. Each of the species within the New Zealand QMS system is divided into between 1 and 10 stocks for management purposes, with 629 stocks in the New Zealand QMS in 2009 (Ministry of Fisheries 2009). As noted previously (Section 2.1), fish stocks in the New Zealand QMS are managed according to a version of maximum sustainable yield (Mace 2001, New Zealand Fisheries Act 1992), which, from early 2009, has been implemented according to the Ministry of Fisheries Harvest Strategy Standard (HSS, Ministry of Fisheries 2008). The HSS are augmented by additional conservation measures as required on a stock-by-stock basis. The HSS promises to deliver welcome clarity to reporting stock levels again management targets. It comprises a target stock level (B_{target}) for each stock and two lower stock levels (B_{soft} and B_{hard}) which indicate levels of over-depletion and require different management action. For example, for hoki, one of New Zealand's biggest fisheries, B_{MSY}=25%B_0, B_{target}=35-50%B_0, B_{soft}=20%B_0, and B_{hard}=10%B_0. However, to date, the status of only a small minority of species have been reported according to HSS target or reference levels (22 from 629: Ministry of Fisheries 2009), with more added "each time they are reviewed, as time allows". In the meantime, stock status of only a small subset of QMS stocks (15–21% of stocks: Ministry of Fisheries website) are reported as "Near or above target levels", "Depleted (overfished)" or "Collapsed". A point-based system is proposed here to score the state of fish stocks relative to management targets (Table 2), and these can be combined to give the Stock Status Index as Equation

6.

Table 2. Scoring table for reporting state of fish stocks against reference points.

Stock Status (SS)	Description	Current stock level	
		No target or reference points exist; no quantitative stock assessment	Target and reference points exist; quantitative stock assessment[1]
1	Above target levels	Probably near or above target levels	$B>B_{target}$
0.75	Overfished		$B_{target}>B>B_{soft}$
0.5		Depleted (overfished)	
0.25	Severely overfished		$B_{soft}>B>B_{hard}$
0	Collapsed	Collapsed	$B<B_{hard}$

[1] Only 22 stocks from a total of 629 were reported in this way in 2009, but together these make up 46% of the total TAC of finfish in the New Zealand EEZ (Ministry of Fisheries 2009).

Stock Status Index (SSI)

$$SSI = \frac{\sum_{i=1}^{n}(\beta_i \cdot SS_i)}{\sum_{i=1}^{n}\beta_i} \qquad [6]$$

Where SS_i is the stock status score of stock i from Table 2, and β_i is relative ecological importance of stock i (as Equation 2). The 4th root of TAC could be used as a preliminary proxy for β_i as suggested in Section 2.2.3.

2.3.8 System Catch-per-Unit-Effort

Catch-per-unit-effort (CPUE) is commonly used investigate whether fishing is making fish harder to catch which may reflect changes in total abundance (Hilborn & Walters 2003). CPUE is an imperfect indicator of fish abundance for many reasons concerned with fish and fishers behaviours (Harley et al. 2001; Clark 2006) but is nevertheless still widely used in New Zealand and elsewhere in fisheries management (Ministry of Fisheries 2009; Hilborn & Walters 2003) and a multi-species CPUE index has been suggested as a useful indicator of ecological state (Fulton et al. 2005). Grouping all fish species caught by a reasonably consistent method (bottom trawling) in a given sub-area would allow a "system-level" CPUE (sCPUE) to be calculated for the New Zealand EEZ (Equation 7).

System-level CPUE in region j

$$sCPUE_j = \frac{\sum_{i=1}^{n} Y_i^j}{D_j} \qquad [7]$$

Where Y_i^j is the catch of species i in region j in a given year, and D_j is the area trawled in the same region and year. Only commercial species (those currently in the QMS) would be included in this indicator because catches of other species are not reliably recorded (Gilbert et al. 2000). The sCPUE index could be standardised in the normal way used for CPUE investigations, to account for variations in (for example) gear-type and the spatial and seasonal distribution of fishing effort between years, (e.g. Campbell 2004). It remains to be seen to what extent it is possible to correct sCPUE calculated in this way for major changes in fishing practice that have occurred in the New Zealand EEZ since 1989, such as the introduction of twin trawls or changes to the mesh type commonly used in trawling. It is also not known at present whether normalisation of sCPUE could compensate for changes not connected with the state of the marine environment, for example, due to changes in fishing effort in response to changing market demand.

Alternatively, sCPUE could be calculated and standardised for small subareas separately and combined into an EEZ mean index according to an estimate of their relative ecological importance (Equation 8). The most appropriate spatial resolution for this calculation may be the 120 New Zealand

Fishery Statistical Areas that span the New Zealand EEZ because this is the spatial resolution at which the location of commercial catches are reported (Ministry of Fisheries 2009). Alternatively, a higher number of subregions based on BOMEC (Leathwick et al. 2009) could be used, provided that catch locations could be allocated spatially using, for example, tow positions derived from TCEPR records as has been done previously (Baird, pers. com.).

System-level CPUE

$$sCPUE = \frac{\sum_{j=1}^{m} \left(\gamma_j \cdot sCPUE_j \right)}{\sum_{j=1}^{m} \gamma_i} \qquad [8]$$

Where there are m subareas and γ_j is a weighting factor that accounts for some subareas being more ecologically important than others. The relative ecological importance of subareas is not known and setting γ_j=1 would assume all subareas were equally important. It is suggested here that as this index specifically measures the state of the demersal fish community, subareas which support higher biomasses of fish should be accorded greater importance than others, and an appropriate weighting factor could be given by Equation 9.

Subarea weighting factor

$$\gamma_j = \sum_{i=1}^{n} \left(TAC_i^{\,0.25} \cdot \frac{B_i^j}{B_i} \right) \qquad [9]$$

Where TAC_i is the maximum constant yield of species i, B_i^j is the relative biomass of species i in subarea j (Leathwick et al. 2006a, b), B_i is the total predicted biomass of species i (Leathwick et al. 2006a, b), and n is the number of species for which TAC values are available (from Ministry of Fisheries 2009). This is equivalent to assigning an ecological importance to each subarea weighted by a function of the fraction of total production of demersal finfish species in the QMS it supports. The quarter power transform on TAC was suggested in Section 2.2.1. Again, the ability of this approach to standardise sCPUE for changes in gear type, fishing practice, or the distribution of effort due to market factors should to be investigated.

2.3.9 Relative Price Index

It has been suggested that changes in the price of seafood could provide information on the state of the marine ecosystem (Pinnegar et al. 2002, 2006). Existing evidence elsewhere in the world suggests that the average market price of a species will increase as it becomes scarce (OECD 1997; Sumaila 1998). Generally, large high trophic level species command higher prices at market than do small low trophic level fishes (Pinnegar et al. 2003). As fishing tends to impact large, high trophic level species more than smaller, low trophic level fishes, the Log Relative Price Index (LRPI) has been proposed which is equal to the slope of the regression line of log price (in NZ$ kg^{-1} for example) against trophic level of the fish or shellfish species (Pinnegar et al. 2006). Increasing scarcity of higher trophic level species due to fishing may be expected to increase their price relative to lower trophic level species (at least at first) and increase the LRPI (Pinnegar et al. 2006). However, it is known that price differs markedly between fish species reflecting supply and/or desirability of the product, irrespective of their abundance or ecological state. A preliminary analysis of New Zealand 2009 deemed values (the Government determined value of QMS species for quota management purposes) and trophic levels from 175 New Zealand stocks of finfish only showed a very weak relationship (R^2=0.058, data not shown) which does not support the utility of this index. A re-analysis using market, rather than deemed, values of seafood may be useful, but is considered low priority.

2.3.10 Ecosystem variability

Increasing variability, greater asymmetry in perturbations in ecosystem properties, or slower recovery from perturbations (“reddening” of the power spectrum) are possible consequences of exploitation (Brock & Carpenter 2006). These approaches may be useful to indicate chronic erosion of ecosystem

resilience, and increasing potential for abrupt and persistent ecosystem change (Carpenter & Brock 2006; Guttal & Jayaprakash 2008; van Nes & Scheffer 2007). Changes to variance can occur in many aspects of ecosystem state, but is most likely to be most strongly manifested close (in an ecological sense) to where the perturbation occurs, especially in fish stocks themselves (Hsieh et al. 2006). In New Zealand, the longest time series' of ecosystem properties are the abundances of major commercial fish stocks and the recruitment year-class strength (YCS). Dunn et al. (2007) collated YCS and biomass indices for 56 New Zealand fish species derived from stock assessment models and three series of research trawl surveys (Hauraki Gulf, Chatham Rise, Southern Plateau). The lengths of these time series ranged from 5 to 31 years (1975–2006), but most are relatively short (<20 years). Indices of climate state were also assembled for these periods (see Sections 2.3.1–2.3.4). The focus of the analysis by Dunn et al. (2007) was to identify possible linkages between climate and YCS. It may be useful to use these data to investigate whether changes in the characteristics of variability are present in these assembled time series of YCS, biomass, and climate.

2.3.11 Fish-based trophodynamic indicators

Reducing the abundance of fish species by commercial exploitation can affect overall marine trophic structure in diverse and unpredictable ways (Cury et al. 2005; Fulton et al. 2005). In extreme cases, fishing can cause readjustment of the food-web through its effect on trophic relationships and increase the likelihood of trophic cascades (Pace et al. 1999; Casini 2008; Daskalov 2007). There are also potential non-trophic indirect effects where changes to piscivore abundance affect the behaviour of other species (Preisser et al. 2005). Ecological indicators based on network analysis of food-webs tended to require more data and be more sensitive to uncertainty in these data than simpler indices based on biomasses of particular types of species present (Fulton et al. 2005). Trophodynamic indicators are here grouped into the following sections: (1) mean trophic level of the catch (MTI, Section 2.3.12); (2) species-based indicators (Section 2.3.13); (3) functional group-based indicators (Section 2.3.15); (4) size-based indicators (Section 2.3.16). There are many more data-intensive approaches to testing for change in emergent properties of marine ecosystems, including the trophic spectra of catches (Gascuel et al. 2005), tests for changes in diet or trophic position of particular fish species, or emergent ecosystem properties such as Overhead, Ascendency, Capacity, Finn's cycling index (Fulton et al. 2005, Shannon et al. 2009) or thermodynamic indicators such as exergy and emergy (Jørgenson 2006). These methods cannot be carried out using data and ecosystem models currently available in New Zealand. Progress towards developing ecosystem models under the FRST Coasts and Oceans OBI may enable model-based indicators of ecosystem structure be used in the future.

2.3.12 Marine Trophic Index

The Marine Trophic Index (MTI) is the mean trophic level of fisheries landings (Pauly & Watson 2005; Equation 10) and was recently recommended for use with commercial catch data by the United Nations Biodiversity Convention as a widely-applicable and cost-effective indicator for monitoring progress towards reducing biodiversity loss in marine ecosystems (CBD 2004).

Marine Trophic Index (MTI)

$$MTI = \frac{\sum_{i=1}^{n} \left(TL_i \cdot Y_i \right)}{\sum_{i=1}^{n} Y_i} \qquad [10]$$

Where Y_i is the total landings of species (or stock or group of species) i in a given year, TL_i is the trophic level of i, and n is the number of species. A gradual decline in trophic level of c. 0.2 since industrialised fishing began has been observed in many finfish fisheries around the world (Pauly et al. 1998a; Christensen et al. 2003), ascribed to fisheries targeting high trophic level species and moving on to lower trophic level species as these large species are depleted, a change called "fishing down the food web". Essington et al. (2006) noted that "fishing through the food web", where higher trophic level fish landings are maintained, but catch of lower trophic level species increases over time.

Historical time series of fisheries catches are available for the New Zealand EEZ (Section 2.2.4), and reasonable estimates of trophic levels for New Zealand fish are available (Appendix 1) so this could be carried out fairly straightforwardly. Indeed, MTI for the New Zealand region based on FAO returns is currently available online (Sea Around Us, 2009). Basing MTI on commercial catch returns, as is almost exclusively done internationally, means MTI will vary with changes in which species are targeted by fisheries between years, how much of the catch is reported, the quality of identification of species, and for other reasons not necessarily associated with effects of fishing (Caddy et al. 1998; Pauly et al. 1998b; Tuck et al. 2009). It seems sensible therefore to calculate MTI based on finfish landings taken only by one type of fishing, with bottom trawling being the predominant fishing method in the New Zealand ocean. This would automatically exclude low trophic level species (<3.25) as recommended by Pauly & Watson (2005) to reduce bias. The quality of identification of species and the quality of catch reporting will prevent MTI for New Zealand being valid earlier than a certain date, perhaps 1989 when reporting requirements for large bottom trawling vessels was improved.

In addition to commercial data, two series of research trawl surveys have been carried out (Chatham Rise, Southern Plateau) and could be used to derive fishery-independent MTI (Tuck et al. 2009). MTI derived from commercial trawl data and the two research surveys together can provide an "envelope" of changes in the Mean Trophic Index (Figure 6). This combined index includes the advantages of using commercial trawl landings data to provide complete coverage of the EEZ for all years, coupled with the consistency of the research survey data for the two main offshore fishing areas of New Zealand. Differences between MTI trends in research trawls and commercial trawl data is likely to reflect changes in fishing behaviour and trawl fishing gear between 1990 and 2008. Preliminary data (Figure 6) are consistent with results from other areas of the world showing that changes in MTI based on commercial catch data tend to underestimate corresponding changes in the ecosystem (Pauly & Watson 2005; Pinnegar et al. 2002; Christensen 1998).

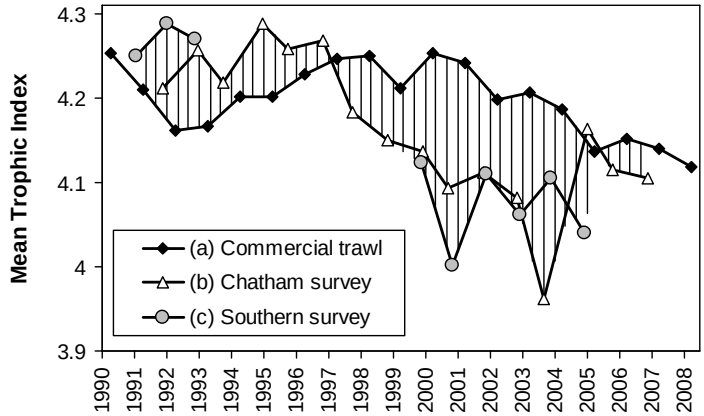

Figure 6. Marine Trophic Index (MTI) for bottom trawling in the New Zealand oceanic EEZ. (a) All commercial trawl catch data taken outside territorial limit; (b) Research trawl data from the Chatham Rise trawl survey (Tuck et al. 2009); (c) Research trawl data from the Southern Plateau trawl survey (Tuck et al. 2009). The "envelope" of MTI data is shown vertically hatched.

2.3.13 Diversity in fish communities

A number of diversity measures have been applied to fish communities. Fishing affects the relative abundances of species in communities because of differences in biology and ecological life-histories between species. Population sizes of longer-lived and later-maturing species will tend to be depleted more by a given fishing mortality than smaller, faster-growing and more fecund species which have higher intrinsic rates of population increase. Fishing a mixed assemblage will hence tend to lead to a change in diversity of species over time, measured alternatively or jointly by how many species are present (richness) and how similar their abundances are (evenness/dominance). Very many measures are available to measure diversity, giving different emphasis to richness versus evenness/dominance (Greenstreet & Rogers 2006). For example, fishing rarely causes large-scale extirpation so measures of total richness are likely to be less sensitive than measures of evenness to the effects of fishing.

Different measures of ecosystem evenness respond variously to fishing, increasing, reducing or being unaffected depending on the initial characteristics of the ecosystem. A community initially dominated by k-selected species would be expected to become more even and show increasing diversity metrics due to fishing, whereas diversity metrics may be expected to decrease after fishing if the ecosystem were originally dominated by r-selected species. Tuck et al. (2009) calculated 8 diversity metrics based on data from the New Zealand Chatham Rise and Southern Plateau series of trawl surveys (Figure 2): Hill's N1 and N2 indices, total species richness, Margalef's D, Pielou's evenness index, Shannon-Wiener index, and the mean and variance of taxonomic distinctness. Overall, there was quite good correspondence between changes to diversity identified by the different metrics, with diversity tending to increase over time for both surveys, consistent with initial dominance by larger, slower growing species (Tuck et al. 2009). Here we define the Demersal fish Diversity Index (DDI) as how many of the 8 diversity metrics significantly changed in the same direction minus the number that significantly changed in the opposite direction, and suggest that this provides a reasonable indication of change for each subarea (Gilbert et al. 2000). Indices could be combined across the 12–20 subareas used by Tuck et al. (2009) by averaging, assuming (for example) that each area is equally ecologically important. This approach could also be applied to commercial catches taken by bottom trawling in subareas visited frequently, though the analysis would then be limited to species within the QMS as only catches of these species are reported. Also, changes in the quality of identification of species over time would need to be corrected for by grouping species, as was carried out by Tuck et al. (2009). Grouping catches spatially would be needed to account for spatial changes in fishing effort between years, and standardisation using New Zealand subareas may be appropriate.

Abundance Biomass Comparison curves (ABC: Warwick 1986, Yemane et al. 2005) have been used to indicate disturbance by examining differences in the relationship between cumulative biomass and species ranked by abundance. Applying this method to research trawl survey data from two New Zealand regions (Chatham Rise, Southern Plateau) was not promising (Tuck et al. 2009), mainly because numbers and weights of only a small number of species of fish (generally only QMS species) were recorded concurrently (Tuck, pers. com.). Concurrent reporting of numbers and weights of fish in the catch by species is not carried out from commercial fishing vessels at all. Given the lack of suitable data at present it is unlikely that this method will have much power to show changes at the scale of the EEZ.

It would be useful to obtain information on the abundance and distribution of benthic invertebrates at the scale of the New Zealand EEZ using commercial fishing. This would provide data that could be used to produce an index of benthic biodiversity to monitor changes in benthic communities over time. At present, reporting of invertebrate bycatch in commercial fisheries is of very variable quality, and generally extremely rudimentary, so that our ability to observe change in this part of the ecosystem is exteremely limited.

2.3.14 Spatial distributions of fish species

Disturbance by, amongst other factors, fishing and climate change can cause changes to the geographic distribution of fish species. The percentage area of a research survey in which most (typically 90%) of the population occurs has been used as an ecosystem indicator (e.g. Fisher & Frank 2004). Tuck et al. (2009) showed that statistically significant changes to the range of a number of QMS species were evident in research trawl survey data from the Chatham Rise and Southern Plateau, with both increasing and decreasing ranges detected. Similar analysis based on commercial catch data is likely to be feasible, and may provide information on changes to the geographic range of species. These changes in spatial ranges of QMS species could then be combined into a single indicator on the state of the New Zealand marine ecosystem, for example, by simply counting the number of QMS species showing a statistically significant decrease in range over time (Gilbert et al. 2000).

2.3.15 Functional group-based fisheries indicators

Changes to the relative abundance of different functional groups in an ecosystem can be used to investigate whether the food-web is changing over time (Fulton et al. 2005; Methratta & Link 2007; Shannon et al. 2009). Functional groups can be based on various descriptors of ecological niche, such as position in the water column (e.g. pelagic, demersal, benthic), trophic guild / feeding type (e.g.

piscivore, pelagic invertebrate feeder, benthic feeder, scavenger), taxonomy (e.g. elasmobranch, gadoid, macrourid), or a combination of multiple ecological and life-history traits (Methratta & Link 2007) which can be combined to suggest high or low resilience (Tuck et al. 2009). A simple and commonly used index is the proportion of piscivorous fish to all fish caught. As piscivorous fish tend to be disproportionately impacted by fishing (Caddy & Garibaldi 2000), their relative abundance in fish assemblages is a measure of ecosystem state. Applying a number of functional group-based methods to data from the time series of research trawls on the Chatham Rise and Southern Plateau, Tuck et al. (2009) found that the piscivorous to total fish catch ratio was a promising indicator of change in the fish community over time. The view that piscivorous fish are likely to be impacted by fishing more severely than invertebrate feeders (Caddy & Garibaldi 2000) appears to be supported by the available data for these two important areas of New Zealand. The demersal to total fish catch ratio was less promising, with few significant trends over time identified. This is maybe not surprising as the presence of differential impacts of fishing on demersal relative to pelagic fish species is not clear. With improved data becoming available on feeding characteristics of many New Zealand fish species and the development of a preliminary guild structure (at least in one geographic area: Dunn et al. 2009), a more sophisticated functional group-based analysis could be repeated based on feeding characteristics or ecological-niche assemblage. As for species-based methods (Section 2.3.13), this approach could potentially be applied to commercial catches using data available at present. The analysis is likely to require data to be stratified by the method of fishing (bottom trawling only perhaps), by area to account for spatial changes in fishing effort between years, and by species (major middle-depths QMS species only). In addition, it is likely that species will need to be grouped to account for changes in the quality of identification of species over time (as Tuck et al 2009).

2.3.16 Size-based fisheries indicators

Size is a key structuring factor in marine ecosystems (Hildrew et al. 2007), and size-based analysis has been shown to be useful in indicating the degree to which fishing has perturbed the system (Jennings & Dulvy 2005; Shin et al. 2005). Fishing changes the size distribution of fish in mixed assemblage by many mechanisms: (1) fisheries targeting higher-value large fish species; (2) less escapement of larger individuals compared to smaller individuals due to gear characteristics; (3) fishing reducing the average age and size of fish of a given species because of increased total mortality; (4) greater reduction of stock size of species with larger maximum size because of lower intrinsic rates of population increase. Applying 10 size-based metrics to data from the New Zealand Chatham Rise and Southern Plateau trawl surveys gave an inconsistent picture of change (Tuck et al. 2009) and this approach does not seem as promising for the New Zealand situation as species-based approaches. Also, it is unlikely that size-based indicators could be extended for use with commercial catch data, as a statistically representative number of fish need to be measured for the application of this method, and this is not systematically carried out at present.

2.3.17 Evolutionary change in fished species

The potential for fishing to cause rapid, evolutionary change in fish species is now well established (Stokes & Law 2000; Stockwell et al. 2003; Swain et al. 2007). Identifying fishing-induced changes in fish, such as maturation occurring at lower age or size, is recommended as part of an effective, long-term fisheries management process (Kuparinen & Merilä 2007). Monitoring fishing induced evolutionary change in New Zealand fisheries is likely to be feasible only for well-monitored species as this requires large numbers of physiological measurements over extended periods (Kuparinen & Merilä 2007). It is recommended here that the existing time series information on the physiological characteristics of well studied New Zealand target species such as hoki, hake, ling, oreos, orange roughy, and southern blue whiting be examined to ascertain whether these can be used to detect fishing-induced evolutionary change in these species. It may also be possible to carry out similar monitoring for inshore species such as snapper (*Pagrus auratus*). In particular, changes to growth rates, age or size at maturity should probably be investigated first.

2.3.18 Top predators

Top predators can be used in two ways as indicators of the state of marine ecosystems. First, an OECD core indicator is the overall ecological threat status of species in the ecosystem, often an emphasis

placed on top predators (OECD 2003). Consistent with this approach, the ecological status of marine species divided as marine fish, marine invertebrates, marine mammals, macroalgae and seabirds is currently reported in the New Zealand state of the environment reporting (MfE 2007). New Zealand resident and endemic species are assessed using a threat classification system developed by the New Zealand Department of Conservation (DoC): Townsend et al. (2008); Molloy et al. (2002); Hitchmough et al. (2007). This system has 8 threat categories: (1) nationally critical; (2) nationally endangered; (3) nationally vulnerable; (4) declining; (5) recovering; (6) relict; (7) naturally uncommon; (8) not threatened. These DoC threat categories are broadly analogous to those of the International Union for Conservation of Nature and Natural Resources (IUCN) criteria and data (IUCN 2009), which is used to assess the status of visiting, migrant and vagrant species in New Zealand.

Second, particular ecological aspects of selected predator species can be used to indicate changes in ecosystems. For example, top predators are widely used in monitoring the ecosystem effects of fishing krill in the Southern Ocean (Reid et al. 2005; Constable 2006), with information on the breeding of penguins, albatross, petrels, and seals collected, summarised and considered in management annually (CEMP 2004; Agnew 1997). Monitoring top predators as "bellweathers" of ecosystem health is also increasingly used elsewhere (Boyd et al. 2006; Ainley 2002) as they are recognised as potentially useful downstream integrators of change in the marine ecosystem, exploit marine resources at similar spatial and temporal scales to humans, and receive high public interest. However, given that predators respond in complex ways to many factors simultaneously, ascertaining the appropriate management response to change of a predator-based indicator is difficult (Boyd et al. 2006). Multispecies indicators, and/or indicators based on multiple metapopulations can be used to create a composite predator index that can improve generality, but research cost is often prohibitive (Boyd et al. 2006). Some comments on the utility of predators for monitoring the state of the New Zealand ocean are given below for seabirds (Section 2.3.19), seals (Section 2.3.20), and cetaceans (Section 2.3.21).

2.3.19 Seabirds

The New Zealand region is especially rich in seabird taxa. Of the 360 species of birds that obtain all or nearly all of their food at sea (hereafter, "seabirds") recognised globally, 86 breed in the New Zealand region and 38 are endemic (Te Ara 2009). We note that this definition of seabird excludes shore feeding birds such as plovers, herons, dotterels, snipe etc. that may be impacted by changes in New Zealand coastal ecosystems but are less likely to be affected by changes in the New Zealand ocean. Taylor (2000) lists 18 major threats to seabirds breeding in the New Zealand EEZ. Many of these threats are not connected to the state of the ocean environment, including effects of introduced mammal or avian predation, lost of nesting habitat and human disturbance. However, at least 7 threats given by Taylor (2000) are potentially indicative of apects of the health of the marine ecosystem and general sustainability of human activities, including interactions between fisheries and seabirds (either by direct bycatch or through reduction in food availability), marine pollution, and global environmental change. In this way, seabirds can be valuable integrators of the state of the marine environment (Montevecchi & Myers 1996; Croxall 2006). Ecological status of New Zealand seabirds are assessed by IUCN (IUCN 2009) and by DoC (Hitchmough et al. 2007), with generally good agreement between the threat status by the two methods. Where they differ, the seabird threat status based on DoC analysis is preferred over the "red-list" information of IUCN as the fact that it is revised more regularly (every 3 years) and is based on better local knowledge is judged to offset the loss of international intercomparability (Townsend et al. 2007; MfE 2007).

In addition to reporting the overall threat status of New Zealand seabirds in terms of number of species in various threat categories, particular seabird species or parameters could be used for more specific ecological monitoring, though this is not done at present. Of particular interest for producing an seabird indicator with a more specific ecological meaning relevant to showing the state of the New Zealand marine ecosystem would be methods to monitor the status of adults, juveniles or breeding success of fish-foraging species that return regularly to colonies to breed, and for which historical data and some ecological understanding exists to help interpret the data (Einoder 2009). It is beyond the scope of this paper to recommend such seabird species or parameters for monitoring, but this may be a useful area for further investigation.

2.3.20 Seals

Three seals species are resident in New Zealand: the New Zealand sea lion (*Phocarctos hookeri*), New Zealand fur seal (*Arctocephalus forsteri*), and the Southern elephant seal (*Mirounga leonine*). The threat status of these are included in New Zealand state of the environment reporting (Hitchmough et al. 2007; MfE 2007). Information on the breeding success (e.g. annual pup production), total population size and/or breeding range of the New Zealand sea lion and New Zealand fur seal could be used as contrasting indicators of the sustainability of human activities in the marine environment. Before human colonisation, the New Zealand sea lion and fur seal were distributed along the coasts of New Zealand main islands but were severely reduced in number and range by commercial sealing during the late 1700s and early 1800s (Worthy 1994; Gill 1998). Despite being protected for over 100 years, the breeding range of the New Zealand sea lion has not recovered and almost all pups are born on the subantarctic Auckland and Campbell Islands, (Gales & Fletcher 1999; Chilvers et al. 2007). The New Zealand sea lion is classified as vulnerable (IUCN 2009) and declining (Hitchmough et al. 2007), with pup production falling by about 50% since 1998 (Chilvers et al. 2007; Meynier 2009). Interactions between the New Zealand sea lion and squid fisheries are well documented, and may be at least partially responsible for this continued decline, though this is disputed (Cawthorn et al. 1985; Meynier 2009; Gales 1995; Smith & Baird 2007; Wilkinson et al. 2003). In contrast, although population numbers of New Zealand fur seals are not well known, this species is now considered non-threatened and numbers are thought to be increasing (Taylor et al. 1995), with recolonisation of breeding sites around the coast of New Zealand South Island (Bradshaw et al. 2000; Lalas & Murphy 1998). Fur seals also interact with various fisheries, through bycatch (Smith & Baird 2009) and possibly competition for fish. There may be effects of fishing on local colonies, but fishing is seemingly having little overall effect on population status (Mattlin 1998). Southern elephant seals are classified as Nationally Critical (Hitchmough et al. 2007), with less than 300 animals living in the New Zealand EEZ, and breeding confined to subantarctic islands (Taylor & Taylor 1989). Little further infomation on their interactions with fishing or other human activities in the New Zealand region is available.

For specific information on aspects of the ecological status of the New Zealand sea lion or the New Zealand fur seal, such as breeding extent or population trends, to be used as an ecological indicators, it would be necessary to understand how the index was related to human activity or overall ecological state. This is unlikely to be available in the short or even medium term, but may be developed in the future.

2.3.21 Cetaceans

The New Zealand Department of Conservation (DoC) identify 26 species of cetaceans as resident in New Zealand waters, of which 5 are endangered (Hitchmough et al. 2007), and the status of these are included in New Zealand state of the environment reporting (MfE 2007). Whereas time series of abundances are potentially estimable for some populations of large whales in New Zealand waters (e.g. Southern Right whales: Patenaude 2003), very little is known about most smaller species of cetaceans, with no assessment possible for 14 species because of lack of data, including 8 species of beaked whale that are of conservation concern internationally (Hitchmough et al. 2007). The cryptic nature and paucity of data on cetaceans makes them generally unsuitable for use as ecosystem indicators.

2.4 Response Indicators

Response indicators measure institutional propensity and capability for furthering sustainability of marine ecosystems, (OECD 2003) and are an important part of a suite of indicators for measuring national progress towards sustainability. Important aspects of response include the current state of the scientific knowledge base needed to manage human impacts, the resourcing available to develop underpinning and applied knowledge, and an evaluation of what actions and systems are in place (or lacking) to promote sustainability.

2.4.1 Marine areas with some form of protection

Areas with legal protection considered here are offshore areas only; here we only consider areas with protection that are not adjacent to the coastline of the North or South Islands. Offshore marine protection have been established in three phases. In 1990, an area of 7280 km^2 around the Kermadec Islands was protected, and in 1997 a further 4980 km^2 around the Auckland Islands was protected. In November 2007 17 Benthic Protection Areas (BPA) with a total area of about 1.2 m km^2 were closed to bottom fishing methods, namely bottom trawling and dredging, in perpetuity [Fisheries (Benthic Protection Areas) Regulations 2007]. The BPAs are not closed to other forms of fishing such as , midwater trawling or bottom longlining. Areas under complete protection from fishing are about 0.2% of the oceanic EEZ, and those within the BPA about 23% of the oceanic EEZ. The "comprehensive-adequate-representative" method is often used to consider to what extent a protected area offers conservation value, and this approach can be modified to estimate how much of the New Zealand marine ecosystem is afforded protection by the BPAs and other spatial protection. There are two possible approaches to this.

First, from an ecosystem perspective, calculate the proportions of each characteristic ecosystem which is protected. Spatial maps where communities are predicted to be ecologically distinct are available from a number of New Zealand classification schemes, namely the Marine Environment Classification (MEC, Snelder et al. 2004, 2006)), the Demersal Fish Classification (DFC, Leathwick et al 2006a, b), and the Benthic Optimised MEC (BOMEC, Leathwick et al. 2009; Baird et al. 2009). The MEC covers the whole New Zealand EEZ, whereas the other two classication schemes only cover part of it. Of the area within the BPAs, 75% is deeper than 1950 m and not covered by the DFC, and 52% is deeper than 3000 m and not covered by BOMEC. In this section we are concerned with measuring *response*, that is, human activities that promote sustainable management of the marine ecosystem. Specifically in terms of measuring the effectiveness of spatial protection from fishing to mitigate adverse effects of this activity on the marine ecosystem. As the BPAs only offer protection to benthic ecosystems from bottom trawling and dredging we are only concerned with measuring their value in areas where bottom trawling and dredging can take place, that is, in depths less than 1950 m. This means that any of the three classification schemes (MEC, DFC, BOMEC) can be used to produce an Area Protected Index (API) as Equation 11.

Area Protected Index

$$API_{scheme} = \frac{\sum_{i=1}^{n}\left(\alpha_i \cdot \frac{A_i^{protected}}{A_i^{classification}}\right)}{\sum_{i=1}^{n}\alpha_i} \qquad [11]$$

Where for any of the classification schemes (API_{MEC}, API_{DFC}, API_{BOMEC}), $A_i^{protected}$ is the area of subregion *i* with protected status from some method of fishing, $A_i^{classification}$ is the total area of classification (MEC, DFC, BOMEC) subregion *i* that is able to be fished by that method, and α_i is a weighting factor that accounts for some subregions being more ecologically important that others. As in Section 2.2.1, in the absence of information to determine variations in ecological importance between large subregions, the default is to assume that subregions are equally important to ecosystem health and set α_i=1 for all *i*. Note that only the area of the classification subregion that is vulnerable to the given fishing method should be used in the denominator because we are using this indicator to measure how much protection is being afforded by the human action of affording spatial protection from fishing. Protection is only relevant to areas that are able to be fished. For example, for bottom trawling, $A_i^{classification}$ is the area of the subregion *i* that is shallower than 1950 m rather than the total area of subregion *i*.

Here, we suggest using this method only for the most extensive fishing method in the New Zealand EEZ, namely, bottom trawling. The three API indicators (API_{MEC}, API_{DFC}, API_{BOMEC}) have varying validity and usefulness. The MEC is a "bottom-to-top" classification and identifies bioregions that differ either in water-column properties, or benthic properties, or both (Snelder et al. 2004, 2006), and API_{MEC} theoretically contains information on the protection offered to the marine ecosystem from

bottom trawling. However, in regions where MEC overlaps with the BOMEC and DFC (i.e. depths less than 1950 m), the MEC is known to offer much poorer separation of regions that are distinct in terms of their demersal fish than the DFC (Sharp et al. 2007), and poorer separation of regions that are distinct in terms of their demersal fish and benthic invertebrate assemblages than BOMEC (Leathwick et al. 2009). To a large extent, BOMEC encompasses and extends DFC by adding 8 classes of benthic invertebrates to the 126 species of demersal fish (Leathwick et al. 2009). This means that we recommend using API based on BOMEC only.

As a second possible approach, one could calculate the proportion of all known species abundance that are protected from fishing. The predicted biomass distribution of 126 demersal fish species (DFC: Leathwick et al. 2006a, b), combined with distributions of squid could be used to generate a Biomass Protected Index (BPI) as Equation 12.

Biomass Protected Index

$$BPI = \frac{\sum_{i=1}^{n}\left(\beta_i \cdot \frac{B_i^{protected}}{B_i}\right)}{\sum_{i=1}^{n}\beta_i} \qquad [12]$$

Where $B_i^{protected}$ is the predicted biomass of species i in a location that is protected, B_i is the total predicted biomass of species i, β_i is a weighting factor that accounts for some species being more ecologically important that others. As in Section 2.2.1 β_i could be set equal to $TAC_i^{0.25}$ for QMS species and to produce an equivalent measure of secondary production for non-QMS species based on catch rates from trawl surveys, assumed catchabilities, and biological parameters from scientific literature. Extending this approach to include species of squid would be feasible since it is likely that distributions of commercially-exploited New Zealand cephalopods could be estimated based on commercial catch data following methods of Leathwick et al. (2006a, b). In the case of squid, no protection is offered by the BPAs as squid are taken from the water column. Including benthic invertebrates in this approach is not feasible at this time because of the lack of information on their biomass or distribution in the EEZ. Note that BPI only considers known distributions of species and if there were considerable unknown biomasses of species (e.g. large biomasses of demersal fish deeper than 1950 m, or substantial unfished squid populations), BPI would underestimate the protection value of the current spatial protection. This is not thought to be the case but further research on this issue would be useful in the medium-long term.

Note that API_{BOMEC} and BPI should not be combined as they measure fundamentally different things; API measures the proportion of BOMEC subregions protected and BPI measures the proportion of known demersal fish (and potentially squid) biomass protected. Improved approaches to evaluate the protection value of BPAs is required in the the medium-long term, and this includes further development and validation of ecological classification schemes such as DFC and BOMEC.

2.4.2 State of knowledge on fish stocks

The quality of information and state of knowledge on the status of fish stocks relative to management targets is an important measure of institutional progress towards the sustainable use of marine resources, and has previously been proposed for inclusion in New Zealand state of the environment reporting in 2000 (ME18, Gilbert et al. 2000). The Harvest Strategy Standard (Ministry of Fisheries 2008) will potentially improve the clarity and transparency of reporting stock levels against management targets, although only 22 stocks from a total of 629 were reported in this way in 2009 (Ministry of Fisheries 2009), and others will be added “as time allows” (Ministry of Fisheries 2009). A simple indicator of progress on the state of science underpinning the fisheries management system in New Zealand would be to assign a value describing the quality of information on its current state to each stock in the QMS, and average this across all stocks in proportion to their ecological importance. A simple point-based system is proposed to score the level of knowledge of stocks (Table 3), and these can be combined to give the State of Knowledge Index as Equation 13.

Table 3. Scoring table for reporting state of fish stocks against reference points.

State of Knowledge about stock status (SK)	Description
1	Quantitative estimate of stock level against specific target level. New fishery, never before fished.
0.5	Qualitative assessment. Plenary uses terms like "likely" or "believed to be"
0	Stock status not known. Plenary contains text like: "No estimates of stock abundance are available", "Biomass estimates are not available", "Estimates of yield are not available", "Fishing mortality is not known"

State of Knowledge Index

$$SKI = \frac{\sum_{i=1}^{n} (\beta_i \cdot SK_i)}{\sum_{i=1}^{n} \beta_i} \qquad [13]$$

Where SK_i is the knowledge status score of stock i from Table 3, and β_i is relative ecological importance of stock i. We propose to set β_i=1 for all stocks as an interim measure. Alternatively, the 4th root of TAC could be used as a preliminary proxy for β_i as suggested in Section 2.2.3 but here applied on a stock-by-stock rather than a species-by-species basis. As before, TAC is merely suggested an interim proxy for stock secondary production in a given fishing year, and where quantitative information of secondary production is available (for example, from a stock assessment), this should be used instead of TAC. The point-based SKI approach is recognised as being an imperfect measure, and the development of more formal assessments of the state of knowledge about the status of each stock relative to management targets would be welcome, for example building on the probability categories given in the 2009 Ministry of Fisheries plenary based on IPCC (2007).

2.4.3 Proportion of landings from stocks not in the QMS

The proportion of total fisheries landings from the New Zealand ocean (not coastal) domain that are from species which are not included in the New Zealand QMS would give an indication of the degree to which the New Zealand quota system is being used to regulate New Zealand fisheries (see also Section 2.2.5). It is reasonable to assume that there is much weaker control of fishing mortality of species not included in the QMS compared to those included in the QMS where catch is limited by an annual TAC limit. We note that this indicator is similar to a previously proposed indicator (ME31, Gilbert et al. 2000). At present, catches of non-QMS species are not systematically recorded, with the only observation likely to be on an ad hoc basis by fisheries observers. Consequently, although important, there is not likely to be a way to produce a useful indicator for this issue. It is recommended that data should start to be collected systematically on the catch of non-QMS species as a matter of urgency.

2.4.4 Ongoing overfishing

For a small number of fish stocks in the New Zealand QMS (76 out of 629), the current level of fishing is compared to that which is assessed to be sustainable under the MSY-based New Zealand Harvest Strategy Standard (Ministry of Fisheries 2008, 2009). A proportion of the stocks evaluated in this way are found to currently being fished at a level that is likely to lead to over-depletion in the future. This information gives a direct indication of the propensity of the management system to further sustainable usage of the marine ecosystem, and could be formalised into an ongoing overfishing indicator. The information on the different stocks should be combined in terms of the ecological importance of the stocks rather than simply the number of stocks, as all stocks are not equal. A point-based system is proposed to score the level of fishing of stocks relative to management targets (

Table 4), and these can be combined to give an Ongoing Overfishing Index as Equation 14.

Table 4. Scoring table for reporting state of fish stocks against reference points. Here, $F_{current}$ is the current fishing mortality on a given stock, and F_{target} is the target fishing mortality, which may be the fishing mortality that leads to B_{MSY} (or variant) in the medium to long-term.

Overfishing Status (OF)	Description	Current fishing level
1	No overfishing	$F_{current}<F_{target}$
0.5	Unknown	Not known
0	Overfishing	$F_{current}>F_{target}$

Ongoing Overfishing Index (OFI)

$$OFI = \frac{\sum_{i=1}^{n}(\beta_i \cdot OF_i)}{\sum_{i=1}^{n}\beta_i} \qquad [14]$$

Where OF_i is the overfishing status score of stock i from

Table 4, and β_i is relative ecological importance of stock i. The 4th root of TAC on a stock-by-stock basis could be used as a preliminary proxy for β_i, with better information on secondary production where available from quantitative stock assessments, as suggested in Sections 2.2.3 and 2.4.2.

2.4.5 Oceans research activity

Better understanding and monitoring of the New Zealand marine ecosystem is likely to indicate progress towards sustainability. It is beyond the scope of this paper to develop specific indicators of this socio-economic aspect of progress towards sustainability, but some suggestions are given to start the discussion. Improved scientific knowledge of the New Zealand marine environment is likely to lead to more sustainable human impacts on marine ecosystems. Whereas it is possible to fish sustainably with very low research cost, this is likely to be feasible only for low yields, and not at the MSY-like level specified by Ministry of Fisheries (2009). Increases in scientific knowledge could be measured as the number of research papers focussing on the New Zealand marine ecosystem (in the broadest context) published in each year. However, this approach would omit the very considerable commissioned research effort that leads to "grey-literature" reports rather than peer-reviewed publications. An alternative method used overseas is to measure how much is spent on oceans-related research each year (Hanson 2003), either as an inflation-adjusted value or as a proportion of fishery earnings (McKoy 2006). Such an index is likely to have high international comparability as similar statistics are produced by other OECD countries.

2.4.6 Cross-agency co-operation

Because of the statutory complexity of management of ocean resources in New Zealand (Willis et al. 2002), and the diversity of stakeholder interests, greater cross-agency engagement between stakeholder groups is likely to lead to improved sustainability of human interactions with marine ecosystems. It is hence likely to be useful and appropriate to develop and report socio-economic and/or institutional indicators of cross-agency engagement between groups including governmental/statutory management bodies, the fishing industry, fisheries research providers, iwi, non-governmental conservation organisations, and the New Zealand public. For example, the amount of research or routine scientific observation carried out from fishing industry vessels could be used to show collaboration between the fishing industry and research organisations. The development of data resources for establishing resilience and change in marine ecosystems could be measured by measuring the duration of fisheries time-series carried out by relatively repeatable methods. There are also examples of New Zealand fishing companies voluntarily choosing to lower catches below allowed levels (Griffith 2008) and this could be quantified and used as evidence of a commitment to sustainability by the New Zealand fishing industry. The commitment of New Zealand fishing companies to sustainability could be measured by their engagement with the Marine Stewardship Council (MSC). For example, the proportion of New Zealand landings which are of species with MSC accreditation could be a useful indicator of progress towards sustainability of New Zealand marine

ecosystems.

3 Results and discussion

From the presentation of this large number of candidate indicators, it is clear that no single indicator suffices – a suite of indicators is required. Furthermore, the most appropriate indicators are very context dependent, being affected by which data are available, the requirements of the stakeholders, and the characteristics of the ecosystem.

3.1.1 Evaluation of candidate indicators

The indicators proposed in Section 2 are evaluated in Table 5. A simple scoring system for each of the six MfE criteria (Table 1) was used, with 1=low, 2=medium and 3=high fulfilment of the criterion. Scores were totalled to give an overall score, assuming all criteria are of equal importance. Suggested indicators were then ranked. This method followed Rice & Rochet (2005), although many studies (e.g. Rochet & Rice 2005; Piet et al. 2008) noted that is it difficult to obtain an objective or unequivocal result using this approach. Detailed justification of ratings did not help improve consistency between different scorers. This result should hence be considered to be a preliminary suggestion of the most promising indicators to consider, with substantial interagency input within New Zealand needed before any indicator is deemed suitable.

Four pressure indicators, eight state indicators and five response indicators are suggested as being the most promising for reporting in 2012 (Table 6). Most of these new indicators will require research before 2012 if they are to be used in the 2012 state of the environment reporting. All will, or course, require close scrutiny by New Zealand stakeholders for validity and usefulness before adoption.

3.1.2 Changes in indicators warranting action

It is a valid concern that many of the indicators suggested here lack specificity: they do not allow stakeholders to disentangle, for example, changes to the marine ecosystem due to fishing from effects of climate change or changes in recruitment from land-based effects, or a combination of many causal factors. In other words, impacts cannot be easily separated from changes in state. However, I argue that first it is important to determine what characteristics of the marine ecosystem are important and can be monitored. Then, if we see an adverse change in this suite of indicators, action to understand the change and determine what (if anything) we can do to mitigate, remedy or manage the change is warranted. As a policy of precautionary resource use, it is wise to seek to understand change in any and all indicators selected, but a number of factors may suggest more urgent investigation is warranted. First, what is the direction of change? A criterion for selecting indicators is that change in a given direction is clearly identified as desirable or undesirable, with undesirable changes warranting more urgent attention. Second, what is the magnitude of the change, with “rapid” change likely to be more important? Theoretical assessment of the significance of the rate of change is likely to be not possible, so an historical perspective, based on a time series of information, will generally be needed. Third, are the fluctuations slowing down or accelerating? Consistent, adverse changes over a number of years, especially where the changes are getting larger each year, are of more concern than a fluctuating indicator. Again, time series are important because they give some indication of the degree of variation between years. These three aspects can be qualified for most indicators recommended here for further consideration (Table 6). Fourth, and more difficult, is the indicator approaching a reference point? The reference point may have ecological meaning (for example, a threshold), but will more often be a management limit. In summarising the results of the SCOR/IOC Working Group on quantitative ecosystem indicators for fisheries management, Cury & Christensen (2005) conclude that indicators only show if the ecosystem is strongly affected so that all unfavourable changes to indicators should be treated as significant, even if reference points are lacking. Finally, if many variables together show unfavourable changes, this consistency provides evidence that more significant underlying change is occurring and warrants more urgent action. This action may involve investigation to see if we can we find any other information outside the set of indicators that can elucidate the underlying causes and suggest appropriate management action.

4 Future work

4.1.1 Relative ecological importance of species and stocks

A recurring theme in this work has been that a method of estimating the relative ecological importance of fish species is required to combine species-specific information into aggregate headline indicators. While secondary production scaled by a quarter power may be an appropriate interim measure, information on the types of ecosystem in the New Zealand oceanic EEZ (e.g. whether wasp-waisted or not, Rice 1995; Cury et al. 2000), and ecological roles of species within these ecosystems are needed to appropriately estimate ecological importance. Such assessment needs to be conducted at both the species and stock level, and ideally include non-QMS as well as QMS species. This is a considerable task, but progress is being made under FRST funded projects (e.g. Coasts and Oceans OBI), and commissioned research (e.g. Ministry of Fisheries, Dunn et al. 2009, Tuck et al. 2009).

4.1.2 Validation and relative ecological importance of bio-regions

Validation and consideration of the relative ecological importance of bioregions identified by various bioregionalisation or marine environmental classification schemes for the New Zealand region is required. Three environmental marine classification schemes are available for the New Zealand EEZ: MEC, BOMEC and the demersal fish community classification. These have different aims, methodologies, cover different proportions of the New Zealand oceanic EEZ and suggest different bioregions. These require substantial further research to validate. It may be appropriate to consider all bioregions from a given classification as equally important ecologically, but from different perspectives, some regions are certainly much more important than others. For example, the proportions of total commercial catches taken from different bioregions are very uneven. Also, total primary productivity, average depth, and the proportion of total area protected vary considerably between bioregions, and this could be used to weight how data are used to provide aggregate headline indicators. Again, research is ongoing on these issues but results are not available at present.

4.1.3 Better fisheries information should be collected

The ability of New Zealand to assess and report some aspects of the sustainability of its oceanic ecosystems is limited by the quality of the available information. As noted by Mace (2004): “The single most valuable tool for assessing the status of individual stocks, biological communities, and habitats has proven to be consistent time series of data on catches, relative abundance, size distributions, and other biological and physical information. Unfortunately, few such time series exist.” Mace (2004) may have been focussing on fishery-independent data (such as research trawl surveys), but ensuring better data from observers on commercial fishing vessels is also important. Improved data collected would enable improved monitoring of the state of the New Zealand marine environment, in particular:

- All catches should be traceable to the location of capture. At present, landing returns must be retrospectively assigned to the location of fishing, a requirement that increases error in data. This could be helped by a requirement for large vessels (>28 m) to estimate and report catches (wet weight) of more species than just the top 5 on each tow on TCEPR (Trawl Catch Effort Processing Returns);
- Vessels less than 28 m completing the Trawl Catch Effort Return (TCER) should be required to give a start and an end position for a tow, rather than just the start position. The current lack of an end position leads to uncertainty in measuring where catches were taken;
- Catches of non-QMS finfish species should be reported. Minor fish species may have an important functional role in New Zealand ecosystems, but the fact that catches of non-QMS species are poorly or not reported at all, fundamentally limits our ability to monitor changes in these species;
- Length of major finish species should be measured routinely to allow changes in size composition in the community to be investigated;
- Catches of benthic invertebrates should be reported by commercial fishers using bottom trawls. Training in identification and provision of identification guides for benthic invertebrates are likely to be needed to improve the quality of this information to a useful standard. Such information, even at a rudimentary (e.g. genus, family) level, is needed to

provide a method of identifying and monitoring long-term change in benthic ecosystems.

4.1.4 Clearer and more transparent reporting on stock status of QMS species is needed

The way in which the status of QMS stocks is reported in the annual Ministry of Fisheries plenary is inconsistent (Ministry of Fisheries 2009). A more consistent method of reporting the estimate of the status of each stock, the sustainability of the present level of fishing on each stock, and the level of uncertainty in these assessments would facilitate reporting this information for State of the Environment and other purposes.

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Table 5. Evaluation of proposed oceans indicators for the New Zealand offshore EEZ. Criteria are from MfE (2007). "Type" is Pressure (P), State/Impact (S/I), Response (R). Each criteria is ranked as 1=low, 2=medium, 3=high. Overall score is the sum of scores for the six criteria, assuming all criteria are equally important. Overall rank is given according to type. Indices discussed in the text but with no definite indicator given are not included. These include the use of aspects of the biology/ecology of individual predator species, work currently underway by the New Zealand Marine Environment Monitoring Programme (MEMP), and indicators of cross-agency co-operation.

Type	**Indicator**	National signif.	Relevant	Credible	Interpret-able	Cost-effective	Internat. signif.	**Overall score**	**Rank**
P	Total fishing removals	3	3	3	3	3	3	18	P1
	Commercial trawled footprint	3	3	3	3	3	2	17	P2
	Area Trawled Index, ATI (based on BOMEC)	3	2	3	2	3	1	14	P3
	Biomass Trawled Index, BTI	3	3	2	2	2	1	13	P4
	Corrected Fishing in Balance Index, cFIB	3	2	2	1	2	2	12	P5
	Fisheries Pressure Index, FPI	3	2	1	1	2	2	11	P6
S/I	Status of fish stocks vs management targets, SSI	3	3	2	3	3	3	17	S1
	Mean Trophic Index, MTI (research & commercial)	2	3	2	3	3	3	16	S2
	Satellite ocean observation (change in EOFs)	3	3	3	2	2	1	14	S3=
	Surface chlorophyll-a concentration, *chl-a*	3	1	2	2	3	3	14	S3=
	System-level CPUE, sCPUE	3	3	2	2	2	2	14	S3=
	Threat status of species (DoC threat classification)	3	2	3	1	3	2	14	S3=
	Demersal fish diversity, DDI (research & commercial)	3	3	2	2	2	1	13	S7=
	Species distributions (commercial catch)	3	3	2	2	2	1	13	S7=
	Feeding type ratios (commercial catch)	3	2	2	2	2	2	13	S7=
	Demersal fish biomass	2	3	1	3	2	1	12	S10=
	Threat status of species (IUCN "red list")	2	1	2	1	3	3	12	S10=
	Net primary production, NPP	3	2	1	2	2	1	11	S12=
	Mesopelagic acoustic backscatter	1	3	2	3	1	1	11	S12=
	Abundance Biomass Curves (commercial catch)	3	2	2	1	2	1	11	S12=
	Climate state (IPO, SOI, Kidson, Trenberth)	3	1	2	1	2	1	10	S15=
	Zooplankton assemblage, CPR	1	2	3	2	1	1	10	S15=
	Relative Price Index	3	2	1	1	1	1	9	17
R	Ongoing overfishing, OFI	3	3	3	3	2	2	16	R1
	Total area with some form of protection	3	1	3	2	3	3	15	R2
	State of knowledge (stock status), SKI	3	3	2	2	2	2	14	R3
	Area Protected Index, API	3	3	1	1	3	1	12	R4=
	Biomass Protected Index, BPI	3	3	1	1	3	1	12	R4=
	Oceans research activity	2	1	2	2	1	3	11	R6

Table 6. Suggested promising candidate indicators for state of the New Zealand oceans for reporting in 2012. Current state of the environment indicators (MfE 2007) are shown in grey. P=pressure, S/I=state/impact, R=response

Type	Indicator	Current indicator	Key work required
P	Total fishing removals	Y	Update current indicator
	Commercial trawled footprint	Y	Update and modify current indicator
	Area Trawled Index, ATI (BOMEC)	N	Determine relative importance of BOMEC regions or assume equally important
	Biomass Trawled Index, BTI	N	Determine relative ecological importance of QMS and non-QMS fish species
S/I	Status of fish stocks, SSI	Y	Determine relative ecological importance of QMS stocks
	Threat status of species (DoC threat classification)	Y	Update current indicator
	MTI (research & commercial)	N	Validate trophic levels of major fish species
	Satellite ocean observation (EOF change)[1]	N	EOF characterisation of New Zealand EEZ (satellite *chl-a* could be used in interim)
	System-level CPUE	N	Determine subareas and their relative ecological importance
	Demersal Fish Diversity, DDI (commercial catch)	N	Determine subareas; correct for changes in species identification quality over time
	Feeding type ratios (commercial catch)	N	Determine functional groups; correct for spatial changes
	Species distributions (commercial catch)	N	Determine species to analyse; develop method for commercial data
R	Ongoing overfishing, OFI	Y	Determine relative ecological importance of QMS stocks evaluated for overfishing
	Total area with some form of protection	Y	Update current indicator
	State of knowledge (stock status), SKI	N	Determine relative ecological importance of QMS stocks
	Area Protected Index, API	N	Validate and determine relative importance of BOMEC regions
	Biomass Protected Index, BPI	N	Biomass of finfish deeper than DFC lower limit (1950 m); distribution of squid biomass in EEZ

[1] Only one indicator of large-scale change in the oceanographic state of the New Zealand EEZ is probably appropriate, and we suggest that changes in characteristic EOFs of temperature, ocean colour and sea-surface height from satellite observations is likely to be a more useful indicator than simple changes in surface chlorophyll-a concentration (*chl-a*) from ocean colour satellite sensors. However, if the required research to determine the EOF oceanographic index cannot be carried out to obtain the before in 2012, *chl-a* could be used instead.

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Appendix 1: Trophic levels of New Zealand marine biota

Trophic levels for marine biota (Lindeman 1942) are commonly estimated in three ways: (1) stable isotope analysis; (2) stomach contents analysis; (3) food web models.

Method 1: The most direct method is to analyse samples of muscle from a number of individual specimens of an organisation for the relative composition of the two stable isotopes of nitrogen. Nitrogen naturally occurs as ^{14}N and ^{15}N in proportions of about 99.6:0.4. The relative abundances of these isotopes are typically expressed as δ representing differences from a given standard in units of parts per thousand (‰). When organic matter is created by phytoplankton growth there is typically enrichment in the heavier isotope (^{15}N) relative to the inorganic nitrogen source. Organic matter is then progressively enriched in ^{15}N by about 3.4 ‰ tends every time matter is consumed (Post 2002). Nitrogen isotopes from different food sources are conservatively, so that values of $\delta^{15}N$ in the tissues of an organism relative to those in a reference organism show the degree of trophic separation (DeNiro & Epstein, 1981; Minagawa & Wada, 1984; Wada et al. 1991; Vander Zanden & Rasmussen, 2001), Equation A1.

Trophic level $$\mathrm{TL}_i = \left(\frac{\delta^{15}\mathrm{N}_i - \delta^{15}\mathrm{N}_{ref}}{3.4} \right) + \mathrm{TL}_{ref} \qquad [A1]$$

Where $\delta^{15}N$ is the fraction of heavy nitrogen isotope in species *i* ($\delta^{15}N_i$) or the reference species ($\delta^{15}N_{ref}$) and TL is the trophic level of species *i* (TL_i) or the reference species (TL_{ref}). Herbivores or grazers (TL_{ref}=2, e.g. small zooplankton) are often used as the reference because the isotopic composition of such species tends to be more stable than those of primary producers. Lipids are typically extracted from fish muscle before analysis as differential lipid concentrations between species can skew estimates of trophic level (Ricca et al, 2007). This approach has been carried out for more than 22 key species of New Zealand fish (Bury et al. 2006), and more are under analysis at the time of writing.

Method 2: Analysis of stomach contents of fish can be used to estimate the proportions of various prey items in their diet. Trophic level of the fish predator is then calculated as the weighted average of the trophic level of the prey organisms plus one. The trophic level of prey organisms are often taken from the scientific literature under the assumption that the trophic level of a given organism is phenotypically constrained and hence the same irrespective of the type ecosystem it occurs in. Agreements in trophic level between gut contents and stable isotopes are usually good (e.g. R^2=0.48, p=0.0001, Stergiou & Karpouzi 2002). Stomach contents analysis of many New Zealand fish species are available (e.g. Clark 1985; Clark et al. 1989; Rosecchi *et al.* 1988; Dunn et al. 2009) but have not yet been used to estimate trophic levels.

Method 3: Finally, mass balance trophic models, where flows of organic matter between organisms are balanced across the whole food-web can be used to estimate trophic levels (Christensen & Pauly 1992). In the absence of comprehensive experimental measurements of trophic level of New Zealand fishes, estimates have been obtained using data in Fishbase (Froese & Pauly 2000) by assuming that species in the New Zealand EEZ have the same trophic level as that species has in other marine ecosystems elsewhere in the world (e.g. Tuck et al. 2009; Knight & Jiang 2009).