Review of New Zealand's coastal and marine habitat and ecosystem classification

Prepared for Department of Conservation

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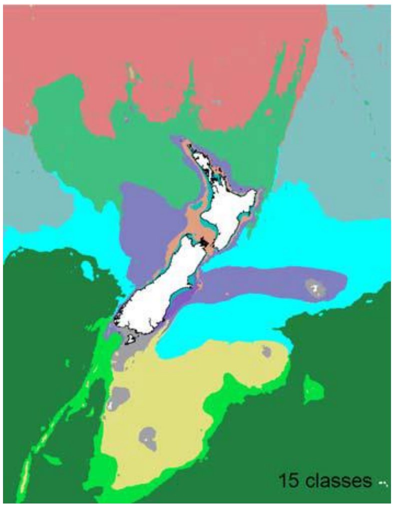

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Contents

Executive summary

Marine habitats and ecosystems are increasingly impacted by human activities such as sedimentation, pollution, eutrophication, bottom fishing, oil drilling, waste disposal, and seafloor mining. These human impacts threaten biodiversity and ecosystem function, resulting in a need for management and conservation of the marine environment. Networks of marine protected areas (MPAs) can be used to exclude harmful human activities, facilitating the protection/restoration of biodiversity and ecosystem functions. In order to design an effective MPA system it is necessary to identify areas that contain, or can be predicted to contain, distinct habitats and ecosystems. There are many means by which habitats/ecosystems can be identified, but classification is among the most common methods used.

The Ministry for the Environment (MfE), Department of Conservation (DOC), and Ministry for Primary Industries (MPI) are currently reviewing New Zealand's approach to establishing MPAs. As part of this initiative, there is an opportunity to review and potentially refine or replace the Coastal and Marine Habitat and Ecosystem Classification (CMHEC) system that is currently designed to inform MPA planning. To inform decisions around the extent to which the CMHEC may need to be refined or replaced, DOC commissioned the National Institute of Water and Atmospheric Research (NIWA) to undertake this review of existing broad-scale marine habitat and ecosystem classification systems, including both New Zealand-specific and international systems, that are or potentially may be used for MPA planning. This review is necessary because: a suite of different MPA types may be applicable in New Zealand; the scope of any new MPA legislation or policy may include the terrestrial, coastal, and offshore marine (the Exclusive Economic Zone (EEZ)) areas; there have been New Zealand and global developments in knowledge of coastal and marine habitats, as well as approaches to modelling distribution of habitats and classifying habitats; and there are known issues with the existing CMHEC.

There are two main types of classification, thematic and numerical, each with pros and cons. This review describes in detail several classifications of both types, and highlights the benefits and drawbacks of the two approaches. There are classifications that involve both classification approaches (so-called 'mixed' classifications), but these are usually based primarily on a thematic approach.

Thematic classifications, such as the CMHEC, have been a popular choice for conservation management - in part because of their conceptual simplicity and relative ease of application, which means they are also easily interpreted and understood by the public. Such classifications also lend themselves to including both pelagic and benthic components of coastal and marine habitats, and can usually be readily aligned with estuarine and/or terrestrial classifications. Of the major thematic classifications of coastal and marine habitats reviewed here, the USA's Coastal and Marine Ecological Classification Standard (CMEC) is probably the best developed and could be a suitable candidate for replacing or modifying New Zealand's CMHEC. However, New Zealand conservation and management agencies would need to commit to actively engaging long-term in the structured feedback process to help maintain, modify and improve CMEC. For example, to practically use CMEC will require immediate additional work around what components, classes and subclasses should be used to create appropriate habitat maps for New Zealand MPA planning. Thus, adopting the CMEC is just the first step in a potentially time-costly process to create modifications that allow CMEC to be fully suitable for local use. There are obvious risks associated with this process being out of full New Zealand control.

There are disadvantages of using thematic classifications such as CMEC, some of which are overcome by the advantages of using numerical classification approaches. Numerical classifications are usually built using biological as well as physical data, and thus represent a direct statistical linkage between environment and biotic assemblages. The underlying models for numerical classifications can be validated, and testing the accuracy of these in the field is relatively straightforward. Uncertainty in the underlying models and thus the classification can also be expressed, including spatially. With the availability of new data, the underlying numerical methodology means that the classifications can be easily and quickly re-run, and thus this type of classification lends itself to continual improvement. Numerical classification approaches are flexible to using different types of data and at different scales, something that is more troublesome for thematic classification approaches. Numerical classifications can also be readily applied at different spatial scales. Numerical classifications can be built by modelling species distributions, which means the individual species distribution models underpinning the classification can also be extracted and used to identify sites for MPAs designed to protect particular species of concern. Derived outputs or component parts of numerical classifications are particularly well-suited for supporting efforts to identify Ecologically and Biologically Significant Areas, Vulnerable Marine Ecosystems, and Key Ecological Areas, and assess the potential impacts of human disturbance. Because the models that are used to build numerical classifications involve a predictive component, they can be used to assess how biodiversity might respond to future climate change, and thus MPA planning can also take these potential changes into account.

However, numerical classifications do have some disadvantages; the main one being that the classification methodologies are not readily understood by non-scientists, and the results are also not always intuitively understood by environmental managers and public. That is, the identified classes do not always lend themselves to obvious names or description that conform to peoples’ perceptions of a habitat or biotic assemblage. This lack of association with a personal viewpoint can be a significant issue when engaging in stakeholder consultation to identify MPAs and design networks. However, the issue can be overcome if appropriate explanation documents/webinars, other web-based products, and application-tools are constructed and made freely available on the internet. Thus, it is the recommendation of this review that a numerical classification or classifications are developed for the coastal and marine habitats of New Zealand. In general, numerical classifications are more flexible in their construction and use, and considerable expertise and experience in developing numerical classifications already exists in New Zealand. However, it will be essential that, once built, sufficient resourcing will be provided to support the on-going maintenance and application of the classification(s).

1 Introduction

Marine habitats and ecosystems are increasingly impacted by human activities such as sedimentation, pollution, eutrophication, bottom fishing, oil drilling, waste disposal, and seafloor mining (Halpern et al. 2008). These human impacts threaten biodiversity and ecosystem function, resulting in a need for management and conservation of the marine environment (Ramirez-Llodra et al. 2011). Because human impacts are ecologically broad-based, conservation requires an ecosystem approach to management; i.e., the adverse effects of human activities on whole ecosystems must be managed considering the linkages between living and non-living components, and the connectivity of marine populations (Arkema et al. 2006). One component of an ecosystem approach to management is the use of networks of marine protected areas (MPAs) to facilitate the protection/restoration of biodiversity and ecosystem functions (Halpern et al. 2010). A substantial scientific literature now demonstrates that well-designed MPA networks can be highly effective tools for conserving biodiversity and ecosystem services (Edgar et al. 2014).

Ideally, the design of MPA networks should follow four sequential steps: (1) Evaluation of conservation needs, (2) Definition of the objectives for establishing the MPAs, (3) Integration of information on the biological characteristics (e.g., life histories, dispersal patterns, species distributions) and habitat distribution of the managed ecosystem, and (4) Selection of suitable sites to serve as MPAs (Protected Areas, 2001). Underlying scientific objectives of an MPA network include the preservation of representative and unique marine habitats, as well as the conservation of marine biodiversity, and ecosystem structure and function (e.g., Roberts et al. 2003). Thus, to design such an MPA system it is necessary to first identify areas that contain, or can be predicted to contain, distinct habitats and ecosystems (potentially also areas that will allow for the restoration of 'lost' habitats). There are many means by which habitats/ecosystems can be identified, but classification is among the most commonly used methods (Costello 2009). In the marine environment, where there is frequently an absence of extensive, comparable, high quality biological data, classification is often performed using environmental variables that define a combination of habitat or ecosystem characteristics that are likely to control the distribution, composition and function of pelagic and/or benthic biodiversity (e.g., Gregr and Bodtker 2007). Although environmental surrogates are assumed to be able to identify areas that support different pelagic and/or benthic faunas, classifications that rely on them wholly are unable to provide detail about the biotic communities, and are further compromised if the underlying assumptions of surrogacy are not tested (Stevens and Connolly 2004, Dixon-Bridges et al. 2014).

1.1 Motivation for this review

The Ministry for the Environment (MfE), Department of Conservation (DOC), and Ministry for Primary Industries (MPI) are currently reviewing New Zealand's approach to establishing MPAs (proposed Marine Protected Areas Act; MfE 2016). As part of this initiative, there is an opportunity to review and potentially refine or replace the *Coastal and Marine Habitat and Ecosystem Classification* (CMHEC) system that is currently designed for the purposes of MPA planning, monitoring and reporting (Ministry of Fisheries (MFish) and DOC 2008).

To inform decisions around the extent to which the CMHEC may need to be refined or replaced, DOC commissioned the National Institute of Water and Atmospheric Research (NIWA) to undertake this review of existing broad-scale marine habitat and ecosystem classification systems that are or potentially may be used for MPA planning, including both New Zealand-specific systems, and those used internationally. Such classification systems may include those based on a biogeographic or bio-regionalization approach, physical surrogates for biodiversity, and/or include biological data.

As part of the MPA reform (primarily to provide an avenue for the science required to inform the development of any new MPA policy and legislation), an Interim Science Advisory Group (ISAG) consisting of science staff from DOC, MPI and MfE has been established, and have provided advice and support for this review. Specifically, the ISAG provided information, based on their experience with using the existing CMHEC, on the strengths and weaknesses they see with that classification.

A review of the existing CMHEC is necessary because, the ISAG has identified that:

- A suite of different MPA types may be able to be applied in New Zealand (e.g., protected areas focussed on species conservation), requiring a different approach to habitat and ecosystem classification;
- The scope of any new MPA legislation or policy may include the coastal (including estuaries) and off-shore marine areas, the Exclusive Economic Zone (EEZ) and (for protected areas focussed on species such as seabirds and marine mammals) terrestrial habitats (e.g., seabird nesting areas; seal haul-outs or colonies);
- There have been developments in New Zealand and globally relating to knowledge of coastal and marine habitats, as well as approaches to modelling distribution of habitats and classifying habitats;
- There are known issues with the existing classification system (e.g., it is composed of two separately structured classifications, and known habitats are not identified; see Section 5 for all identified issues).

1.2 Aim and objective

The aim and objective of this project is to: deliver a report that provides comprehensive, peer-reviewed technical advice to the ISAG on coastal and marine habitat classification schemes. This report will review classification schemes and approaches (national and international) with the aim of informing decisions around the extent to which the existing New Zealand classification schemes may need to be refined or replaced.

2 Main types of classification

There are a variety of classification techniques available, but the two most popular methods are thematic (also known as hierarchical), and numerical (also known as multivariate) classifications. There are also classifications that combine elements of both approaches. Each of the two main approaches has advantages and disadvantages (Table 2-1). Numerical classifications are generally bottom-up statistical grouping of multiple (usually) continuous variables. The grouping procedure is largely objective and the ability of this type of classification to produce ‘natural’ groupings, although debated, is more likely (i.e., reflect direct relationships among environmental parameters, with links to species composition data when included). However, such classifications are relatively conceptually complicated and the process and output not so readily understood by stakeholders. In addition, there is no standard analytical procedure, and the choice of the number of classification groups can be subjective. Despite their mathematical robustness, they have not been widely adopted by environmental managers.

On the other hand, thematic classifications are generally top-down sub-divisions of individual information layers. The divisions are subjectively made and thus there is the potential for identifying 'unnatural' groupings (i.e., ones that don't necessarily reflect a direct relationship between environmental parameters and species composition of a habitat). However, the thematic classifications are a very simple concept, there is essentially only one method, and the process and output readily understood by stakeholders. Thus, this type of classification has been widely used to catalogue and identify habitats, and is thought to be particularly suited to large-scale conservation planning programmes such as MPA network identification (e.g., Zacharias and Roff 2000).

Table 2-1: Summary of the advantages and disadvantages of the two main types of classification used for classifying habitats and ecosystems for conservation and management.

Classification	Advantages	Disadvantages
Thematic	simple concept	subjective decisions
	understandable process and output	potential for unnatural groupings
	one basic method	
Numerical	objective	complex concept
	potential for natural groupings	process not readily understandable
		various methods

3 New Zealand classifications

A number of classifications of estuarine, coastal and marine habitats and ecosystems have been developed and produced for the New Zealand region. There are land and river classifications also, which are first briefly included in the review below because of the potential usefulness of aligning elements of terrestrial and marine classifications.

3.1 Land

Two primary classification systems have been developed for terrestrial environments in New Zealand. The Land Cover Database (LCDB v4.1) is a thematic classification system that identifies land cover (i.e., what vegetation type is growing on the ground or what feature covers the ground), and is based on grouping together of similar feature classes that can be identified in satellite images (https://lris.scinfo.org.nz/). LCDB includes a total of 33 mainland classes, with two additional offshore Chatham Island classes. The Land Environments of New Zealand (LENZ) is a numerical classification that uses 15 climate, landform, and soil factors to develop a surrogate classification based on features that are considered likely to influence the distribution of animal or plant species. LENZ is available at classification levels of 20, 100, 200 or 500 land environments (Leathwick et al. 2002).

3.2 Rivers

The River Environment Classification (REC) classifies the rivers of New Zealand (Snelder et al. 2004, updated 2010). The REC groups rivers, or parts of rivers, at six hierarchical levels (Climate, Source-of-Flow, Geology, Land-Cover, Network-Position and Valley-Landform) which are further subdivided into categories that discriminate variation in these characteristics. Therefore, each level of the REC classification hierarchy reflects differences in a set of processes (for example hydrological processes) that are assumed to be the cause of patterns in physical and biological characteristics at a range of spatial scales. The location of each REC class can be mapped so that the class of any section of a river in New Zealand can be identified. Mapping can be used to assess the spatial distribution of river resources and values across an area based on an understanding of the way factors control the physical and biological characteristics of rivers (Snelder et al. 2004, updated 2010).

3.3 Estuaries

The first classification of coastal environments in New Zealand was for estuaries and based on residence time (Heath 1976). Later, more comprehensive estuarine classifications were based on geomorphology and hydrology, and made for the purpose of supporting resource management (Hume and Herdendorf 1988, Hume et al. 2007). The current *Coastal Hydrosystem Classification* (Hume et al. 2016) builds on the *Estuarine Environment Classification* of Hume et al. (2007) and therefore includes estuaries, but it also links with the New Zealand's *River Environment Classification* (Snelder et al. 2004), and includes features recognised in related wetland classifications (e.g., lagoons; Johnston and Gerbeaux 2004) as well as features not recognised by previous classifications (e.g. deep fjords). The *Coastal Hydrosystem Classification* is a six-level, multi-spatial scale thematic classification primarily focused on hydrology and geomorphology, but it also includes levels that relate to biogeography/bioregions and habitat. The single class within the highest Global level is identified using the Marine Ecoregions of Spalding et al. (2007), while classes within the lower Structural and Composition levels are identified with reference to habitat components such as substrate, vegetation and dominant biota. There are 11 classes and up to 5 sub-divisions within some of these classes; resulting in a maximum of 26 classes/sub-classes in this classification (Table 3-1). The distribution of these classes around is shown in Figure 3-1).

Table 3-1: (a) High-level classes, and (b) example of sub-classes for Geomorphic class of the Coastal Hydrosystem Classification (source: Hume et al. 2016).

(a)	Level	Controlling factors	Spatial scale (km^2)
I	**Global** Temperate Australasian Realm	Climate, landmass, watermass	Macro 10^6-10^4
II	**Hydrosystem** Palustrine, lacustrine, riverine, estuarine, marine	Landform, water regime	10^3 Meso 10^1
III	**Geomorphic Class** 11 classes and 21 subclasses	Geomorphology, hydrodynamics	
IV	**Tidal Regime** Subtidal, intertidal, supratidal	Inundation by the tide	
V	**Structural Class** Vegetation, substrate, water structure	Bio-, geo-, and hydro-components	1 Micro 0.1
VI	**Composition** Dominant biota, substrate and water types	A mixture of above	

(b) Geomorphic Class	Subclass
1. Damp sand plain lake	
2. Waituna-type lagoon	A. Coastal plain depression; B. valley basin
3. Hāpuna-type lagoon	A. Large; B. Medium; C. small; D. intermittent
4. Beach stream	A. Hillside stream; B. damp sand plain stream; C. stream with pond; D. Stream with ribbon lagoon; E. intermittent stream with ribbon lagoon
5. Freshwater river mouth	A. Unrestricted; B. deltaic; C. barrier beach enclosed
6. Tidal river mouth	A. Unrestricted; B. spit enclosed; C. barrier beach enclosed; D. intermittent with ribbon lagoon; deltaic
7. Tidal lagoon	A. Permanently open; B. intermittently closed
8. Shallow drowned valley	
9. Deep drowned valley	
10. Fjord	
11. Coastal embayment	

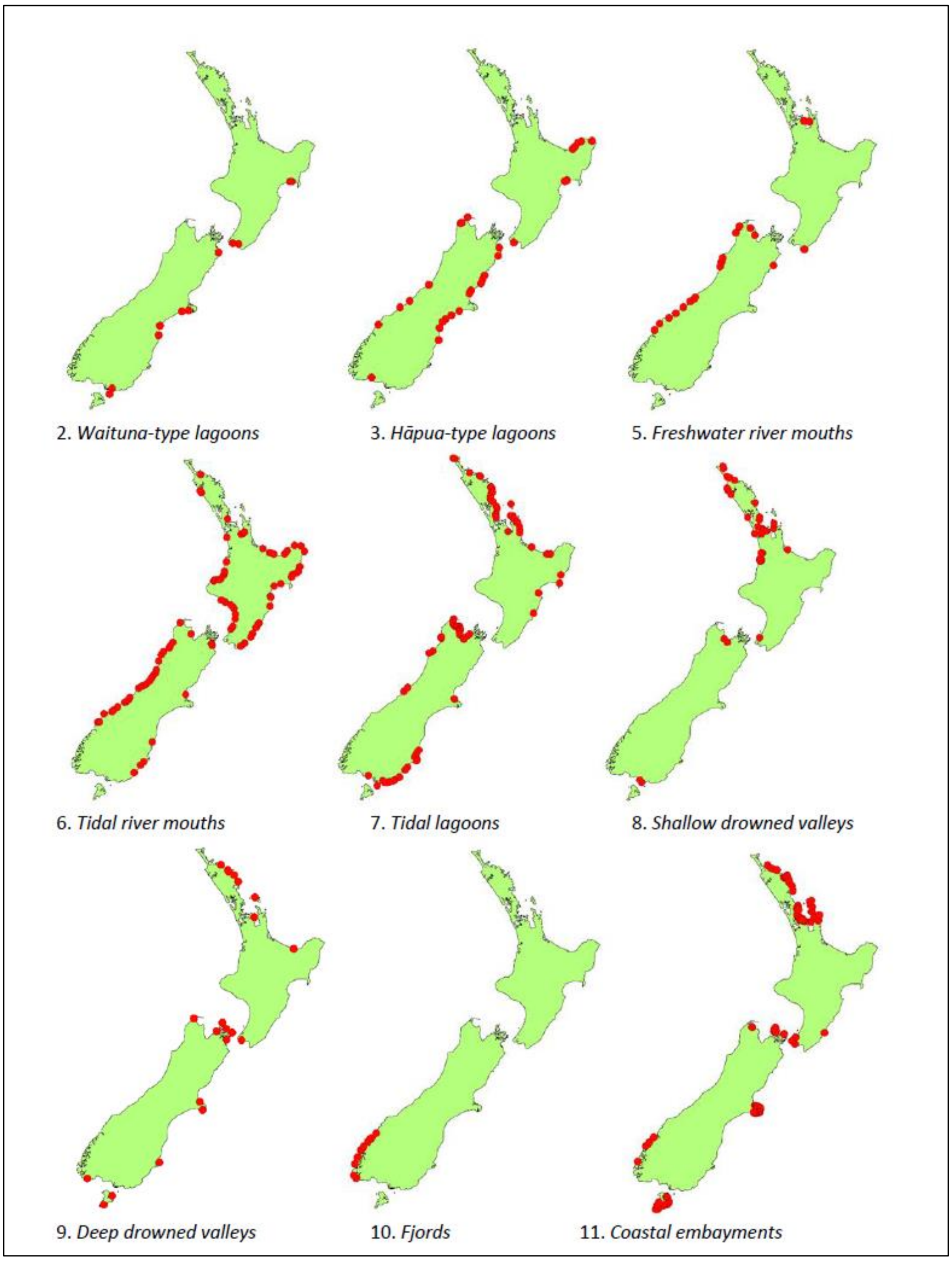


Figure 3-1: Distribution of the geomorphic classes (2, 3, 5-11) of the Coastal Hydrosystem Classification throughout New Zealand (source: Hume et al. 2016).

3.4 Seamounts

The New Zealand *Seamount Classification* is a numerical classification based on a multivariate (group average hierarchical clustering) analysis of thirteen mostly physical surrogates (only one biological variable, Chlorophyll *a*, was included) for seamount habitat (Rowden et al. 2005). Twelve classes, or groups of seamounts with similar environmental characteristics, were identified by this analysis (using a subjective similarity cut-off level that would produce a relatively small number of groups whilst still retaining a relatively high level of dissimilarity between groups). The groupings of seamounts generally displayed a geographic distribution throughout the New Zealand region (EEZ, Extended Continental Shelf and beyond), and were largely characterised by a combination of four variables (depth at peak, depth at base, elevation, and distance from continental shelf). The biological meaningfulness of these classes was not tested, and the analysis included only a subset of just over half the known seamounts at the time (environmental variables were not available for all seamounts).

A later thematic five-level *Global Seamount Classification*, which used four physical surrogates, nested within a biogeographic scheme (Figure 3-2), was compared with the results from a numerical classification using the same variables for a sub-set of seamounts from the New Zealand region (Clark et al. 2011). In this case, the number of groupings identified by numerical classification of New Zealand seamounts were objectively identified (using a statistical test for the presence of group structure in data) rather than using a subjective similarity cut-off as applied by Rowden et al. (2005). As expected, the unconstrained numerical classification produced more classes overall than the thematic classification (57 v 22 classes, respectively). Nonetheless, the correlation between the two types of classification was statistically significant, although it was not particularly strong (ρ 0.476 at P = 0.1%). However, there was a degree of visual concordance between the spatial distributions of the seamount classes produced by the two classifications in the New Zealand region (Figure 3-3). This result suggested that the numerical classification represented, in effect, a finer spatial scale representation of the thematic classification, rather than a fundamentally different classification (Clark et al. 2011).

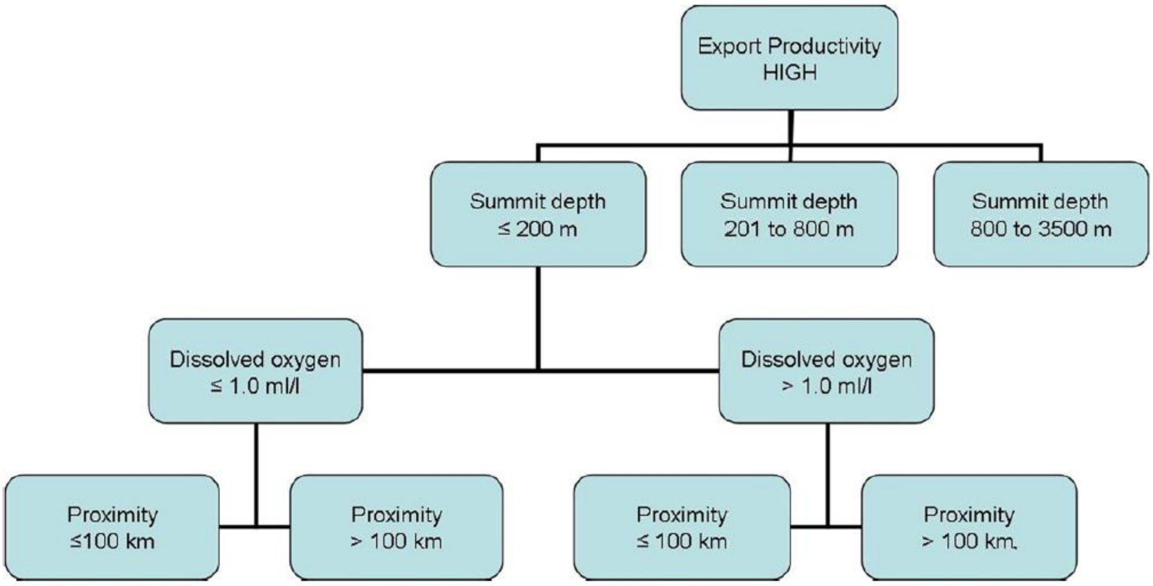


Figure 3-2: Organisation schema of the Global Seamount Classification showing divisions within a Biogeographic Province (top level of classification) and Summit Depth <200 m (divisions within other Biogeographic Provinces and Summit Depth classes are the same (source: Clark et al. 2011).

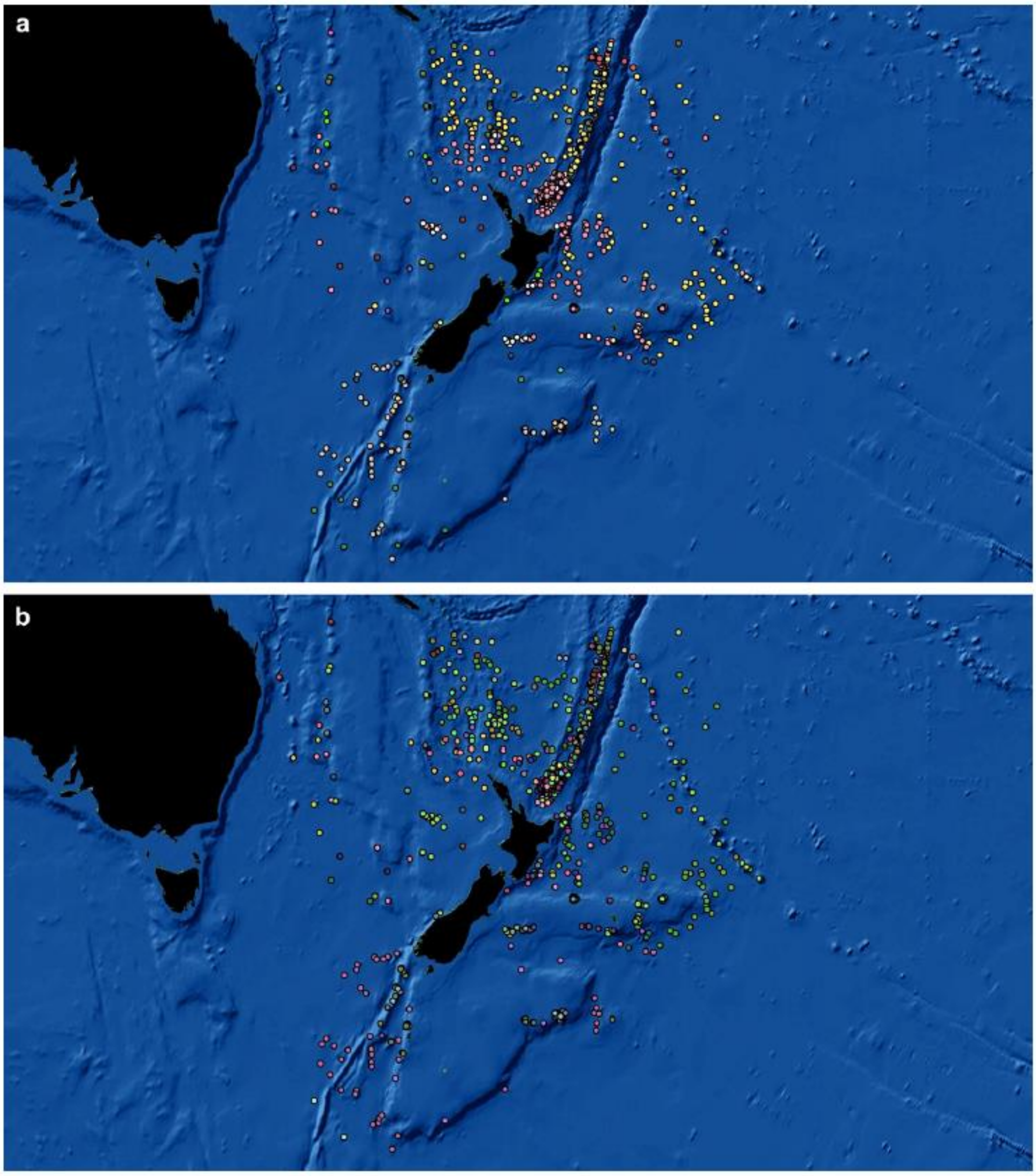


Figure 3-3: (a) Results of the Global Seamount Classification for seamounts in the New Zealand region compared to (b) a numerical classification using the same variables (source: Clark et al. 2011; coloured dots represent seamounts in different classes, which are independent in each classification).

3.5 Marine environment

The *Marine Environment Classification* (MEC) took five years to develop and complete, and its purpose was to provide a spatial framework for resource and conservation management by subdividing the geographic domain of the New Zealand EEZ into units having similar environmental and biological character (Snelder et al. 2005, later published in the scientific literature as Snelder et al. 2007). This numerical classification used multivariate analysis (a two-stage process; a non-hierarchical procedure followed by a hierarchical clustering approach) based on eight physical variables, which was "tuned" (a classification optimising process for selecting, weighting, and transforming variables) using biological data sets representing pelagic and benthic components of the biota (Chlorophyll-*a*, demersal fish, benthic invertebrates). The classification can be displayed at any number of class levels from 2 to 290. This allows users to choose a level of classification detail that is most suitable for particular applications and is consistent with the view that environmental management must occur at different levels of detail depending on the activities being considered. At the higher levels of classification, differences between classes mostly reflected variation in depth, water temperature, and solar radiation. Further divisions at the 4-class level approximately defined subtropical waters, the plateaus and subtropical front, and the sub-Antarctic waters. Subtropical waters were further subdivided at the 9-class level into bathyal and abyssal environments, and the plateaus and subtropical front waters into bathyal subtropical front, central continental shelf, and southern continental shelf environments. Similarly, the coastal environment was subdivided into three classes at the 9-class level: northern, central, and southern areas. The 20-class level further defined environmental groups that are mostly differentiated by variation in depth (Figure 3-4, Table 3-2).

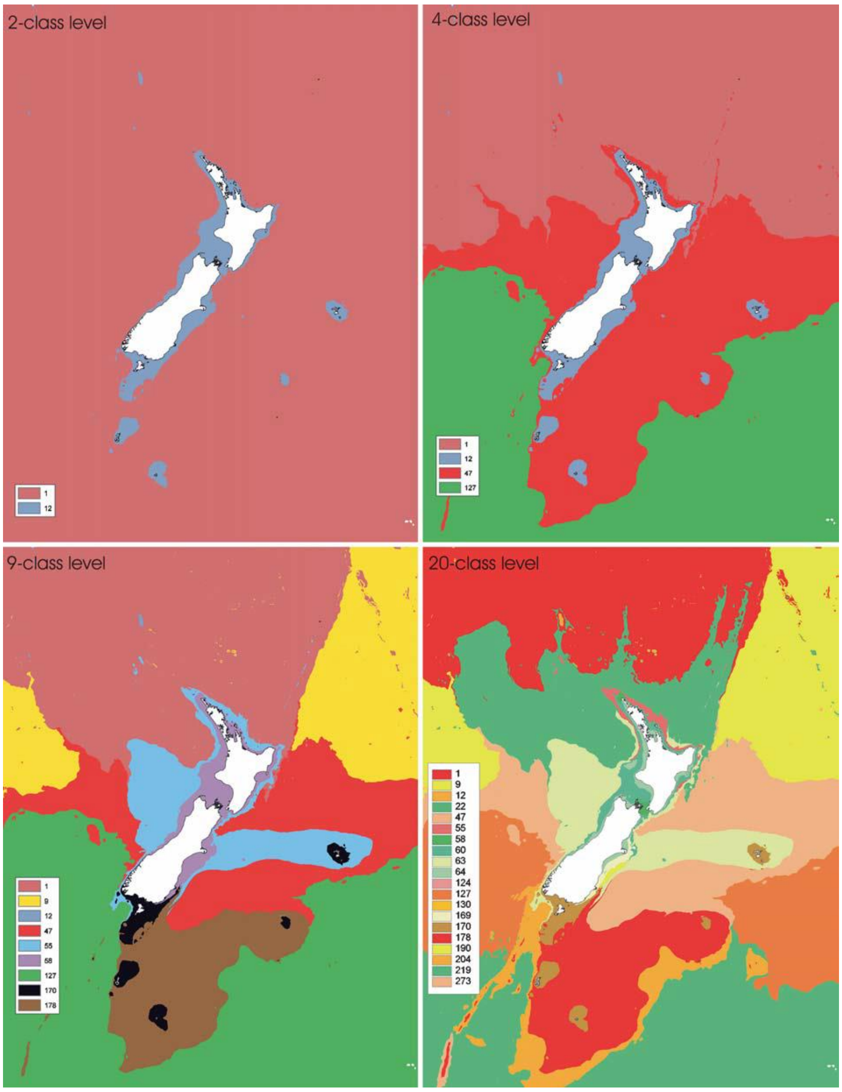


Figure 3-4: The Marine Environment Classification for the New Zealand region showing the location of classes at the 2-, 4-, 9- and 20-class level (source: Snelder et al. 2007). [Note: not all classes are visible at this map size]

Table 3-2: Average values of each of the eight defining environmental variables used in each class of the 20-class level of the Marine Environment Classification (source: Snelder et al. (2007).

Class	Area (km^2)	Depth (m)	Seabed slope (cm m^{-1})	Orbital velocity (ms^{-1})	Mean annual solar radiation (W m^{-2})	SST amplitude (°C)	SST gradient (°C km^{-1})	Wintertime SST (°C)	Tidal current (ms^{-1})	2-class level	4-class level	9-class level
1	88503	3001	1.4	0	17.5	2.3	0.01	19.5	0.06	Subtropical		Bathyal
22	53368	1879	1.5	0	15.4	2.4	0.01	16.3	0.11			
9	64306	5345	1.4	0	14.8	2.6	0.01	16.1	0.03			Abyssal
47	60053	2998	1.0	0	12.1	2.4	0.01	11.6	0.07	Oceanic	Plateaus and subtropical front	Central
55	2213	334	1.6	0	15.5	2.4	0.02	15.1	0.20			
63	26626	754	0.9	0	12.8	2.4	0.02	12.1	0.18			
178	39360	750	0.4	0	9.5	1.3	0.01	7.6	0.15			Southern
127	60884	4830	0.5	0	10.7	1.7	0.01	10.0	0.05	Sub-Antarctic		
204	18277	2044	3.0	0	9.2	0.9	0.01	8.0	0.08			
273	805	2550	9.1	0	8.4	1.4	0.03	4.4	0.05			
219	93982	4779	0.06	0	8.9	1.0	0.01	6.7	0.04			
12	149	94	0.9	0.10	17.8	2.3	0.01	19.3	0.30	Coastal		Northern
58	394	117	0.7	0.06	14.7	2.2	0.03	13.0	1.09			Central
60	4084	112	0.3	0.02	14.4	2.5	0.02	13.2	0.26			
64	2689	38	0.3	0.27	14.2	2.9	0.02	12.6	0.19			
124	68	8	0.4	0.84	13.4	2.3	0.02	12.7	0.00			
130	14	10	0.4	0.35	14.1	2.4	0.09	11.9	0.21			
169	932	66	0.2	0.11	12.4	2.7	0.04	9.9	0.21			
190	339	321	1.9	0.00	12.3	2.3	0.06	9.4	0.10			
170	5208	129	0.3	0.01	10.2	1.3	0.02	9.3	0.55			Southern

The classification strength of the MEC was assessed at a range of class levels. Strength values generally increased for each of the 'tuning' biological data sets as the classification detail was increased, indicating that finer levels of classification detail defined more biologically distinctive environments. However, the increase in the classification strength became more gradual once the number of classes exceeded approximately 75 (Figure 3-5) (Snelder et al. 2007). The difference in the performance (strength) of the classification with different biotic groups suggests the possibility of tailoring individual classifications to discriminate different ecosystem components, e.g., fish or invertebrates, pelagic or benthic. However, because they considered that a single integrated classification provides the best starting point for encouraging ecosystem-based management of marine biodiversity and resources, Snelder et al. (2007) recommended that more focused classifications should be discouraged unless the MEC was shown to be inadequate for particular applications.

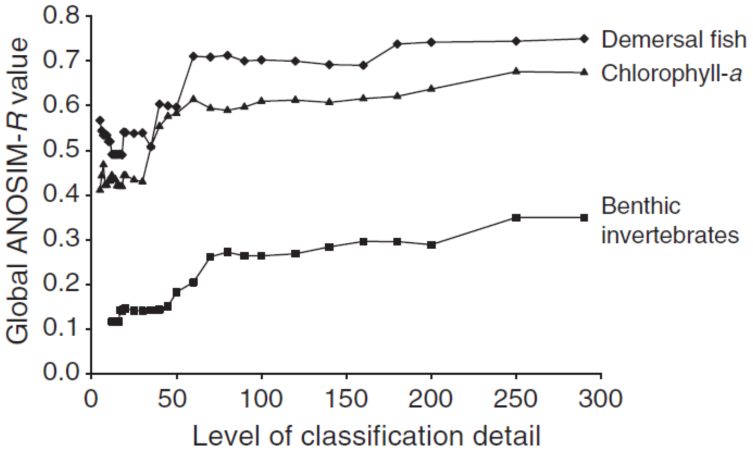


Figure 3-5: Results of the global ANOSIM analysis for the three biological data sets at varying levels of classification detail (note the general levelling-off of classification strength at >75 classes) (source: Snelder et al. 2007).

Having considered that the MEC was inadequate for assessing and managing the impacts of bottom trawling on benthic organisms, MPI commissioned the development of a *Benthic-Optimised Marine Environment Classification* (BOMEC) (Leathwick et al. 2012). The development of the BOMEC took advantage of a relatively new and more sophisticated numerical classification technique than used for the MEC; Generalised Dissimilarity Modelling (GDM). GDM was used to model the relationships between species turnover (dissimilarity among assemblages of 8 benthic taxonomic groups) and environmental variables (the same physical ones as used in the MEC plus additional benthic focused ones such as seabed sediments and seabed relief). The average fitted function results from the GDM analysis were used to transform the environmental variables, which were then classified (using non-hierarchical k-means clustering) to identify 300 groups, before these groups were further classified (using hierarchical agglomerative clustering) to 15 groups or classes (Leathwick et al. 2012). These 15 classes were strongly separated in relation to depth, comprising three inshore classes, three shelf classes, and nine classes in deeper waters of the continental slope and troughs (Figure 3-6). The identification of 15 classes was subjectively considered an appropriate classification level for use at a whole-of-EEZ scale, and was used to demonstrate the potential utility of BOMEC. However, the analytical approach means that such a classification can be also be used at other levels of detail, for example when higher levels of classification detail are required to discriminate habitat variation within areas of more limited extent (Leathwick et al. 2012). Leathwick et al. (2012) considered the BOMEC only *preliminary* and primarily a demonstration of the GDM-based approach. They suggested a range of improvements before the production of a final BOMEC, and also considered that it would be feasible to produce a higher resolution version for inshore waters (e.g., at 200 m resolution).

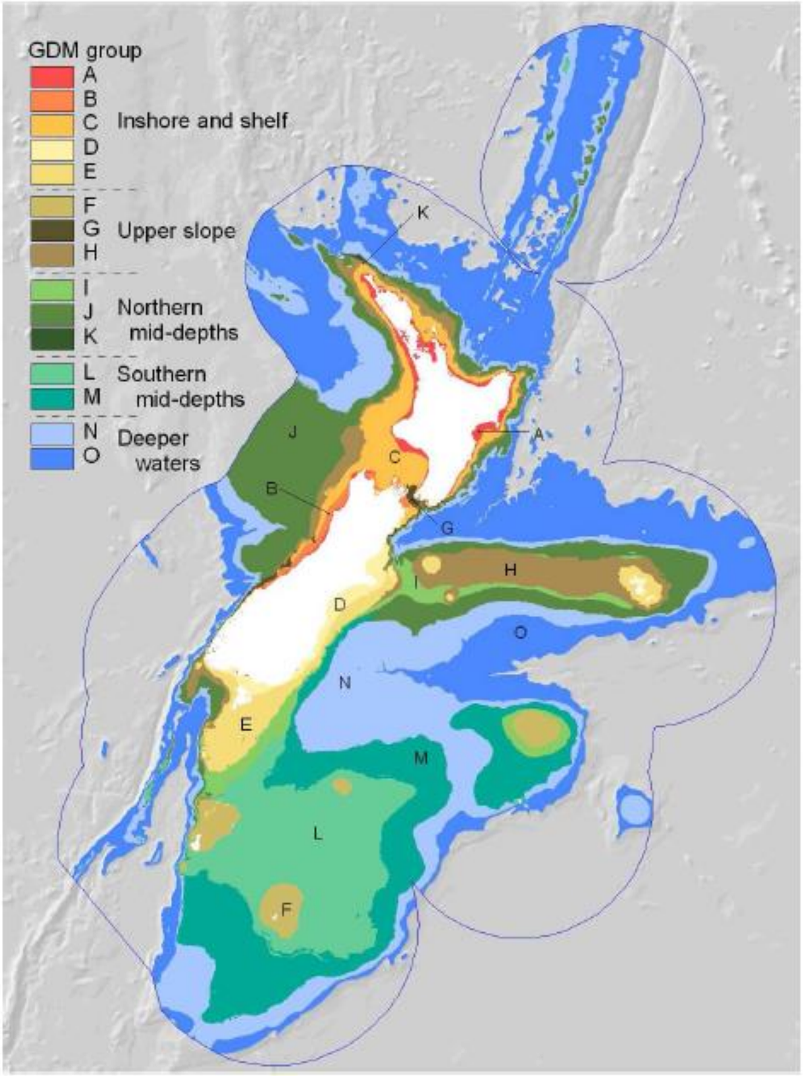


Figure 3-6: Distribution of a 15-class (GDM group) preliminary Benthic Optimised Marine Environment Classification in the New Zealand EEZ (source: Leathwick et al. 2012).

A demersal fish optimised MEC has also been produced for the New Zealand EEZ (Leathwick et al. 2006), but this is not considered here because it is a classification focused on a single ecological group of one major taxon (i.e., it doesn't qualify in the broad sense as a marine habitat classification).

3.6 Coastal environment

The *Interim Nearshore Marine Classification* (Walls 2006) built on existing biogeographical classifications (Moore 1949, Knox 1963, King et al. 1985), and identified and grouped marine biogeographic units based on biological, geological and oceanographic information. This classification divided the nearshore region (extending out to 12 nautical miles) into 8 biogeographic regions, with further divisions into smaller coastal and offshore island units (limited to 2 nautical miles) based on local expertise. Shelf units were also identified extending out to 200 m depth, and correspond to the shelf units defined by King et al. (1985). Shears et al. (2008) evaluated six biogeographic classifications focused on New Zealand coastal areas using macroalgal and invertebrate data (Moore 1949, Knox 1975, Nelson 1994, Walls 1995, Francis 1996, Apte and Gardner 2002), and developed a new classification identifying 11 bioregions within a northern and a southern biogeographic province. Concordance of these biogeographic classifications with population genetics has also been investigated, showing poor correspondence between biogeographic and phylogenetic boundaries (reviewed in Ross et al. 2009).

During the development of the broad-scale MEC for the EEZ, a regional-scale classification was also developed for a trial area (Snelder et al. 2005). The purpose of this regional classification was to assess the feasibility of producing higher resolution inshore or coastal classifications relevant to the more intensive management issues that frequently occur there. The Hauraki Gulf was selected as the trial area, and this numerical classification was made using the same analytical procedure as for the EEZ-scale MEC but at a finer spatial resolution (200 m compared to 1 km), using some additional environmental variables, and a more comprehensive biological data set for 'tuning' the classification. The classification covered the inshore Hauraki Gulf from the mean high-water line (but not including estuaries) to water depths of approximately 200 m on the continental shelf. The resulting classification differentiated areas within the gulf, mainly along a gradient from the inner to outer gulf that corresponded to differences in freshwater influence, depth-related factors and current speeds (Figure 3-7) (Snelder et al. 2005). Limited classification strength testing indicated that, at least for the fish tuning dataset, beyond the 5-class level the strength of the classification was relatively invariant (Snelder et al. 2005).

Snelder et al. (2005) noted that the MEC (including the regional-scale version) did not represent estuaries, despite including estuaries in the classification grid. They suggested that the results of the *Estuarine Environment Classification* (then still under development) could be used to replace estuarine grid cells in the MEC. Snelder et al. (2005) considered that the key limitation of the MEC at both the EEZ- and regional-scale was discriminating environmental character in coastal areas. Fundamental to this limitation was neither classification includes seabed substrates (e.g., mud, sand, rock etc.) as defining variables at the appropriate scale. Substrates vary at small spatial scales in the coastal area and are a specific cause of habitat heterogeneity. This means that classes in coastal areas are more likely to encompass significantly greater environmental and biological heterogeneity than classes in offshore areas.

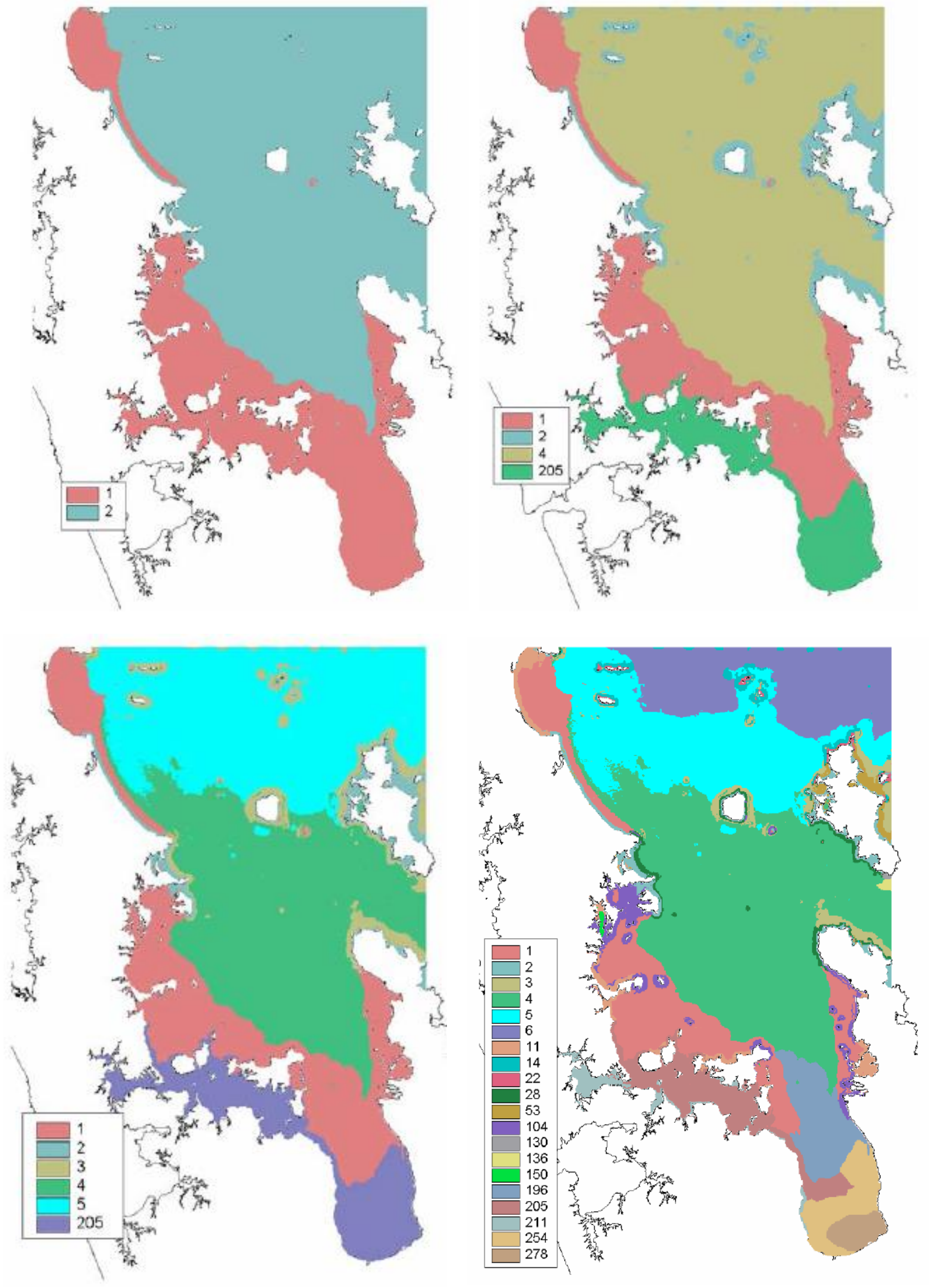


Figure 3-7: Regional-scale Marine Environment Classification for the Hauraki Gulf at the 2-, 4-, 6- and 20-class level (source: Snelder et al. 2005).

In part because of the limitations expressed by Snelder et al. (2005) in the regional-scale or coastal version of the MEC, and the fact that this version was never developed beyond the Hauraki Gulf, another type of coastal classification was developed subsequently. This *Coastal Classification* (described below) was brought together with a *Deepwater Classification* (also described below) to become the *Coastal and Marine Habitat and Ecosystem Classification* (CMHEC) (MFish and DOC 2008). The reasons given explicitly for using separate classifications within the CMHEC was that "the scale and nature of the information [for biota and habitats] available [for coastal compared to deepwater areas] necessitates a different approach to classification" (MFish and DOC 2008).

4 The Coastal and Marine Habitat and Ecosystem Classification

The CMHEC categorises the physical environment, operating under the assumption that different physical environments will contain different biological communities (MFish and DOC 2008). The classification is three-dimensional, taking into account of surface, water column and benthic features. The *Coastal Classification* component of the CMHEC is a five-level, multi-spatial scale thematic classification. The first level of the *Coastal Classification* is Biogeographic Region, identified using 14 coastal biogeographic regions based largely on the *Interim Nearshore Classification* (Walls 2006). The second level is Environment Type, which distinguishes between estuarine and marine areas, while the remaining three levels are used to classify the environment according to factors (depth, exposure, substrate) that are "thought to most strongly influence a site's biology" (MFish and DOC 2008). The number of sub-divisions within each of these levels varies depending on the preceding level type, with the fifth level (Habitat Type) having the most sub-divisions (up to 7). This coastal classification scheme can be used to identify up to 44 classes of habitat between mean high water and 200 m water depth (Table 4.1).

Table 4-1: Coastal Classification scheme (source: MFish and DOC 2008).

Level 1

Biogeographic region (14)

Level 2		Environment type	Estuarine		Marine						
Level 3		Depth	Intertidal	Subtidal	Intertidal (MHWAS – MLWS)			Shallow subtidal (MLWS – 30m)			Deep subtidal (30m – 200m)
Level 4		Exposure	Low	Low	Low	Med	High	Low	Med	High	Low
Level 5		Habitat type	Mud flat Sand beach Gravel beach Coble beach Boulder beach Rocky platform	Mud flat Sand flat Gravel field Coble field Boulder reef Rocky reef Biogenic reef	Mud flat	Sandy beach Gravel beach Cobble beach Boulder beach Rocky platform	Sandy beach Gravel beach Cobble beach Boulder beach Rocky platform	Shallow mud	Shallow sand Shallow gravel field Shallow cobble field Shallow boulder reef Shallow rocky reef Shallow biogenic reef	Shallow sand Shallow gravel field Shallow cobble field Shallow boulder reef Shallow rocky reef Shallow biogenic reef	Deep mud Deep sand Deep gravel field Deep cobble field Deep boulder field Deep rocky reef Deep biogenic reef

Beyond 200 m depth another thematic classification was constructed to identify marine habitats and ecosystems. This *Deepwater Classification* also has five-levels, and was designed to identify habitats and ecosystems at spatial scales from 10s to 1000s of kilometres. However, the structure of the classification is not directly comparable to the *Coastal Classification*. The first Biogeographic Range level is identified using the 20-level MEC (Snelder et al. 2007), which is considered to “provide a useful surrogate for ecological (biological and environmental) variation” at broad-scales. Within each MEC class, there are further divisions by Environment (benthic versus pelagic), Depth, Substrate and Habitat/Ecosystem. There are four and five depth sub-divisions within the Benthic and Pelagic environment sub-divisions, respectfully. The sub-divisions within the benthic substrate level are not specifically defined. The sub-divisions for the final Habitat/Ecosystem level of the classification are also not finite, and only examples are provided (Table 4.2).

Table 4-2: Deepwater Classification scheme (source: MFish and DOC 2008).

Large Scale				**Small Scale**
Biogeographic range	**Environment**	**Depth**	**Substrate**	**Habitat and ecosystem examples**
MEC	Benthic or seafloor	Upper continental slope (200-500m)	Represent the biologically-significant variation in substrate type	High-relief hard bottom or deepwater reefs Hydrothermal seeps and vents Seamounts and guyots Banks Submarine canyons Trenches Marine terraces Plains
		Mid continental slope (500-1000m)	Represent the biologically-significant variation in substrate type	
		Lower continental slope (1000-4000m)	Represent the biologically-significant variation in substrate type	
		Abyssal plain (>4000m)	Represent the biologically-significant variation in substrate type	
	Pelagic or water column	Sea surface (surface 0 m)	N/A	Eddies Mixed layers Upwellings Frontal boundaries Benthic boundary layers Stratified layers
		Epipelagic		
		Mesopelagic (200-1000m)		
		Bathypelagic (1000-4000m)		
		Abyssalpelagic (4000-7000m)		
		Hadalpelagic (>7000m)		

The report describing the CMHEC noted that the Deepwater Classification would require further development before it was required to inform MPA design in deepwater environments (MFish and DOC 2008). The report also made it clear that as additional data are gathered “gaps in the hierarchy will be filled and the classification will continue to grow” (MFish and DOC 2008). No such modification has yet taken place, but assessments of issues associated with the CMHEC have been undertaken because the classification has been applied to coastal MPA implementation processes.

5 Issues with the CMHEC

Prior to this review, the ISAG held a workshop (6 September 2016) in which issues with the CMHEC were discussed, identified and noted. The notes from that meeting, the results of a gap analysis (DOC and MFish 2011), discussions at a workshop that included ISAG representatives and the NIWA project team (14 June 2017), and a subsequent independent assessment by the NIWA team are summarised here to identify the main issues with the CMHEC.

Structural issues

The current CMHEC classification:

(1) Is comprised of two separately structured classifications. While two classifications were used in the CMHEC in response to differences in data availability, and differences in the drivers and scales of processes between coastal (0-200 m) and deepwater environments (>200 m), it is not straightforward to merge the results of the two classifications on a map. In some instances, water depths of >200 m occur within the 12-nautical mile limit of the territorial sea. Consequently, the identification and siting of MPAs at the boundary between the coverage of the two classifications is problematic in practice. Furthermore, habitats at the classification boundary (i.e., the shelf break) are likely to be particularly heterogeneous because the environment changes substantially at continental margins. Thus, it is likely that habitat heterogeneity in the cross-shelf region will be poorly or inconsistently represented. Some macro-habitats (e.g., canyons) often occur across the classification boundary zone. Thus, designing an MPA network that captures the connectivity that can occur between coastal and deepwater habitats is also going to be compromised by using two separate classifications.

(2) Does not represent the pelagic ecosystem in the *Coastal Classification* component. While the *Deepwater Classification* includes a high level benthic/pelagic environment division, the *Coastal Classification* does not. Classifications that include both pelagic and benthic components of the ecosystem are generally considered to be the most useful for marine spatial planning, and are the type of classification recommended by the Convention on Biological Diversity (CBD) (COP IX/20, CBD 2008).

(3) Does not adequately sub-divide some classification levels. Within the *Coastal Classification* component examples include: For the Depth classification level, the Shallow Sub-tidal category (0-30 m) is considered too broad. Biological habitats change significantly across this depth range, and the classification does not reflect this situation. For the Habitat Type classification level, the sub-divisions mud and sand are considered inadequate to represent habitat variation that occurs in soft sediment. In contrast, habitat variation within hard substrates, is represented by four sub-divisions (gravel, cobble, boulder, rocky reef/platform). There is also no sub-division that represents the potential ‘mixed’ character of a substrate. Mixed substrate habitats often support high levels of species diversity, and might be habitats that require specific protection. The *Deepwater Classification* also provides a depth-related example of not adequately sub-dividing some classification levels, where the Depth level

for the benthic environment is sub-divided by relatively conventional depth zones. However, these are not extended to include the hadal zone (6000 – 11,000 m), which is present in the New Zealand EEZ and has been shown to have a distinct fauna (Jamieson et al. 2012).

(4) Has non-defined sub-divisions within the Substrate level of the *Deepwater Classification*. While the substrate sub-divisions of the *Coastal Classification* are defined, for the *Deepwater Classification*, the classification notes only that that these sub-divisions should "represent the biologically-significant variation in substrate type". Not defining sub-divisions within a level is antithetical to the principles of a thematic classification. If sub-divisions are not defined it is possible to arrive at an almost infinite number of non-standard habitat classes. One of the strengths of a thematic classification is that they produce a relatively limited number of habitat classes that can be compared directly from location to location.

(5) Has sub-divisions within the Habitat/Ecosystem level of the *Deepwater Classification* that are not finite and vary in spatial scale. The sub-divisions of the final Habitat/Ecosystem level of the Deepwater Classification are offered as examples rather than a complete list of possible sub-divisions. Thus, there are other habitats/ecosystems that could be defined by the classification, but the user of the classification may or may not be aware of these features. Therefore, their inclusion or omission from the classification could be inconsistent across applications, and lead to potentially incomparable and incomplete identification of habitats. In addition, those example sub-divisions listed for the final Habitat/Ecosystem level, which are nested within the presumably broader expression of substrate variability (for the benthic environment only) (see above), relate to features that vary in spatial scale (compared to the presumably similar spatial scales of the sub-divisions within the final habitat level of the *Coastal Classification*). For example, the habitat/ecosystem sub-division of hydrothermal seeps and vents could represent sites at the scale of 10s of metres compared to the trenches habitat/ecosystem sub-division that will identify features that extend over 1000 km. The example habitats/ecosystems listed for the Pelagic Environment also encompass a range of spatial scales. This variation in sub-division scale occurs in contradiction of the labelling of the final Habitat/Ecosystem level as being "Small Scale".

(6) Uses sub-division categories for which there are limited national-level data.
Currently the *Coastal Classification*, includes Habitat Type sub-divisions defined by substrate (gravel, cobble, boulders, etc.) for which such raw data are limited. Furthermore, available data on substrates does not always comply with the definitions used in the classification (i.e., might not differentiate between cobble and boulder, or substrate information may be expressed by subdominant and dominant types). While coastal and oceanic sediment maps have been produced for the New Zealand region, these vary in format, and efforts to combine them have previously met with limited success (see comment in Leathwick et al. 2012). This situation means that the finest-scale level of this classification cannot currently be consistently applied at a national-scale (NB: new New Zealand regional substrate maps are currently under production by NIWA). Even bathymetric data are not currently complete enough to identify everywhere the 0-30 m Subtidal sub-division within the Depth level of the classification (e.g., around the Subantarctic Islands). While it is sensible to include sub-divisions in a thematic classification if there is a reasonable expectation that such data will become available in the future to identify them, it is important in the meantime that the classification is not applied inappropriately. That is, it is understood that the classification can only be applied consistently across the New Zealand region at a particular higher level until data for certain sub-classes becomes

available, or that it can only be applied in particular regions to the lowest classification level where data are available.

Performance issues

The current CMHEC classification:

(1) Does not allow for the identification of some known habitats. In addition to those habitats that are not captured by the current definition of biogenic habitat in the *Coastal Classification* (i.e., seagrass beds, etc., see above), and the depth sub-divisions used (i.e., trench habitats) in the *Deepwater Classification*, other habitats which are unlikely to be identified by the classification include deep habitats (> 200 m) found within the 12-nautical mile boundary, but are not included in the *Coastal Classification* (e.g., Jackson and Lundquist 2016).

(2) Includes a classification level that introduces known classification errors. The use of the Exposure level of the *Coastal Classification* has resulted in the similar classification of habitats that are known to be very different. For example, in the Hauraki Gulf region the classification identifies high current areas in a harbour as the same habitat as a highly exposed offshore island. This error results because the elements of wave exposure, tidal current and fetch, are not separately represented in the classification.

(3) Does not use a term for biogenic habitat that is consistent with the term used in the current MPA Implementation Guidelines, nor does the term used represent adequately the observed variation in biogenic habitats. For the *Coastal Classification*, biogenic habitat is defined as "biogenic reef", whereas the MPA Implementation Guidelines refer to the need for MPAs to protect "biogenic structures". The term biogenic structures is more encompassing, and extends beyond reefs made by elevated structures on the seabed constructed by living and dead organisms such as corals and sponges. For example, significant biogenic habitat can be made of the calcareous remains of shelled molluscs, worms, and bryozoans. Remarkably, the other examples of important biogenic habitat not included in the *Coastal Classification* are seagrass beds, mangroves, algal forests, and sponges. Furthermore, the biogenic structures made by even potentially reef-building organisms do not always occur as reefs – sometimes they can provide significant habitat as smaller patches of clumps or 'thickets', or even as solitary individuals/colonies. Finally, the definition specifies habitats elevated above the seafloor, thus excluding biogenic habitats that occur within the sediment (e.g., infaunal shellfish beds, crab burrows). These issues were significant enough for the mapping exercise conducted as part of gap analysis (DOC and MFish 2011) to redefine the term biogenic habitat. Biogenic habitat is not specifically included as a sub-division within the *Deepwater Classification* (see issues above).

(4) Does not include an ability to associate a measure of confidence/uncertainty with the identification of habitat classes in general, nor to map uncertainty at specific locations. When engaging with stakeholders in the process of identifying MPAs, it has proven important to be clear around limitations of the classification. Confidence can be assessed and expressed in multiple ways, both qualitative or quantitative, from those based on a simple analysis and visual summary of data quality and density, to statistical metrics that relate directly to underlying data models that have been used to derive the classification results. Uncertainty may have both a spatial and temporal component.

Inadequacy issues

The current CMHEC classification:

(1) Does not represent the nature or quality of the terrestrial environment immediately adjacent to the coast. Information about the terrestrial environment could be useful for defining coastal habitat that is relevant to identifying and protecting areas for species of particular importance (e.g., Species-specific Sanctuaries for seals that rely on haul-outs on land of a particular type), or modifying the character of the coastal habitat that impacts on its potential suitability as a site for a MPA (e.g., sediment and/or nutrient run off from intensively farmed land).

(2) Only considers estuaries as a single Environment Type. It is known that there are different types of estuaries, and that these are likely to represent different habitat at the scale of individual estuaries. Some habitat variation within an estuary is captured by the lower levels of the classification, but not biogenic habitat variability, which is particularly important in estuaries (see above).

(3) Uses inadequate biogeographic/bioregional schemes to define the highest classification level. This issue with the *Coastal Classification* is illustrated most obviously by the Subantarctic Islands. The fauna of the four Subantarctic Island groups are known to be different from each other (Freeman et al. 2011), as presumably is the habitat/ecosystem at a bioregional scale, yet these islands are classified as having the same range of habitats. Thus, the biogeographic regions for the coastal environment needs to be modified to at least reflect the known distinction between the Subantarctic Islands. The MEC at the 20-class level is used to define biogeography in the *Deepwater Classification*. The MEC, while a potentially useful proxy for biogeography across the New Zealand marine region (because temperature variables are among the primary drivers of the classification), is not *per se* a biogeographic scheme. Recently published biogeographic schemes that include the New Zealand region (e.g., O'Hara et al. 2012, Watling et al. 2013) generally divide the region into a relatively small number of biological provinces that correspond more closely to the 4-class level of the MEC.

(4) May be missing key drivers that influence the nature and distribution of coastal and marine habitat and ecosystems. Thematic classifications used in other countries include variables not incorporated in the CMHEC (see section 7). For example, those that relate to the geology of an area (e.g., the geological type of rock is known to affect the composition of biological communities that occur on rocky reefs), or the turbidity of the water (e.g., biological communities in coastal areas will be different in permanently turbid waters compared to non-turbid or temporally turbid waters). Such information is available for New Zealand, for example data layers that represent turbidity were developed for the MEC as well as for other variables (e.g., bottom currents), which can be key drivers of the distribution of fauna.

(5) Does not include the ability to capture fine-scale information about the communities that occupy the habitat classes. That is, there is no facility, where relevant information exists, to use habitat "modifiers". For example, a biodiversity modifier that incorporates information such as rarity, endism, diversity, and functional types. Such biodiversity information, as well as habitat type, is also potentially important when using the classification to identify representative sites for MPAs as part of a MPA network, or potentially assessing the sensitivity of habitats identified by the classification.

(6) Does not include the ability to capture information on habitat quality. While the CMHEC was not designed to include information on habitat/ecosystem quality, this is something that would be useful to include in a classification, potentially as another form of habitat modifier. That is, if the information is available it would be important to know when identifying representative and suitable sites for MPAs whether habitats are of relatively high quality or have been impacted. In addition, such information provides for the opportunity to identify MPAs that could include sites where there is a potential for habitat restoration and remediation e.g., reversal of ecological impacts caused by fishing or sedimentation.

Fundamental issues

The current CMHEC classification:

(1) Has not been tested widely to confirm the assumption that the present habitat classes are a good surrogate for biodiversity. There is a fundamental need to include a field verification step in the development of a robust classification, ideally before it is utilised for its intended purpose (Stevens and Connolly 2004). Such a ground-truthing also allows for quantitative measures of confidence in the classification results (see above). Currently the CMHEC is intended, as part of the MPA planning process, to identify habitats/ecosystems as surrogates for “biological pattern”. Other definitions of biodiversity may be adopted by future MPA-related legislation, which the habitats identified by the CMHEC or its successor will be expected to represent. For example, the New Zealand Resource Management Act defines “biological diversity” as “the variability among living organisms, and the ecological complexes of which they are a part, including diversity within species, between species, and of ecosystems”. This definition is similar to that used by the Convention of Biological Diversity (CBD). While the CMHEC has not been formally tested, indications are that the *Coastal Classification* is not a good expression of biodiversity. Jackson and Lundquist (2016) used a modified version of the *Coastal Classification* (following the recommendations for classifying biogenic habitat in DOC and MFish 2011) to undertake an analysis that suggested that variation in demersal fish biodiversity was poorly correlated with the lowest class level of the classification in the Hauraki Gulf. Another New Zealand thematic classification, the *Estuarine Environment Classification* (identified principally by geomorphic variables), has also been found to not be a good predictor of biodiversity. Lundquist et al. (2003) found estuary type was not well correlated with macroinvertebrate biodiversity patterns in Auckland estuaries, and Francis et al. (2011) found that this classification also provided little explanatory power for assemblage patterns in small estuarine fish across 68 New Zealand estuaries.

(2) Does not sit within a structured and governed process that allows modifying it in light of additional data or information. There is a recognised need for classifications to be flexible rather than fixed tools. For example: in response to predictable or known changes in the environment (e.g., events related to climate change); when new data become available that require modifications of the classification data content and/or structure; and/or when scientific advances mean that improved methodologies are available to construct a more robust classification procedure. Currently, data and information are being gathered through various agencies and funding streams that are not being used to improve the classification as a whole (although some local undocumented regional modifications have occurred). Any modifications of the classification *structure* should occur within a structured and nationally-governed process. Also, any results of the *application* of the classification should be flexible to modification during the planning phase based on additional information from stakeholders.

6 Fit-for-purpose classification

It is clear from the above that there are significant issues with the CMHEC which would benefit from improvement. Before considering what modifications or substitutes could be made or adopted, we here first consider in general terms what attributes the fit-for-purpose classification should possess, before highlighting a few specific characteristics it should ideally incorporate.

Although final decisions relating to the MPA policy reform have not been made, in order to have the ability to inform the development of a representative network of MPAs across New Zealand's marine environment, a relevant classification will need to:

- be able to identify at suitable scales the full range of New Zealand's coastal and marine habitats/ecosystems (as a surrogate for biodiversity), including those in the EEZ;
- be relevant for the full range of potential future categories of MPAs, such as those focussed on particular species;
- be mappable at both regional and national scales (i.e., at relevant conservation and planning scales);
- be able to account for variability in data availability and quality across the New Zealand region;
- be flexible to modification during MPA planning process (based on additional information);
- be easily interpreted and understood by the general public.

Specifically, and ideally, the classification should:

- include relevant information about the adjacent terrestrial and river environments – potentially as a habitat "quality modifier" (e.g., land use, river sediment loads) or be directly relatable to the Terrestrial and Freshwater Classifications;
- be a single whole-ecosystem classification (i.e., pelagic and benthic, and coastal and deepwater);
- represent the latest biogeographic information for the region;
- recognise different estuary types;
- recognise different biogenic habitats;
- recognise currently 'missing' habitats (e.g., vegetated habitats such as seagrass beds, mangroves, macro-algal beds; and non-vegetated biogenic habitats such as infaunal shellfish beds, non-calcareous tube worm mats);
- include 'missing' key environmental drivers (e.g., turbidity in coastal areas);
- be able to capture fine-scale variability in habitats (particularly in coastal areas where the use of the classification results are likely to be used to identify MPAs at relatively small geographic scales) while balancing the loss of confidence in capturing this variability when data are sparse;

- be capable of aligning with habitat descriptors used in local site surveys;
- be able to include relevant information about species or community attributes of habitats/ecosystems – potentially as habitat "biodiversity modifiers" (e.g., species rarity, endemism, diversity, functional types);
- be potentially linkable to ecosystem goods and services provided by habitats;
- align with the identification of features that may be unique, vulnerable, or special (and that may not be adequately captured in a broad-scale classification system; e.g., VMEs, EBSAs, key ecological areas etc);
- be able to be used in a predictive capacity (i.e., to understand influence of climate change on habitats/ecosystems, and therefore the future usefulness of siting representative MPAs);
- provide the ability to incorporate measures of uncertainty/confidence (at a range of scales; at overall classification level and for each class, and also when results mapped to locations);
- be linked directly to freely available and documented sources of data/databases/expert opinion so that it can be understood and updated easily
- sit within a structured and governed process for its future modification.

There are innumerable classifications that have been developed for coastal and marine habitats and ecosystems, however many of these will not satisfy the fit-for-purpose criteria for a New Zealand classification (as described above). That is, in the main, most of the existing classifications are for one type of environment (e.g., coastal or deepwater, benthic or pelagic). Below we review only those international broad-scale classifications (but not global schemes) that could potential satisfy a fit-for-purpose classification for New Zealand (with some modification), have undergone formal testing or some other sort of evaluation, and have been used to inform the design of MPA networks. While there are a number of thematic classifications (which we review by country/region) that satisfy these review criteria, numerical classifications of coastal and marine environments are less common – so we have relaxed these criteria for this type of classification in order to continue to provide a contrast of approach (after reviewing regionally relevant examples, we consider others all together to demonstrate the range of available analytical methodologies).

7 Thematic classifications

7.1 Canada

Roff and Taylor (2000) described a thematic classification approach, based on geophysical variables, for marine conservation purposes in Canada. This national framework was based on an earlier test classification developed for British Columbia (Pacific coast of Canada) (Zacharias et al. 1998). The classification has six levels representing the different scales of influence that environmental variables have upon the provision of habitat for biotic communities. The first level essentially represents biogeography, and at the second level the classification splits to separately categorise the pelagic and benthic environments. The third level is Depth/Light for both types of classification, while the fourth level is represented by Substrate Type and Stratification/Mixing regime for the benthic and pelagic environments, respectively. The fifth level for the benthic classification is Exposure/Slope, while there is no fifth level for the pelagic component of the classification. Overall, the Canadian marine classification aims to identify habitats from the intertidal, but does not purposefully include estuaries. Roff et al. (2003) considered that the latter environments should be represented in a separate classification. The classification scheme was applied to the entire Canadian coastline, and to an area of the continental shelf off Nova Scotia (Atlantic coast of Canada) to illustrate its utility at national and regional scales (Roff et al. 2003). For this application Roff et al. (2003) modified the structure of the original framework (Table 7-1).

For mapping purposes, the results from the benthic and pelagic classifications were brought together to map "seascapes" using a GIS overlay approach based on Boolean logic (Figure 7-1).

Table 7-1: Hierarchical classification of geophysical factors used to define habitat types on the Canadian shelf (source: Roff et al. 2003).

Level 1	Level 2	Level 3	Level 4	Level 5	Level 6	Level 6b	Level 7	Level 8
Environment type	Geographic range	Temperature	Sea-ice cover	Segregation of benthic and pelagic realm	Vertical segregation	Benthic temperature	Mixing and wave action	Benthic substrate
Marine	Atlantic	Boreal (avg. temp >0°C)	Not applied	Pelagic	Pelagic		Pelagic stratification	
		Temperate (avg. temp >0°C)			Epipelagic (0-200m)		Stratified (*da* >1000)	
					Mesopelagic (200-1000m)		Non-stratified (*da* <100)	
		Subtropical (avg. temp >6°C in winter, >18°C in summer)			Bathypelagic (1000-2000m)		Frontal (100 <*da* <1000)	
					Abyssal/hadal (>2000m)			
				Benthic	Benthic		Benthic-exposure	Benthic sediments
					Euphotic (0-50m)	Cold subarctic (<6°C)	Exposure (depth <50m)	Mud
					Dysphotic/aphotic (50-200m)	Moderate temperature (6-9°C)	Benthic -slope	Mostly sand (20-80% sand)
					Bathyal (200-2000m)		High slope (slope >20%)	Partially sand (0-20% sand)
					Abyssal/hadal (>2000m)	Warm gulf stream (>9°C)	Low slope (slope <2%)	Partially gravel (5-50% gravel)
								Mostly gravel (>50% gravel)

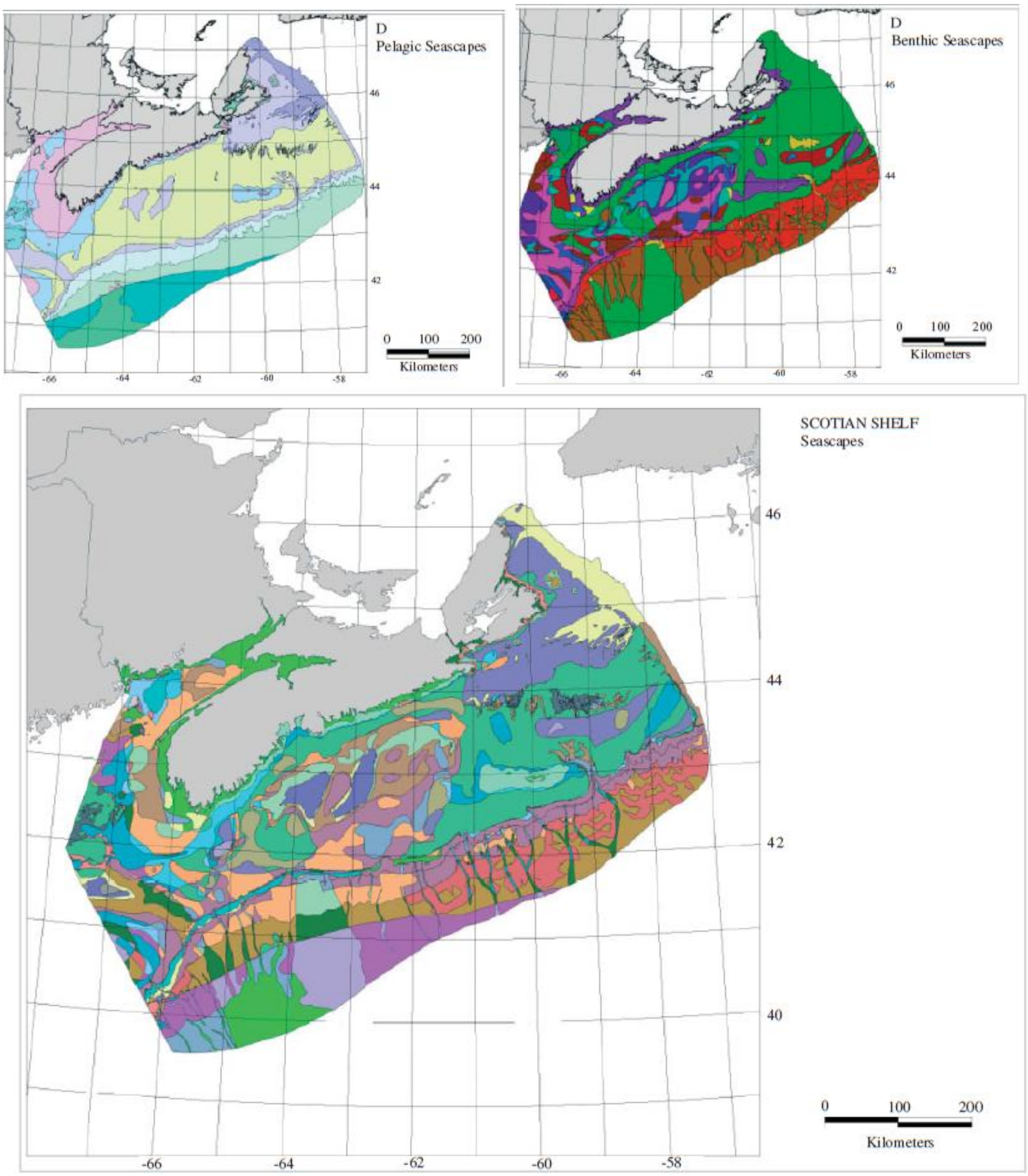


Figure 7-1: (a) Pelagic and (b) Benthic seascapes, and (c) the combined seascapes of the Scotian Shelf based on overlay of distributions of geophysical factors for the pelagic and benthic seascapes (source: Roff et al. 2003).

From the literature reviewed, it is unclear to what extent this classification has been used for designing MPA networks, nor if the effectiveness of the classification has been assessed formally. Nonetheless, the scheme is included here because it is relatively early 'modern era' example of a broad-scale classification.

7.2 Europe

The European Union Nature Information System (EUNIS) includes separate thematic classifications that describe habitats for terrestrial (9 classifications), coastal and marine environments (Davis et al. 2004). The EUNIS Coastal Classification refers to habitats that are generally above the high-water mark (e.g., dunes), but it also includes the littoral zone. Habitats below the high-water mark are also included in the Marine Classification component. The marine component of EUNIS was based on the Marine Habitat Classification for Britain and Ireland (Connor et al., 2004). The Marine Classification was initially developed to four levels (Davis et al. 2004), which in 2007 was extended to six levels (with Marine being Level 1, https://www.eea.europa.eu/themes/biodiversity/eunis/eunis-habitat-classification/habitats). At level 2, there are eight categories – with Sublittoral sediments, the Deep-sea bed, and Pelagic water column each represented by a single category, while habitats above the low water mark are represented by 4 categories. Variables that are used to define and sub-divide Level 3, most of which are physico-chemical factors (e.g., substratum, topography), vary among the categories identified at Level 2. Levels 4, 5 and 6 are mostly biological in nature, but again vary depending on the nature of the preceding level. Because of these differences and inconsistencies among levels, the EUNIS marine classification is difficult to illustrate clearly in a single table or figure. Table 7-2 is used here to represent the Deep-sea bed section of the Marine Classification because this part of the classification is considered below as an illustration of the issues that have been identified with EUNIS.

Table 7-2: Deep-sea bed section of the EUNIS habitat classification (source: Howell 2010).

Level 1	Level 2	Level 3	Level 4	Level 5	Level 6
A: Marine	A6: Deep-sea bed	A6.1: Deep-sea rock and artificial hard substrata	A6.11: Deep-sea bedrock		
			A6.12: Deep-sea artificial hard substrata		
			A6.13: Deep-sea manganese nodules		
			A6.14: Boulders on deep-sea bed		
		A6.2: deep-sea mixed substrata	A6.21: Deep-sea lag deposits		
			A6.22: deep-sea biogenic gravels (shells, coral debris)		
			A6.23: Deep-sea calcareous pavements		
			A6:24: Communities of allochthonous material	A6.241: Communities of macrophyte debris	
		A6.3: Deep-sea sand	A6.31: Communities of bathyal detritric sands with *Grypheus vitreus*		
		A6.4: Deep-sea muddy sand			
		A6.5: Deep-sea mud	A6.51: Mediterranean communities of bathyal muds	A6.511: Facies of sandy muds with *Thenea muricata*	

Level 1	Level 2	Level 3	Level 4	Level 5	Level 6
				A6.512: Facies of fluids muds with *Brissopsis lyrifera*	
				A6.513: Facies of soft muds with *Funiculina quadrangularis* and *Apporhais seressianus*	
				A6.514: Facies of compact muds with *Isidella elongata*	
			A6.52: Communities of abyssal muds		
		A6.6: Deep-sea bioherms	A6.61: Communities of deep-sea corals	A6.611: Deep-sea *Lophelia pertusa* reefs	
			A6.62: Deep-sea sponge aggregations	A6.621: Facies with *Pheronema grayi*	
		A6.7: Raised features of the deep-sea bed	A6.71: Permanently submerged flanks of oceanic islands		
			A6.72: Seamounts, knolls and banks	A6.721: Summit communities of seamount, knoll or bank within euphotic zone	
				A6.722: Summit communities of seamount, knoll or bank within mesopelagic zone, i.e., interacting with diurnally migrating plankton	
				A6.723: Deep summit communities of seamount, knoll or bank (i.e., below mesopelagic zone)	
				A6.724: Flanks of seamount, knoll or bank	
				A6.725: Base of seamount, knoll or bank	A6.7251: Moat around base of seamount, knoll or bank
			A6.73: Oceanic ridges	A6.731: Communities of ridge flanks	
				A6.732: Communities of ridge axial trough (i.e., non-vent fauna)	
				A6.733: Oceanic ridge without hydrothermal effects	
			A6.74: Abyssal hills		
			A6.75: Carbonate mounds		
		A6.8: Deep-sea trenches and canyons, channels, slope failures and slumps on the continental slope	A6.81: Canyons, channels, slope failures and slumps on continental slope	A6.811: Active downslope channels	
				A6.812: Inactive downslope channels	
				A6.813: Along slope channels	

Level 1	Level 2	Level 3	Level 4	Level 5	Level 6
				A6.814: Turbidites and fans	
			A6.82: Deep-sea trenches		
		A6.9: Vents, seeps, hypoxic and anoxic habitats of the deep sea	A6.91: Deep-sea reducing habitats	A6.911: Seeps in the deep-sea bed	6.9111: Cold seep benthic communities of hadal zone
				A6.912: Gas hydrates in deep sea	
				A6.913: Cetacean and other carcasses on the deep-sea bed	
			A6.92: Deep-sea bed influenced by hypoxic water column		
			A6.93: Isolated 'oceanic' features influenced by hypoxic water column		
			A6.94: Vents in the deep sea	A6.941: Active vent field	
				A6.942: Inactive vent fields	

The EUNIS classification is aligned with a database containing information for using the classification, and it has been used widely for mapping marine habitats, and to produce predictive maps of habitat distribution for both research and practical applications. For example, application of the EUNIS classification at Level 3 was used to identify a network of Marine Conservation Zones in shallow waters off the United Kingdom (JNCC 2012).

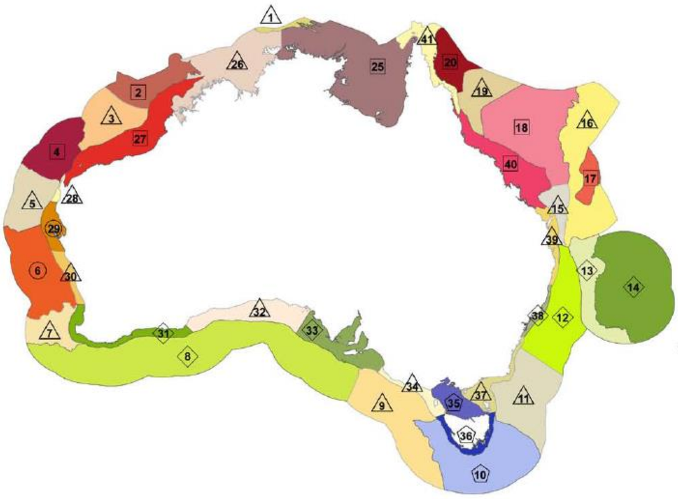
However, Howell (2010) believed that that EUNIS classification scheme would not be fit-for-purpose to implement MPA networks in the deep NE Atlantic, because the deep-sea section contained fundamental flaws. The identified flaws included: the lack of a high-level depth-related division in the classification, and a mixed geomorphological and substratum level - which she considered should have logically been split, and with geomorphology placed at a higher level than substratum. Overall, the inconsistencies in the levels at which divisions occur and the lack of appropriate division within the hierarchy were likely to cause major problems from a mapping and MPA design perspective (Howell 2010). Because of these issues, Howell (2010) developed a four-level thematic classification system that she thought better suited for the deep-sea environment. This system was arrived at after first reviewing the usefulness of physical surrogates commonly used in deepwater classifications. Howell (2010) was particularly critical of the use of geomorphology in deep-sea classifications, because biotic communities associated with features such as seamounts, canyons, banks, plateaux, ridges etc, are not always distinct from one another. For this reason, Howell (2010) chose to omit geomorphology as a level in her proposed scheme (which contained only Biogeography, Depth, Substrate, and Biology).

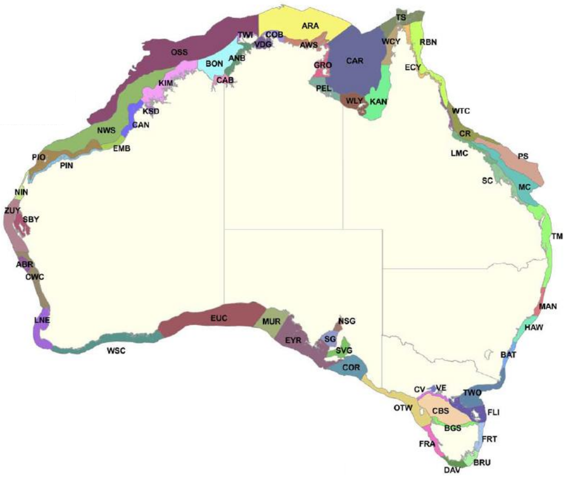
The widespread application of the EUNIS classification has indeed provided examples of a number of problems with the marine section of the EUNIS classification, and it became evident that further development and revision was necessary. Galparsoro et al. (2012) report on an extensive and comprehensive review of the EUNIS Marine Classification, based on a workshop that was attended by 95 users of the classification. The aim of the workshop was to identify the issues with the classification, how it could be improved, and to devise a process to further develop the marine component of the EUNIS habitat classification. The review identified numerous issues with the EUNIS Marine Classification (see Galparsoro et al. 2012 for detail). In general terms these related to: structure and hierarchy (e.g., inclusion of littoral and supralittoral habitats in both the Marine and Coastal classifications, inconsistence application of factors at different levels); the relationship between lower biological levels and upper abiotic levels (e.g., the same communities were sometimes related to different parts of the abiotic part of the classification); terminology (e.g., inconsistent application of terms such as habitat, biotope, biocenosis, and peuplement); mapping and modelling (e.g., mismatch between data that can be obtained from remote-sensing mapping or habitat suitability modelling and the habitat classes); and proposed future developments (e.g., ability to remove the lowest biological levels and allow these to be defined at a regional level, include new habitat classes, or include 'goods and services' description). Galparsoro et al. (2012) concluded their review by noting the importance of setting up a clear process (including mechanisms and timing) to develop and revise the EUNIS classification. This process would require input from scientists and government science/policy managers, who would all need defined roles in order to ensure the continuity of the classification and versions based on newly available data. A significant European-wide update of EUNIS was expected to be completed by 2014, however the latest revision (2012, https://www.eea.europa.eu/themes/biodiversity/eunis/eunis-habitat-classification) does not appear to have addressed many of the issues raised by Galparasoro et al. (2012).

7.3 Australia

A plethora of marine classifications have been developed for Australia, most notably for coastal areas and designed to operate at relatively fine-spatial scales, and often specific to particular States (e.g., for Victoria), or habitat types (e.g., coral reefs) (see review by Ball et al. 2006). Broad-scale classifications in Australia have therefore avoided classifying State waters (i.e. the intertidal and out to 3 nautical miles), and the lowest level of habitat seen in other similar classifications elsewhere (see above and below). As such they have been termed 'bioregionalisations' rather than habitat or ecosystem classifications. Australia first developed an Interim Marine and Coastal Regionalisation of Australia (IMCRA; IMCRA Technical Group 1998) that covered inshore waters to 200 m water depth (i.e., shelf break). This classification was followed by a National Benthic Marine Bioregionalisation (Heap et al. 2005) and a Pelagic Regionalisation (Lyne and Hayes 2005), which extended coverage offshore from the shelf break to the deepest waters of the Australian EEZ. These two offshore classifications were combined to produce the National Marine Bioregionalisation of Australia (NMBA; Department of the Environment and Heritage 2005). Finally, the IMCRA and the NMBA were combined to produce the Integrated Marine and Coastal Regionalisation of Australia (which confusingly also has the acronym IMCRA; Commonwealth of Australia 2006). The IMCRA consists of separate benthic and pelagic bioregionalisations. The benthic bioregionalisation incorporates three separate layers of information: (1) Provincial bioregions that reflect biogeographic patterns identified by an analysis of bottom-dwelling fish distributions (24 provinces and 17 transitions); (2) Meso-scale regions on the continental shelf using regional biological and physical information, and geographic distance along the coast (to produce 60 meso-scale regions); and (3) Geomorphic units based on a cluster analysis that identified features with similar seafloor geomorphology (14 categories) (Figure 7-2). The pelagic bioregionalisation is divided into two components: (1) Continental shelf which is based on a classification of pelagic fish species diversity and richness (4 bioregions); and (2) Offshore which is divided into water masses based on an analysis of physical properties (25 water masses), to which is added information on sea surface circulation patterns and energetics at regional scales.

The IMCRA is aligned and associated with: databases for using the bioregionalisation; a set of stated ongoing research needs (e.g., links between benthic and pelagic systems, testing assumptions of surrogacy); methodological development needs (e.g., analytical methods for large and complex datasets and techniques for visualisation of complex three and four-dimensional models of the ocean); and an explicit process for updating and revising the bioregionalisation, complete with identified responsibilities of the actors (Commonwealth of Australia 2006).





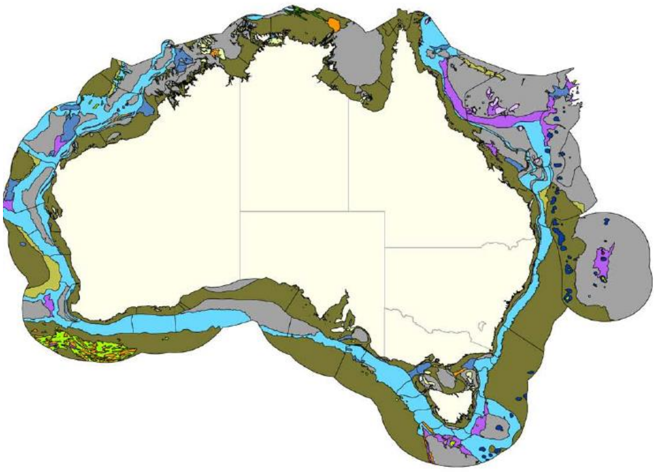


Figure 7-2: (a) Provincial Bioregions, (b) Meso-scale Bioregions, and (c) Geomorphic Units of the Integrated Marine and Coastal Regionalisation of Australia (IMCRA) (Commonwealth of Australia 2006).

The first MPA network under Australia's National Representative System of Marine Protected Areas was formally declared in July 2007. The design of this MPA network, comprising of 13 Commonwealth Marine Reserves in the southeast region of the Australian EEZ, was based on the IMCRA. Concerns about the use of geomorphology to define the bioregionalisation, led Williams et al. (2009) to investigate (using video survey data for mega-epifauna and substrate habitat) the degree to which geomorphic features (e.g., seamounts, canyons) act as surrogates for biodiversity. This study found that the IMCRA design process placed insufficient emphasis on the hierarchical relationship between depth-related faunal distributions and geomorphic features, and failed to identify some areas of differing biodiversity. This issue led to false within-class homogeneity for geomorphic features, and indicated that (along with depth) size, complexity, configuration, and anthropogenic impact need to be used as 'modifiers' to allow geomorphic features to be useful surrogates for biodiversity. Williams et al. (2009) observed that as a consequence of using unmodified geomorphic surrogates, and not nesting geomorphic features within depth, there was less recognition of the importance and comparative rarity of megafaunal biodiversity on the continental margin (<1500 m water depth). Considering the inherent problem of using untested abiotic surrogates, Williams et al. (2009) concluded that rather than modifying the IMCRA, it would be better to identify representative MPAs using bottom-up approaches such as modelling the relationship between abiotic variables and the distribution of biodiversity (i.e., numerical-based classifications).

Huang et al. (2011) proposed an alternative benthic-focused classification for the Australian EEZ that included a spatial data analysis method. Their approach consisted of a thematic classification, coupled with unsupervised object segmentation technique and a supervised fuzzy classification to implement the classification. The three-level classification (Light penetration; Nutrients/Oxygen/Temperature; Substrate/Local topography/Exposure) were defined using up to 5 variables, and sub-divided by up to 5 categories (Table 7-3). The segmentation process identified relatively homogeneous areas (objects) of seabed based on variable values, while the fuzzy classification assigned each object a habitat class (Figure 7-3). The fuzzy classification recognises that transitions between habitats are usually gradual, and provides a measure of confidence in the classification that allows for testing of the reliability of the results. Haung et al. (2011) also note that their approach uses datasets often or increasingly available over large areas of seabed in many nations, and employs widely available software.

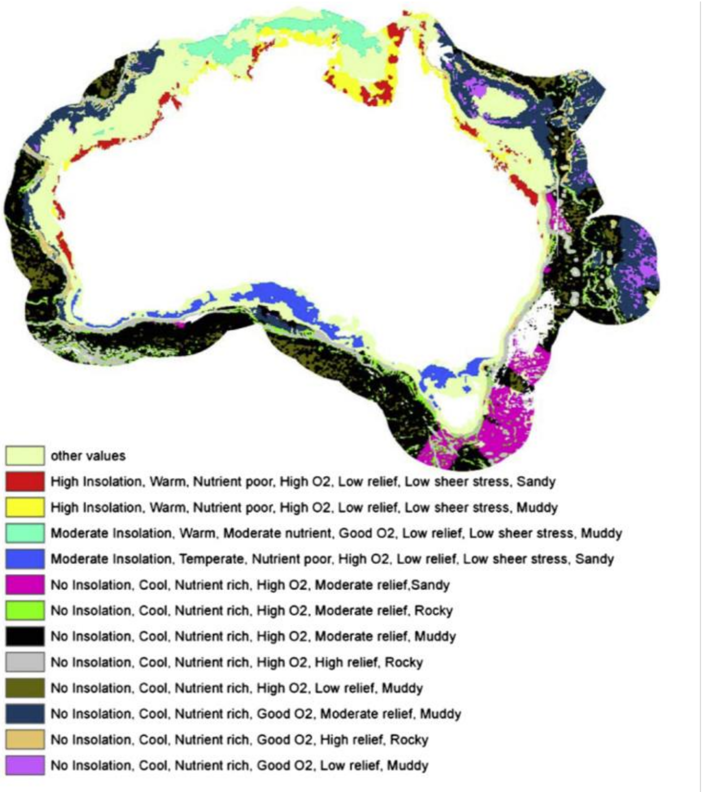
Table 7-3: Categories of the thematic classification for Australian EEZ (Source: Huang et al. (2011)).

Level 1		
Light	**Depth range (m)**	**Light penetration zone**
High insolation	0-50	
Moderate insolation	50-120	Euphotic
Poor insolation	120-200	
Minimum insolation	200-1000	Dysphotic
No insolation	>1000	Aphotic

Level 2	
Nutrients	**Combination rule**
Nutrient rich	NO_3 high or PO_4 high
Nutrient moderate	(NO_3 poor and PO_4 moderate) or (NO_3 moderate and PO_4 poor) or (NO_3 moderate and PO_4 moderate)
Nutrient poor	NO_3 poor and PO_4 poor
Dissolved oxygen	**O_2 content (ml/L)**
High O_2	>4
Good O_2	2.8-4
Moderate O_2	1.68-2.8
Bottom temperature	**Temperature (°C)**
Warm	>20
Temperature	10-20
Cool	<10

Level 3			
Substrate	**% Mud**	**% Gravel**	**Local relief (m)**
Muddy	>15.5		
Sandy	<15.5		
Gravelly		>65	
Rocky			>350

Relief	**Local relief (m)**
High relief	>500
Moderate relief	50-500
Low relief	<50
Seabed sheer stress	**% Exceedance**
High	>50
Moderate	15-50
Low	<15



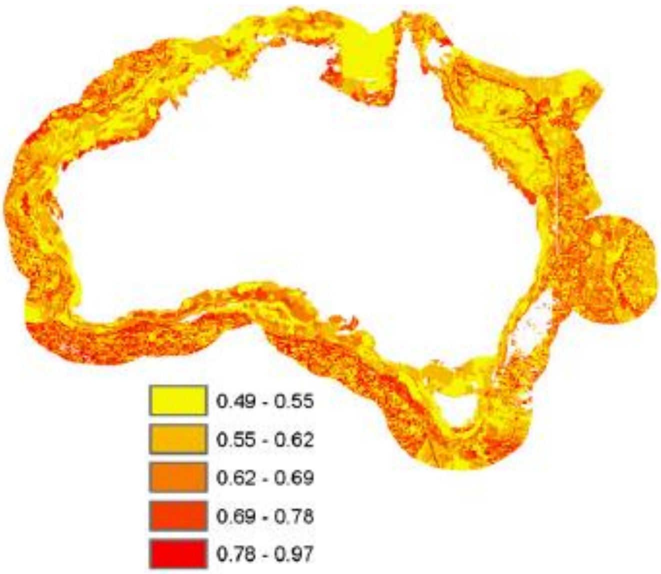


Figure 7-3: (a) Combined classification of benthic habitats of the Australian EEZ, and (b) fuzzy class membership of the classes (i.e. higher values indicate a higher confidence that class is correctly assigned) (source: Haung et al. 2011).

7.4 Southern Ocean

In support of efforts by the Convention on the Conservation of Antarctic Marine Living Resources (CCAMLR) to establish a representative system of MPAs in the Southern Ocean, Douglass et al. (2014) produced a hierarchical classification scheme of ecoregions, bathomes, and geomorphic features to identify "environmental types" for benthic fauna (Figure 7-4). Ecoregions were identified according to environmental drivers (e.g., temperature, ice concentration) and dispersal barriers for benthic organisms. Bathomes or broad-scale depth classes were based on depths at which rapid transitions in benthic assemblage composition were expected (using available species-depth studies). Geomorphic features were identified by modifying an existing Antarctic-wide seafloor geomorphology scheme. Analysis of available data to describe these components of the ecosystem resulted in the identification of 23 benthic ecoregions, 9 bathomes, and 28 geomorphic feature types. Combined within the classification framework, 562 environmental types were identified (Figure 7-5).

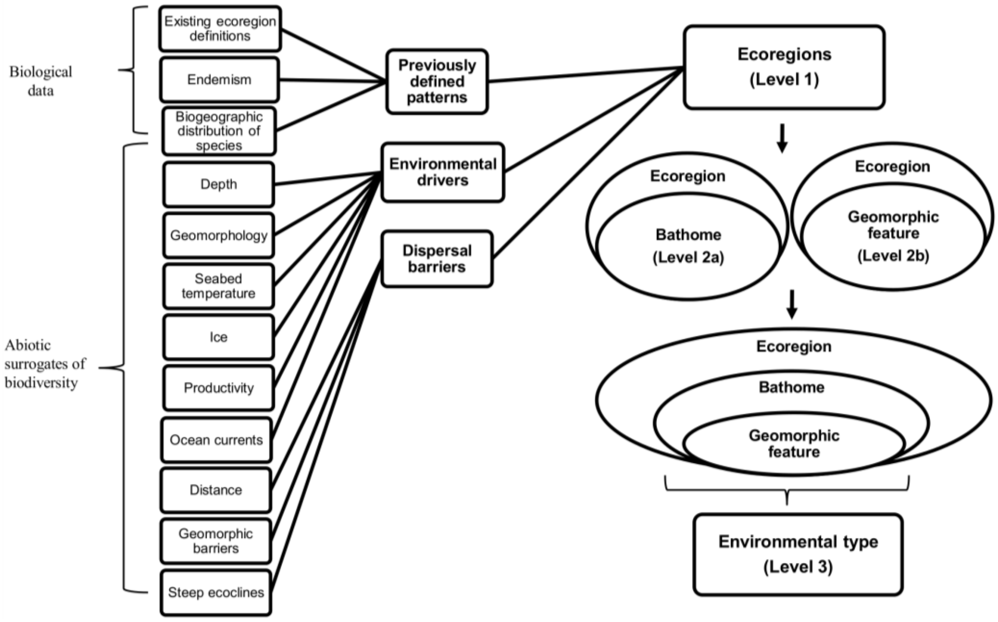


Figure 7-4: **Framework used to classify ecosystems in the Southern Ocean (source: Douglass et al. 2014).**

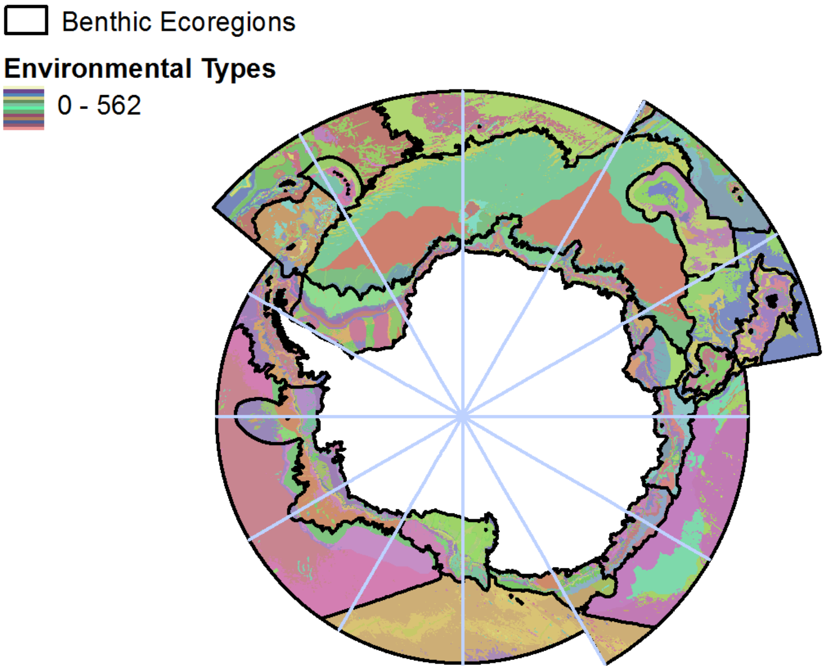


Figure 7-5: **Distribution of Environmental types in the Southern Ocean (within Ecoregions) (source: Douglass et al. 2014).**

Douglass et al. (2014) used the results of their classification analysis to identify the representation of environmental types within existing MPAs, and to suggest further locations for MPAs. They found that representativeness in general was low, for example, none of the 23 ecoregions have a system of MPAs that is representative of the diversity of benthic environmental types they contain. There were also over a hundred environmental types which were rare (restricted distribution), and largely lacking in any form of existing or proposed protection (e.g. seamounts with shallow summits). Douglass et al. (2014) considered that their thematic classification scheme was an improvement on previous numerical classifications (which they didn't identify, but see description of classifications by Grant et al. 2006 and Sharp et al. (2010) in section below). They suggested that expert-driven thematic classifications based on known and probable relationships between the variables they chose would be better than numerical classifications based simply on statistical measurements of similarity/dissimilarity in the biota/environment between areas (Douglass et al. 2014).

7.5 United States of America

In the United States of America (USA), the development of the Coastal and Marine Ecological Classification Standard (CMECS) was led by the National Oceanic and Atmospheric Administration (NOAA). The current version of CMECS (FDGC 2012) was developed with input from over 100 coastal and marine habitat experts over more than a decade, and was based on several precursors and versions (Allee et al. 2000, Madden et al, 2004, 2005, 2008). The CMECS was subject to review by scientists, environmental managers, and the public during a four-month process prior to its final adoption and approval by the Federal Geographic Data Committee in 2012. The classification was designed specifically to be compatible with other national classifications (e.g., terrestrial vegetation, freshwater) to facilitate mapping across the transition of the terrestrial and coastal aquatic ecosystems. While it was developed primarily for application in the territorial waters and EEZ of the USA, the CMECS's architecture and underlying approach do not preclude application to other parts of the world (FDGC 2012).

CMECS first classifies the environment into biogeographic and aquatic settings. The biogeographic setting is divided according to separately published schema: the estuarine and coastal environments are classified by the Marine Ecosystems of the World (MEOW) (Spalding et al. 2007), and the ocean environment by the Global Open Ocean and Deep Seabed (GOODS) classification (UNESCO 2009). CMECS then divides the aquatic setting of the coastal and marine environment into three main systems; Marine, Estuarine and Lacustrine. Each of these systems can be divided into subsystems (e.g., Nearshore, Offshore and Oceanic for the Marine system). The CMECS is further organised into four components to define units within each biogeographic and aquatic setting; Water Column, Geoform, Substrate, and Biotic. The Biotic and Substrate components are subdivided hierarchically (e.g., Class, Subclass, Group, Community for the Biotic Component), but this is not the case for the other two components which don't lend themselves to subdivision by hierarchies (e.g., Layer, Salinity, Temperature, Hydroform, Biogeochemical Feature for the Water Column Component). The CMECS components include a standard list of modifiers to increase the specificity and detail of resulting classification. While Biotopes are not included formally in CMECS, they are recognised as a combination of abiotic features and associated species that users can derive and define from the classification (i.e., by identifying where Communities of the Biotic Component are consistently associated with combinations of environmental units from the CMECS settings and/or other components) (Figure 7-6).

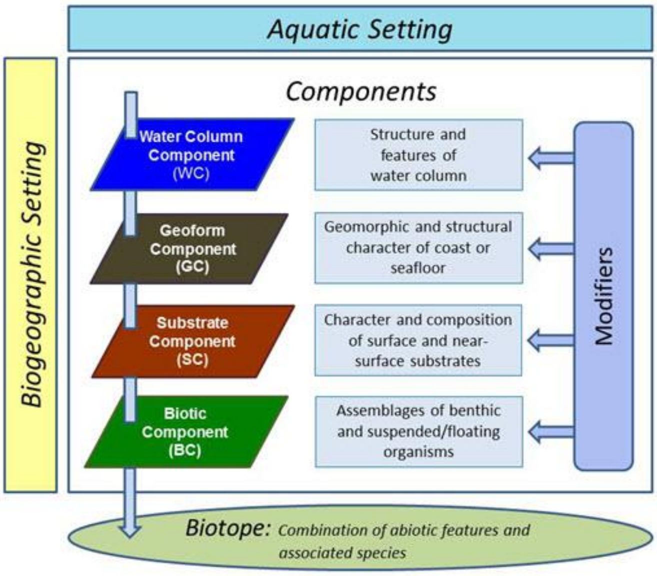


Figure 7-6: Relationship between Settings, Components, Modifiers and Biotopes of the Coastal and Marine Ecological Classification Standard (CMECS) (source: FDGC 2012).

The settings and components can be used independently or combined, as needed, to classify the environment into ecological units depending on the interests, observations, methods and objectives of the CMECS user. Figure 7-7 provides an example of how the classification can be applied to the level of biotope.

Biotope: *Phragmatopoma lapidosa* Reefs on High Energy Sand
Biogeographic Component:
Realm: Tropical Atlantic
Province: Tropical Northwestern Atlantic
Ecoregion: Floridian

Aquatic Setting:
System: Marine
Subsystem: Nearshore
Tidal Zone: Intertidal, Subtidal

Geoform Component:
Tectonic Setting: Passive Continental Margin
Physiographic Setting: Continental/Island Shore Complex
Geoform Origin: Geologic, Biologic
Level 1 Geoform: Beach
Level 1 Geoform Type: Wave Dominated Beach
Level 2 Geoform: Worm Reef
Level 2 Geoform Type: Linear Worm Reef, Patch Worm Reef

Substrate Component:
Substrate Origin: Geologic Substrate
Substrate Class: Unconsolidated Substrate
Substrate Subclass: Fine Unconsolidated Substrate
Substrate Group: Sand, Muddy Sand

Biotic Component
Biotic Setting: Benthic Biota
Biotic Class: Reef Biota
Biotic Subclass: Worm Reef
Biotic Group: Sabellariid Reef
Biotic Community: *Phragmatopoma lapidosa* Reef

Figure 7-7: Example biotope resulting from the application of the CMEC classification (source: FDGC 2012).

The CMECS is well documented and supported, with the classification, application protocols and tools (including webinars on its use), and a searchable online catalogue of units and their descriptions all available for download online. CMECS users are encouraged to provide suggestions for changes and adjustments to the classification, and there is a regular peer review and revision cycle.

There have been several independent assessments of the efficacy of the CMECS during its development. Keefer et al. (2008), using the Columbia River Estuary, USA as their test area, evaluated whether the estuarine component of the classification could adequately classify habitats across several spatial scales at sites spanning a range of conditions. These authors focused on the usefulness of CMECS for describing the physicochemical characteristics of the water column, and seabed substrate rather than biotic distributions. The aims of this study included determining the sensitivity of CMECS to spatial and temporal variability in both vertical and horizontal gradients in environmental conditions, and its potential for habitat mapping of mobile (e.g., fish) as well as sessile (e.g., benthic infauna) organisms. Keefer et al. (2008) found that the CMECS provided a useful spatially explicit framework to define habitats at 100 m^2 to >1000 m^2 scales, and that the classification of their study sites aligned well with the habitats defined by more extensive studies. They also thought the framework was flexible enough to be useful for defining the habitat of organisms with large distribution ranges, in which the habitats of organisms with more limited ranges could be adequately nested. However, they thought the CMECS could be improved by: (1) including a mechanism to describe hybrid or transition habitats where more than one of the predefined classifications options is appropriate; (2) refining classification thresholds to better reflect ecological functionality; and (3) including a temporal framework for the classification to capture predictable temporal variability in environmental parameters. The latter recommendation has been incorporated into the current version of CMECS (as a Spatial-Temporal Framework for CMECS Components) (FGDC 2012).

Guarinello et al. (2010) compared their own (unnamed) classification scheme to CMECS using data collected from a case study area in Narragansett Bay, USA. Their framework was a combination of a top-down and bottom-up approach and was focused on identifying fine-scale classification levels which they felt were underdeveloped or untested in the other classification schemes at the time. The comparison revealed that CMECS failed to classify some habitats adequately (e.g., the substrate underlying a biogenic structure), and was unable to describe ecological attributes of the habitat landscape (e.g., habitat area), community (e.g., key species beyond the dominant fauna), and function (e.g., filtration rates of mussels) that Guarinello et al.'s (2010) framework had been designed to capture because they thought it integral to a useful classification, particularly for Ecosystem-Based Management. Overall, Guarinello et al. (2010) considered that the CMECS did incorporate important ecological detail, including about the water column and geoforms, but it did not acknowledge sufficiently the biological complexity of the environment, nor the related ecological processes. Furthermore, they considered that the temporal modifiers used by CMECS make it difficult to characterise ecosystem dynamics. Guarinello et al. (2010) proposed that environmental classification frameworks should be supported by space-time schematics that can serve as communication and planning tools for scientists and managers by allowing them to match data, available resources, and programme aims with habitat classification levels. This recommendation was incorporated and expanded upon in the current version of the CMECS (e.g., Figure 7-8).

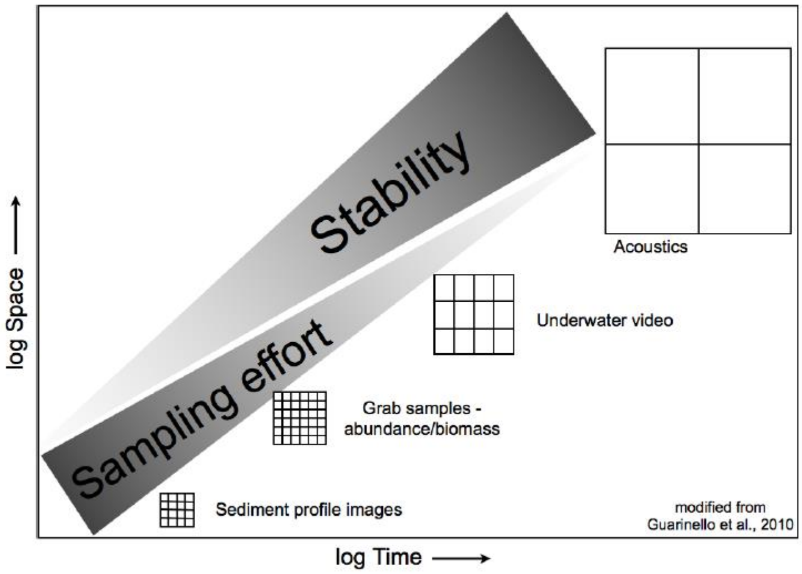


Figure 7-8: Example of the schemas included in the CMECS to illustrate how to apply its spatial-temporal framework: The stability of ecosystem features as they relate to sampling efforts and strategies (source: FGDC 2012).

Shumchenia and King (2010) used CMECS in a study to test top-down and bottom-up classification approaches for integrating physical and biological data to create habitat maps of coastal and marine environments. For their test they used data collected by standard benthic mapping tools (acoustically-derived bathymetry and backscatter data from side-scan and single beam echosounder, sediment and macrofauna data from grab samples) in Narragansett Bay, USA. The top-down approach matched (based on frequency of occurrence) identified macrofaunal assemblage groups to separately identified acoustic environment groups. The bottom-up or numerical classification of habitats was generated by a multivariate analysis routine (see below for more detail on the methodology of this sort of classification) that simultaneous linked environmental and macrofaunal data. The outputs from the two approaches were then translated into CMECS units to assess their relative compatibility. The top-down approach generated two broad and simplistic habitats, which were broadly comparable with units of the Substrate and Biotic components of the CMECS. The bottom-up classification generated more detailed CMECS biotopes which were largely nested within those two habitats identified by the top-down approach. However, some biotopes identified by the bottom-up approach occurred across the boundaries of the Substrate Component units, indicating that equating substrate type with biological assemblage type (which is an underlying assumption of the CMECS and other thematic classifications) is sometimes flawed and mean that important habitat patterns may not be identified. That is, much more habitat detail was revealed by the bottom-up approach than would have been evident if only a top-down approach had been used to apply the CMECS. In addition, information obtained from the bottom-up approach linked changes in environmental variables to biological assemblages, which could be used to predict the spatial occurrence of the macrofaunal biotopes. Overall, Shumchenia and King (2010) concluded that their analysis provided another example that mapping habitats with a bottom-up methodology or numerical classification creates more ecologically relevant habitat units.

8 Numerical classifications

Several traits of existing thematic classification schemes make them incompatible with ecological concepts and unattractive to scientists (though not necessarily environmental managers). Foremost among these issues is the top-down nature of most existing systems (e.g., EUNIS, CMECS) assumes that biological communities align with discrete environmental units even at small spatial scales (e.g., substrate classes) which ignores the abundance of contrary evidence from research that strong environmental-biological associations only usually occur at relatively board scales (Zajac et al. 2000, 2003; Hewitt et al. 2004; Stevens and Connolly 2004; Zajac 2008; Jackson and Lundquist 2016). Furthermore, most thematic classification schemes are by design prescriptive (e.g., EUNIS, CMECS), i.e., users must match their data to provided lists of habitats, which makes them inflexible and can limit their ability to identify habitats in regions where background habitat information is limited and/or changing (Fraschetti et al. 2008). Numerical classifications provide an alternative approach to classifying the environment that is a more scientifically robust, however, such classifications are not without their own separate issues for use in conservation and management planning.

8.1 New Zealand

Before reviewing a range of numerical classification methodologies that have been designed elsewhere for restricted components of marine habitats and ecosystems, it is worth remembering that both the MEC (Snelder et al. 2007) and preliminary BOMEC (Leathwick et al. 2012) are numerical classifications. The MEC has been used to design a representative network of Benthic Protection Areas throughout the EEZ (Helson et al. 2010), and the preliminary BOMEC has been used to assess the sensitivity of benthic habitat to fishing disturbance (Baird et al. 2015).

In terms of evaluating the usefulness of these classifications, data from Ocean Survey 20/20 surveys of seafloor communities on Chatham Rise and Challenger Plateau in 2007 provided the opportunity to examine the ability of the MEC and the BOMEC approach to map benthic habitats and fauna communities (Bowden et al. 2011). For this assessment, data for benthic invertebrate fauna from video and epibenthic sled samples from these surveys were first analysed to derive an independent set of 12 “biotic habitats” (Hewitt et al. 2011a). A goodness-of-fit test was then performed to determine how well the classifications (at a hierarchical class-level that divided the study area into a number of groups similar to that for the “biotic habitats”) discriminated these “biotic habitats”. To evaluate the performance of the MEC and BOMEC to discriminate the full detail of the community data (i.e., not first assigned to “biotic habitats”), two measures of classification strength were used to compare how well each classification (up to the 60 class-level) grouped the faunal data into discrete classes. The final test involved determining how well the MEC and BOMEC divided the Chatham Rise and Challenger Plateau sample sites in relation to a suite of five biodiversity metrics (e.g., taxon richness, proportion of taxa rare in abundance) previously calculated from video faunal data at each sampling site (Hewitt et al. 2011b).

Using the statistical tests of goodness-of-fit, Bowden et al. (2011) demonstrated that neither of the classification schemes discriminated well between the “biotic habitats”. Overall, both MEC and BOMEC also performed poorly when evaluated against the full detail of community data, but with the latter performing slightly better than the former. The highest levels of classification strength for this analysis were generally achieved around the ~50 and ~30 class-level, for the MEC and BOMEC respectively. Agreement between the classifications and each of the five biodiversity metrics tested were also low.

While broad-scale visual comparisons of the classifications with the distribution of “biotic habitats” (derived in part from cluster analysis) showed that the MEC and BOMEC indicated similar spatial patterns in benthic communities at scales of 100s–1000s km (Figure 8-1), the statistical testing carried out by Bowden et al. (2011) suggests that neither classification are useful for discriminating changes in benthic fauna or habitats at spatial scales less than about 100 km. Bowden et al. (2011) identified several potential reasons for the poor predictive abilities of the classifications at smaller spatial scales. These possible explanations included: mismatches between the spatial and temporal resolution of environmental data layers used in the classifications and the factors that influence the local-scale distribution of benthic fauna; weak environmental gradients at small spatial scales; faunal distributions are not entirely dependent on environmental parameters (e.g., biological parameters such as predation and competition, as well as stochastic factors, also influence faunal distribution); and a variety of potential reasons associated with the classification methodologies.

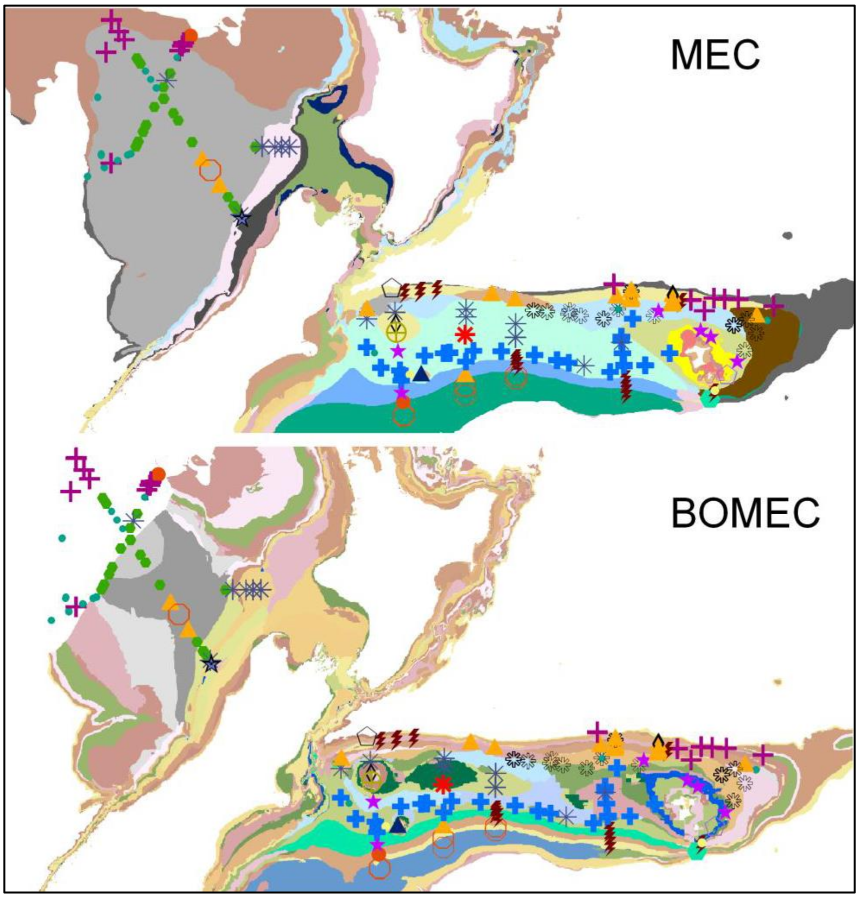


Figure 8-1: Comparison at the 150-class level of the Marine Environment Classification (MEC) (upper panel) and Benthic Optimised Marine Environment Classification (BOMEC) (lower panel) with biotic habitat groupings (coloured symbols) which were derived independently from cluster analysis of OS 20/20 sample data (source: Bowden et al. 2011).

8.2 Southern Ocean

Two other numerical classifications have been developed with a New Zealand context; bioregionalisations of the Southern Ocean and Ross Sea (Grant et al. 2006, Sharp et al. 2010). The bioregionalisation of the Southern Ocean was the initiative of a group of experts supported by the World Wide Fund for Nature (WWF) and the Antarctic Climate and Ecosystems Cooperative Research Centre. This classification was developed in response to the identification by CCAMLR (of which New Zealand is a member) of the need for a bioregionalisation to underpin the development of MPAs in the Southern Ocean. The Southern Ocean regionalisation initiative included scientists from New Zealand to help design the methodological approach to the classification, and to assess the results.

A two-stage non-hierarchical and hierarchical clustering approach (to reduce computational complexity) was used to construct a primary classification that distinguished between sites based on their similarity/dissimilarity (using Gower metric). This primary classification was based on data for bathymetry, sea surface temperature, nitrate and silicate. These data were mapped to grid cells covering the study area, which were then subjected to non-hierarchical clustering (using Clustering Large Applications (CLARA), an extension of the K-means method) to identify 40 groups. A hierarchical clustering (using unweighted pair-group method with arithmetic mean; UPGMA) of the mean data values for these groups was then performed to produce a dendrogram and final clustering, which could be identified at different classification levels. The assignment of each grid cell to a cluster group allowed for the spatial mapping of the classification or regionalisation (Figure 8-2).

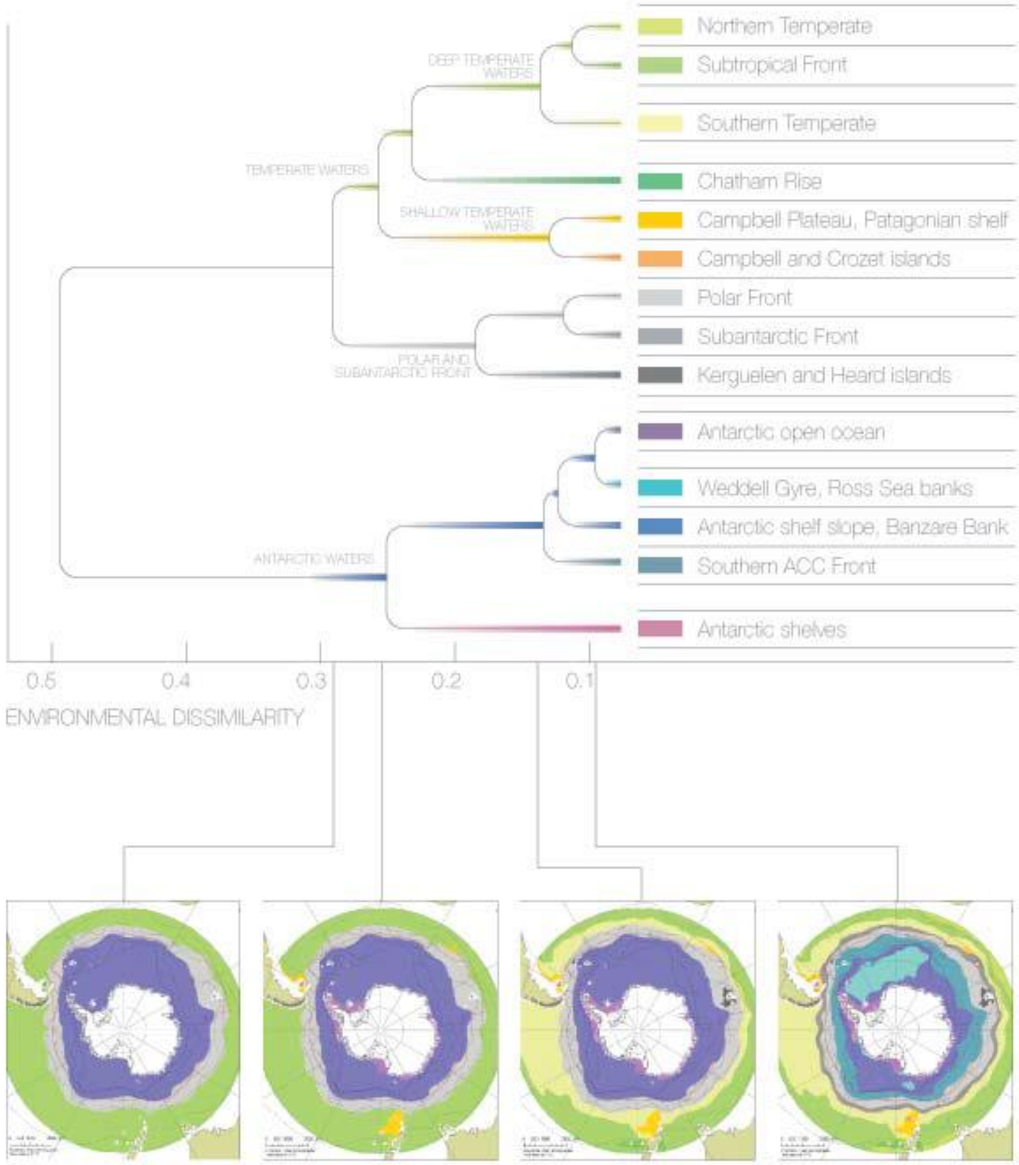


Figure 8-2: Primary regionalisation classification of the Southern Ocean; maps show regionalisations at different levels of the classification hierarchy (source: Grant et al. 2006).

The uncertainty associated with the clustering procedure was also calculated by two different methods. These methods included a metric that showed which grid cells lie at the boundary between being allocated to one or another cluster group, i.e., it is less certain that a grid cell is strongly typical of the cluster to which it has been allocated (Figure 8-3).

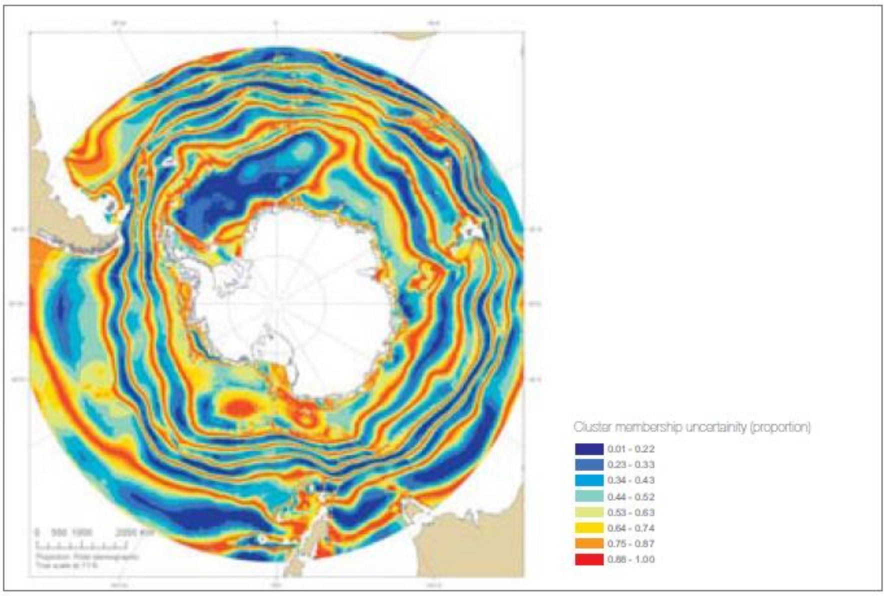


Figure 8-3: Map showing the cluster group membership uncertainty associated with the primary regionalisation classification (source: Grant et al. 2006).

A secondary classification was also carried out that included two additional datasets, for ice concentration and chlorophyll *a*. The influence of these variables on the overall classification was assessed separately and together. This analysis illustrated the environmental heterogeneity that can arise at smaller spatial scales (Figure 8-4), and was considered exploratory because the authors were unsure what regional level of separation was appropriate for using such secondary datasets, and what other sort of secondary data could also be useful (Grant et al. 2006).

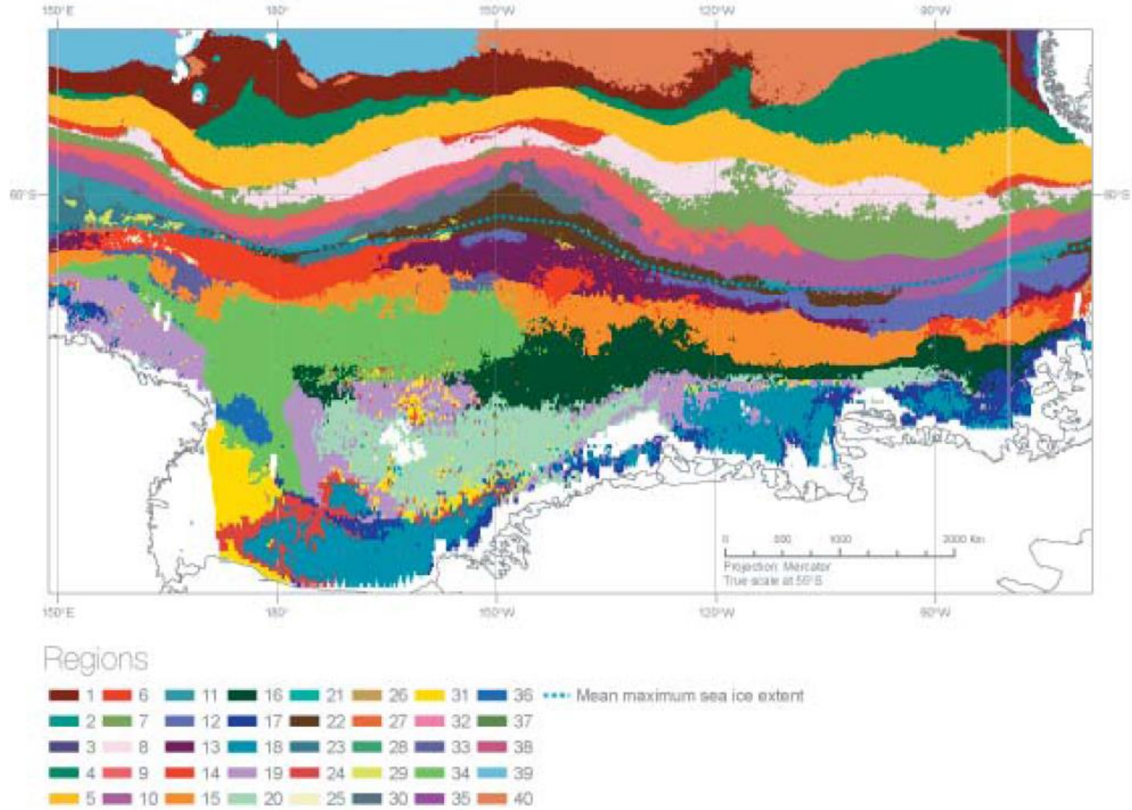


Figure 8-4: Map showing the secondary regionalisation classification for the Pacific Ocean sector (including Ross Sea) of the Southern Ocean (source: Grant et al. 2006).

The results of the classification were subject to an expert review to determine if the defined regions were consistent with current knowledge of the ecosystem. The experts were assisted in their assessment of the validity of the classification (on an ocean sector by sector basis) by overlaying known information for features such as fronts, gyres, seamounts, maximum ice extent etc. on the classification outputs. For the Pacific Ocean sector (that includes the Ross Sea), this assessment noted that major ocean and coastal features (e.g., Ross Sea shelf and slope areas) were identified, but other important features were not recognised by the primary classification (e.g., Ross Sea polynya (large open area of sea surrounded by ice), ridges in eastern part of sector). The secondary classification did identify the heterogeneity of the environment associated with island and ridge systems in the eastern part of the Pacific Ocean sector, and the expected complexity associated with the Ross Sea Gyre. However, this classification did not differentiate between the wider Campbell Plateau and southern temperate waters according to expectation (Grant et al. 2006).

The report on the regionalisation carried out by Grant et al. (2006) finished by recommending that further work be undertaken before a final classification is used to designate MPAs in the Southern Ocean. Priorities included the use of additional data, particularly biological data, and possible refinement of the statistical methods used.

In 2008 CCAMLR used the regionalization in a process to identify eleven areas in which MPA designation was considered a high priority. Members of CCAMLR were encouraged to progress spatial management planning at a regional scale, and use finer-scale bioregionalisations (i.e., classifications) to identify areas for protection, and to thereby achieve a representative network of MPAs. New

Zealand’s interest in progressing marine protection for parts of the Ross Sea, necessitated the production of such a regional-scale bioregionalisation. Following another expert workshop involving New Zealand scientists, and using the same numerical classification techniques as Grant et al. (2006), Sharp et al. (2010) used data for more than 60 environmental variables (note the substantial increase in data input compared to the classification of Grant et al. (2006)) to produce fine-scale benthic and pelagic bioregionalisations of the Ross Sea region. The benthic classification was composed of 17 benthic regions, while the pelagic realm was divided into 18 pelagic bioregions (Figure 8-5).

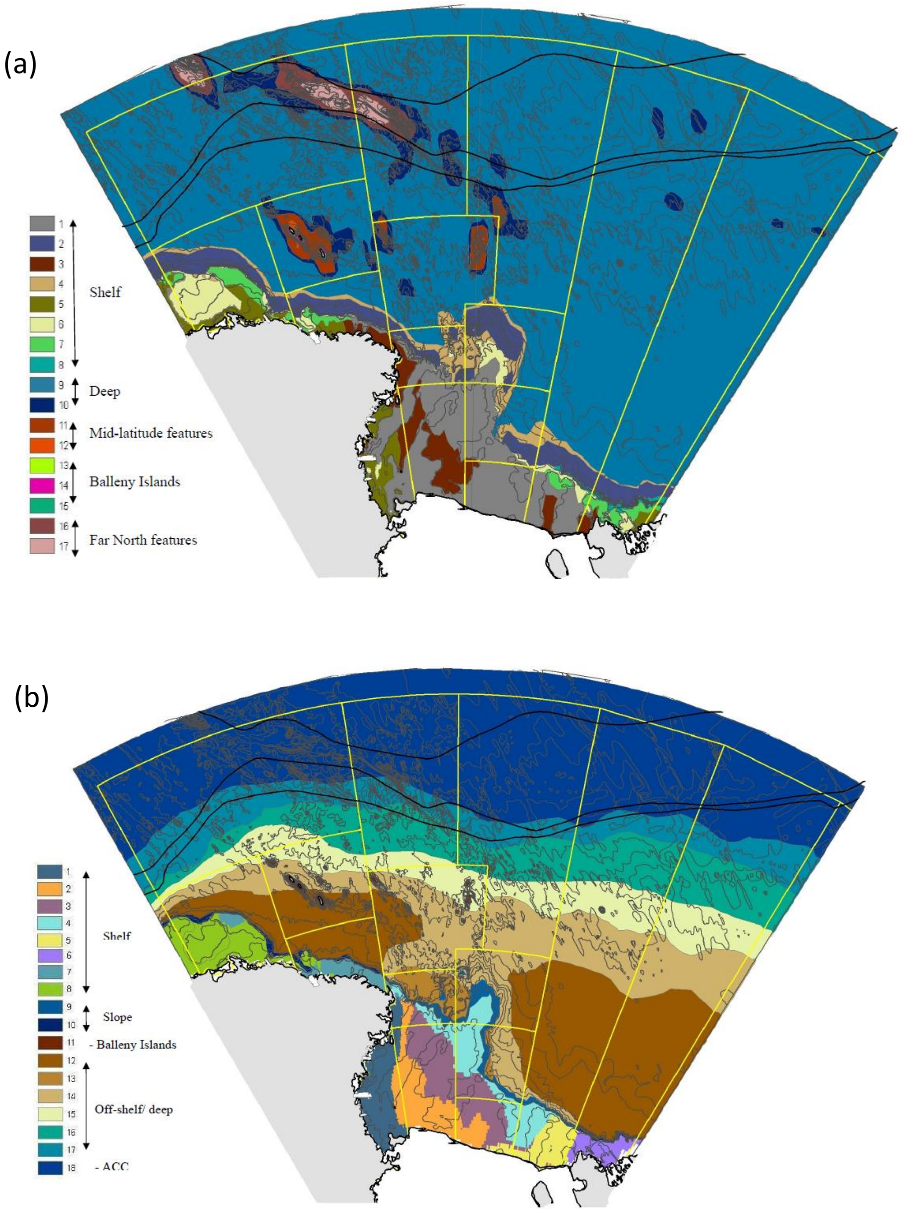


Figure 8-5: (a) Benthic and (b) pelagic bioregionalisation of the Ross Sea region (source: Sharp et al. 2010).

These bioregionalisations were used, together with maps that described spatially bounded ecosystem processes (e.g., flexible pelagic processes related to ice dynamics), to identify 26 areas of particular importance (e.g., multi-year ice zone in eastern Ross Sea that supports late and post-breeding penguins, and pupping and moulting seals) for the conservation of the Ross Sea ecosystem (Sharp et al. 2010). All this information was used to develop New Zealand's plan for a Ross Sea region MPA in 2011, and subsequent proposal by New Zealand and the USA, which came into force in 2017 following refinement. There is a requirement in the Ross Sea region MPA research and monitoring plan (Dunn et al., 2017) to investigate over the next 35 years the effectiveness of the MPA in delivering representative protection of the bioregions and protecting ecosystem processes represented in the maps.

8.3 Other numerical classifications

Numerical classifications, for use in coastal and marine habitat mapping, have developed from earlier advances in statistical analysis techniques that were scientifically-focused on identifying and describing natural assemblages and understanding the environmental factors that explained their structure and distribution (e.g., clustering, ordination and Principal Component Analysis (PCA)) (Legendre and Legendre, 1998). Among the relatively early uses of numerical classifications for conservation and management were those instances where numerical classifications of biotic data were *used in combination* with thematic classifications of abiotic data. An example of this approach is provided by Mumby and Harbonne (1999) who produced a classification for coral reefs in the Turks and Caicos Islands and Belize, with the stated aim of establishing a systematic and objective approach to coastal habitat classification. They used an agglomerative hierarchical clustering technique (with group-average sorting) to classify the benthic assemblages (based on between sites similarity measured using the Bray-Curtis Similarity coefficient). Characteristic and discriminating species of each benthic class were determined using Similarity Percentage (SIMPER) analysis. This numerical classification was used, together with a separate a thematic classification of the geomorphology of the coral reef, to assign habitats to maps. However, despite championing the use of multivariate statistical analyses to identify and describe benthic assemblage classes, Mumby and Harbonne (1999) were concerned about relying solely on these tools to define and label these classes. These authors considered that it was important to use habitat names that reflected the users' perception and intuitive expectation of the classification scheme. For example, in their case study, no benthic class was dominated by hard coral (rather macroalgae always provided the dominant cover), but to describe coral reefs with the highest coral cover as algal-dominated they considered would be politically unacceptable and confuse interpretation. Thus, they abandoned systematic accuracy for naming (provided in this case by the SIMPER) and changed the criteria around the nomenclature of the classes to aid intuitive acceptance of their scheme.

One of the early uses of a *stand-alone* numerical classifications to be developed for the marine environment was by Zacharias et al. (1999), with the specific purpose of using it to map habitats for conservation, resource management, and coastal planning. The objective of their approach was to develop statistical associations between biotic and abiotic components to create meaningful biotopes. They used the clustering routine TWINSPAN to define intertidal species associations (based on presence-absence data), which were then used as response variables in a regression tree model with environmental predictors (shoreline morphology, fetch, salinity, temperature and current velocity). The model was able to predict seven intertidal biotopes with relatively high accuracy (72%) and was used to assign biotopes to over 1000 km of shoreline of the Strait of Georgia, Canada. Zacharias et al. (1999) considered the method particularly useful because it avoided the drawback of using biotopes typically associated with thematic classifications (e.g., EUNIS), some of which are artefacts of human interpretation that cannot be defined using quantitative analysis.

Numerical classifications techniques, such as those used by Mumby and Harbonne (1999) and Zacharias et al. (1999), became widespread for identifying, describing and predicting the spatial distribution (i.e., mapping) of benthic assemblages/communities across a range of scales in the 2000s and 2010s, often within a conservation context. A variety of statistical techniques have been employed, including MaxEnt, Boosted Regression Trees, Gradient Forest, Generalised Dissimilarity Modelling, Object Based Image Analysis (e.g., Compton et al. 2012), and new methods are promoted for use in the marine environment on a regular basis (e.g., relatively recent examples include Finite Mixture Models, Dunstan et al. 2011; Region of Common Profile, Foster et al. 2013). Below a selection of studies are described (chronologically to indicate development of approaches) to provide examples of the diversity of methodological approaches and applications in marine conservation and management contexts, while also highlighting some of their advantages and disadvantages.

Oceanic-scale variation in biological and physical parameters are often used as the largest-scale component of classifications of the marine environment (e.g., global biogeography or large marine ecosystem schemes). Most of these classifications are thematic and/or subjective in nature (e.g., Spalding et al. 2007, Sutton et al. 2017). The study of Saraceno et al. (2006) provides an example of a numerical classification approach to identifying and defining oceanic provinces using ocean surface satellite data (chlorophyll-a, sea surface temperature and temperature gradient). They used a classification method based on an artificial neuronal network (Kohonen’s self-organising map and Hierarchical Ascending Clustering algorithm) to objectively define the optimal number of biophysical classes and class boundaries in the south western Atlantic Ocean. In comparison to previous classifications of the same ocean area, Saraceno et al. (2006) considered their approach to have produced a more accurate description of the major circulation patterns and frontal positions that influence the distribution of marine organisms. They also noted that by using different temporal climatologies for their data, classifications could be produced that reflected the dynamic position of the biophysical regions (Figure 8-6), a significant advantage over the fixed boundaries typically produced by thematic/subjective classifications.

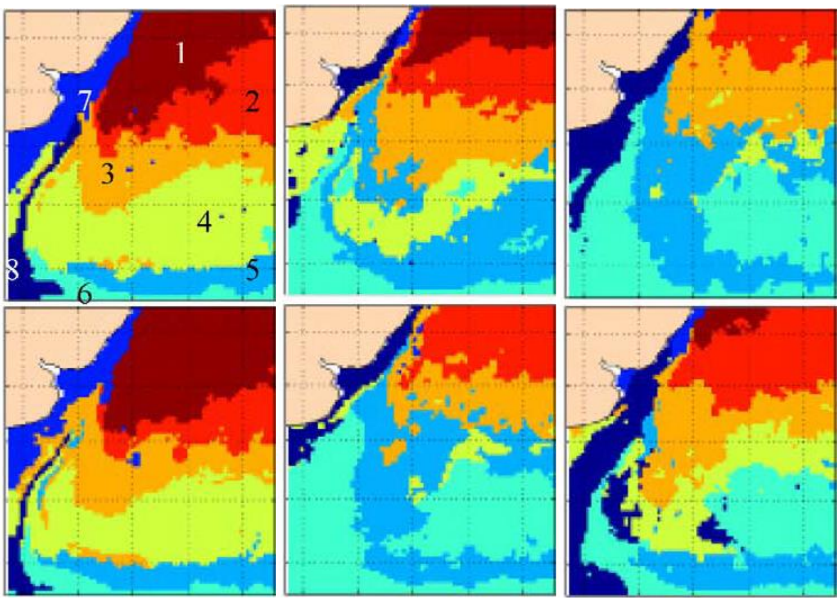


Figure 8-6: Seven biophysical regions in the south western Atlantic Ocean identified by an artificial neuronal network approach, based on different monthly climatologies of chlorophyll-a, SST, and SST gradient (left panel - January (top), March (bottom); middle panel – May (top), July (bottom); right panel – September (top), November (bottom)) (source: Saraceno et al. 2006).

A fundamental challenge for environment and species management is defining and characterising the dynamics of boundaries and transition zones. Demonstrating that quantitative analysis methods can be used to address this challenge was one of the motives for the study by Gregr and Bodtker (2007). Their aim was to identify biologically-meaningful ecosystem regions in the North Pacific Ocean based on only physical oceanographic data. They identified 15 distinct regions using image classification algorithms (a cluster routine called ISOCLUST) applied to physical data derived from an oceanographic circulation model (Regional Ocean Modeling System). The method they applied allows for the classification strength of these regions to be calculated. Gregr and Bodtker (2007) considered their classification produced a more realistic ecosystem division of the North Pacific Ocean than classifications that have been used previously at the high-level of thematic classification schemes (Figure 8-7). The regions identified by Gregr and Bodtker (2007) corresponded well in statistical tests to data for biological data chlorophyll-a, demonstrating that regions with different biological properties can be delineated using only physical variables. Analyses using the physical data partitioned into different time periods (seasonal, decadal), revealed the extent of the temporal changes in the regional boundaries, and was suggestive of broad-scale patterns in the seasonal development and variable distribution of primary production in the North Pacific (Gregr and Bodtker 2007). To be able to identify areas of spatial transition/variability, and their characteristics (e.g., elevated species diversity, fishery production), is an important attribute of a classification approach that can be flexible and adaptive enough to use in MPA planning and wider ecosystem-based management. Despite this positive attribute of numerical classifications, Gregr and Bodtker (2007) noted that variable selection and particularly the multicollinearity of variables (which could result in unintentional weighting of certain variables and interactions), the danger of unnecessary model complexity, and the scale of the variables used were all issues that had to be considered when constructing numerical classifications.

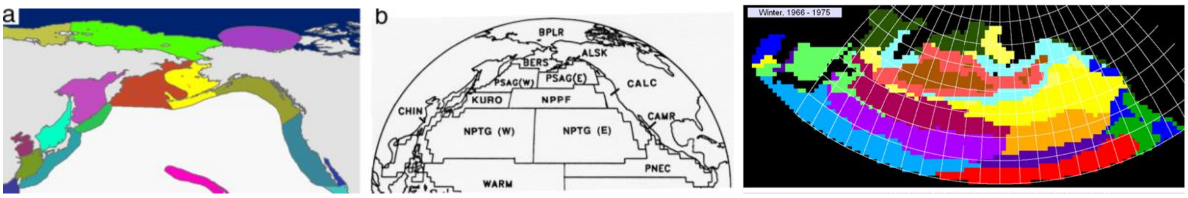


Figure 8-7: Ecosystem boundaries applied to the North Pacific for two subjective classifications (a) Sherman (1986) and (b) Longhurst (1998), compared to (c) the objective classification of Gregr and Bodtker (2007) (source: Gregr and Bodtker 2007).

Pereira and Ebecken (2009) incorporated a machine-learning approach to automatically describe the relationship between environmental variables and natural assemblages at a single site in the coastal waters of southeast Brazil. Their input data were meroplankton larvae of epibenthic fauna and the physico-chemical variables that describe the periods of upwelling and downwelling of different water masses that influence the distribution of these larvae. They used two statistical techniques (principal component analysis and K-means clustering) to identify which abiotic and biotic variables were grouped together. Expert opinion was then used to set thresholds for the influence of the environmental variables on ecological processes. These thresholds were applied, using a fuzzy rule-based association model, to the results of the ordination/clustering analysis, to determine the ‘assembly rules’ that structure the natural assemblages. The method allowed for the calculation of the accuracy and error (classification uncertainty) in their association model. Pereira and Ebecken (2009) noted how important it is to determine and report the accuracy of a classification (theirs was 85.33%), and warned that the model error can have a ‘cost’ that needs to be considered when applying the classification to practical management of the environment. Pereira and Ebecken (2009) used their approach to develop understanding of how the local ecosystem ‘worked’, and considered their findings useful for selecting appropriate ecological indicators of change for management purposes. But like other methodologies for generating numerical classifications, such an analysis not only provides understanding in terms of what environmental processes structure natural assemblages, and how they may respond to environmental changes/stressors, but also provides information to predict where such natural assemblages may occur elsewhere.

Shumchenia and King (2010) in their evaluation of CMECS (see above) used a bottom-up or numerical classification using a statistical routine called Linkage Tree (LINKTREE). LINKTREE is a modification of a multivariate technique regression tree technique (Clarke et al. 2008). It uses a binary decision procedure to produce hierarchical sub-divisions of groups or classes of sample/sites/stations, where each branch of the tree corresponds to a group defined by its biotic composition and threshold levels in associated environmental variables. Valesini et al. (2010) used LINKTREE to predict local-scale nearshore habitats in Swan and Peel-Harvey estuaries (Australia). But before using LINKTREE they identified distinct habitats using hierarchical agglomerative/group-average clustering and a similarity profile (SIMPROF) test which identifies significant group structure. Data used to classify these habitats represented a combination of biotic and abiotic variables. These so-called “enduring environmental variables” were submerged aquatic vegetation, bivalves (both derived from ground-truthed aerial photographs and satellite images), bathymetry, exposure, and location with respect to marine/river water sources (the latter as surrogate for “non-enduring” or changing physico-chemical parameters such as salinity, temperature etc). LINKTREE was used to determine which of the environmental variables, and their thresholds, were most linked to the progressive sub-division of the habitats identified by the preceding classification procedure. SIMPROF was used with LINKTREE to terminate the sub-division process at the point where there was no more significant structure among the remaining data (Figure 8-8). The variables and thresholds identified by this procedure were then used to predict the habitat at other sites where the same environmental data was available (but not included in the classification procedure).

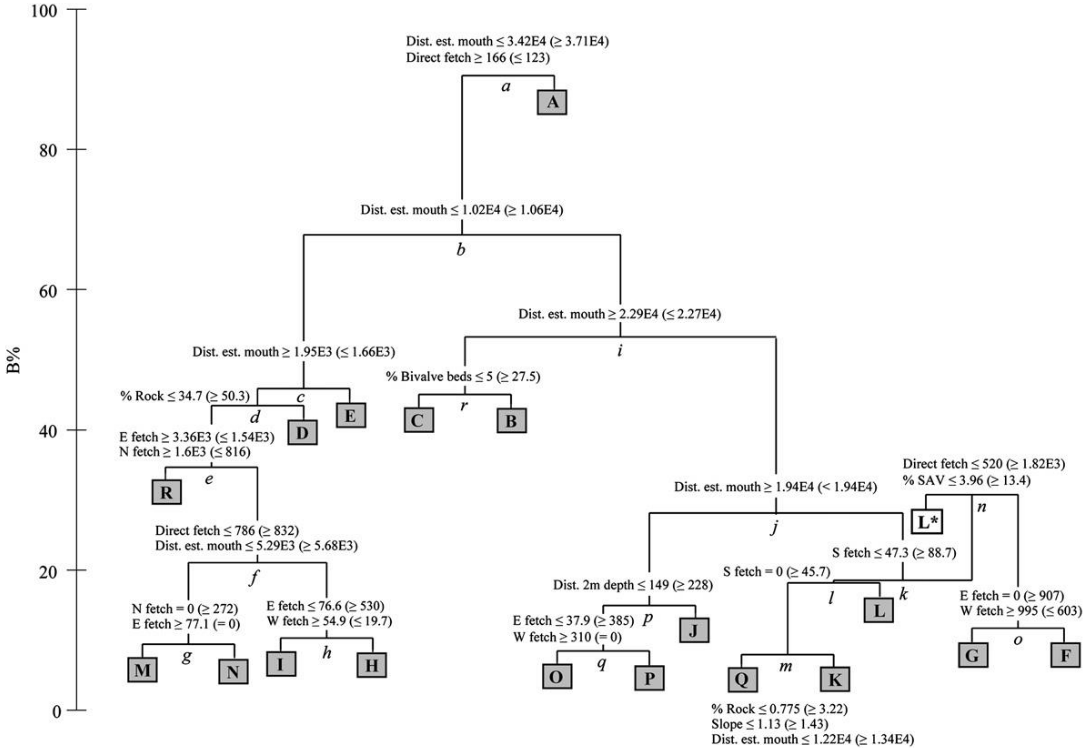


Figure 8-8: Linkage Tree and associated environmental variable thresholds for assigning habitat type in the Swan Estuary (terminal nodes in grey boxes). B% reflects the extent of inter-habitat differences as a proportion of that between the most dissimilar habitats. (source: Valesini et al. 2010).

Vaselini et al. (2010) considered that their quantitative approach to habitat classification was an important development because it removed the ambiguity associated with previous thematic and numerical classification and prediction procedures (i.e., subjective identification of classes and decision making about the most important predictor variables and thresholds), and ensured a reliable and repeatable procedure for mapping habitats for conservation planning. Furthermore, they argued that the use of "enduring" environmental variables (those not subject to much variability at a particular site over time - e.g., slope), including those acting as surrogates for "non-enduring" variables (subject to change over time – e.g., temperature), was an important strength of their approach. First because "enduring" variables are more often easily and accurately measured from mapped data (e.g., water depth). Scond because habitats based on such variables are expected to remain distinct, and display largely similar patterns of relative difference over time because they are often surrogates for non-enduring variables. To support the latter contention, Vaelini et al. (2010) noted that the relative difference between habitats in their study estuaries defined by their enduring characteristics was well correlated with that defined by their non-enduring water physico-chemical characteristics (e.g., salinity) in each season. Finally, these authors note that with the increasing availability of continuous maps for "enduring" variables, their rule-based habitat prediction technique could be automated within GIS to easily and quickly produce habitat maps for any given estuary - although strangely they didn’t produce such a map in their publication.

Pesch et al. (2011) did produce a habitat map, and used (but without a prior cluster analysis) a decision tree model approach (CART; Classification and Regression Trees) to compute a classification for benthic organisms (target or dependent variables) in the North Sea (Europe) according to a dataset describing bottom water and sediment characteristics (predictor variables) (Figure 8-9). The associations or class structure between the two sets of variables identified by the analysis, and gridded data for the environmental variables, were used to predict and map the distribution of 12 classes of benthic habitats in the study area (Figure 8-10).

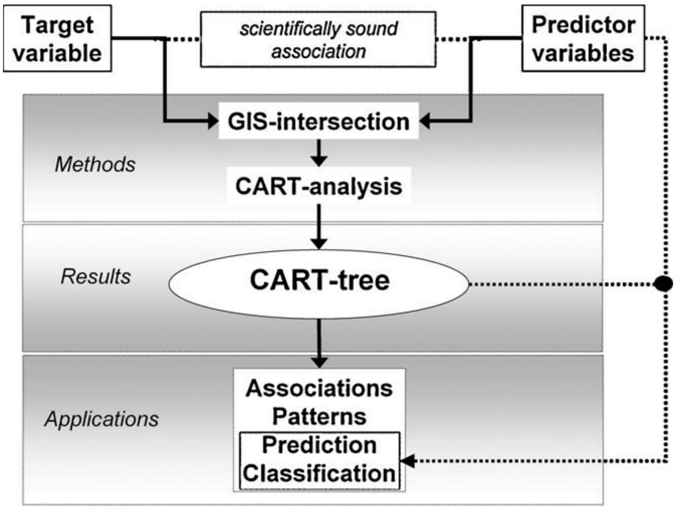


Figure 8-9: Methodological overview of a Classification and Regression Trees (CART) decision tree analysis (source: Pesch et al. 2011).

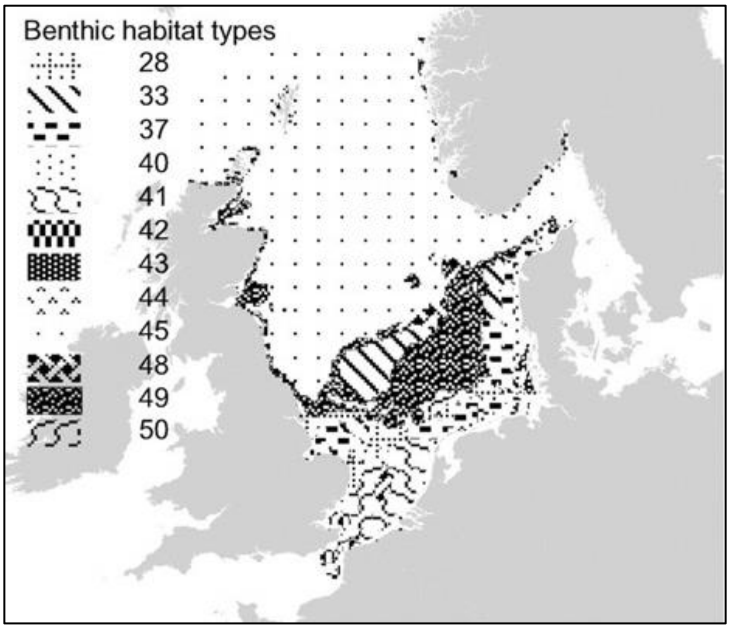


Figure 8-10: Spatial distribution of benthic habitats in the North Sea identified by the Classification and Regression Trees (CART) decision tree analysis (source: Pesch et al. 2011).

Among the benefits of this type of classification approach is that CART can handle both categorical and continuous data without data transformation, and a misclassification rate or error risk can be estimated by dividing all misclassified cases by the total number of predicted classes (Pesch et al. 2011). Furthermore, if it is necessary/desirable to simplify the spatial patterns of the target classes for habitat maps, then the results from CART and other similar decision tree methods can be aggregated to produce a less detailed output (i.e., the branches of the tree can be 'cut' at a higher/simpler level). However, decision tree models like CART produce trees by subdivision of a given dataset into a series of subclasses via binary splits, according to the features of the predictor variables. In the case of CART, this subdivision is chosen according to the predicting variable showing the highest statistical association to the target variable, with the aim of optimising the homogeneity of the target variable in the succeeding sub-division. The type of statistical association used by decision tree methods for classification vary (see Shumchenia and King (2010) and Valesini et al. (2010) for comparison), and this will affect the identification of the number and characteristics of the sub-divisions. Thus, understanding the ecological implications of the decision-making rule of such methods is important when they are being used to identify habitat classes.

Another type of numerical approach for producing ecosystem, biogeographic, or seabed community classifications and maps, and that which has seen much recent application, is exemplified by O'Hara et al. (2011). O'Hara et al. (2011) used habitat suitability models for numerous species (also known as species distribution models), in combination with traditional clustering techniques, to identify distinct regions characterised by a particular faunal composition and environmental conditions. They used presence-only records for species of ophiuroids (which they considered to be a suitable model organism, including because it is found in a wide variety of habitats) and environmental predictors (e.g., temperature, salinity, organic carbon flux etc) sourced from publicly accessible databases. These data were used to model (maximum entropy modelling, MaxEnt) and map the probability of presence of >200 species for grid cells in a large area encompassing parts of the Indian, Pacific, and Southern oceans. The probability data were then used in a 2-stage clustering process (non-hierarchical k-means and hierarchical agglomerative/group-average) to identify seven assemblage groups or classes, which were then mapped to the study area using the model relationship between assemblage classes and environmental parameters in unsampled cells (Figure 8-11). This is an example of a 'predict then classify' approach (compared to 'classify then predict') to mapping assemblages (Ferrier and Guisan 2006).

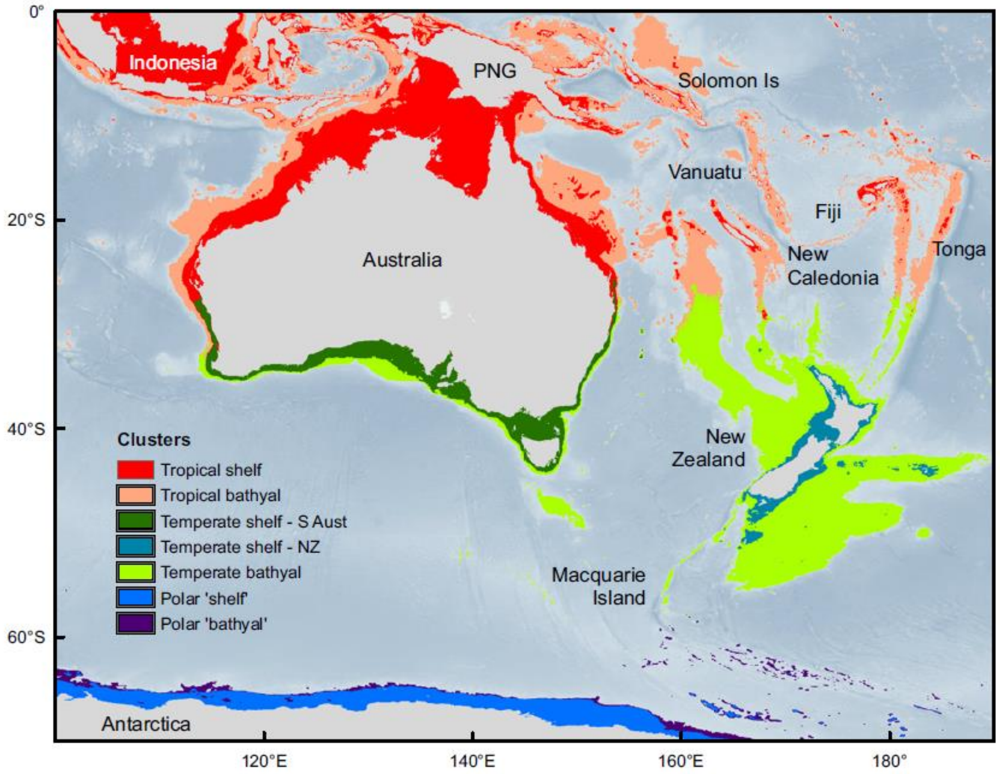


Figure 8-11: Map of the classification of seafloor assemblages generated by cluster analysis of the output predictions from maximum entropy modelling of >200 ophiuroid species (source: O'Hara et al. 2011).

One of the advantages of the approach used by O'Hara et al. (2011), and other similar modelling approaches, is that it is possible to extract information from the model outputs that can also be useful for conservation and management planning. For example, they can be used to determine the spatial patterns of species richness (Figure 8-12) and other univariate metrics of biodiversity such as rarity, uniqueness etc. In addition, as well as using site or area-based cluster analysis to classify assemblage groups, species-based cluster analysis can be performed to generate visualisations of turnover in species composition, for example, the proportion of tropical, temperate, and polar species (Figure 8-13). The usefulness of predicting and mapping species turnover for conservation purposes will be considered further in the Discussion.

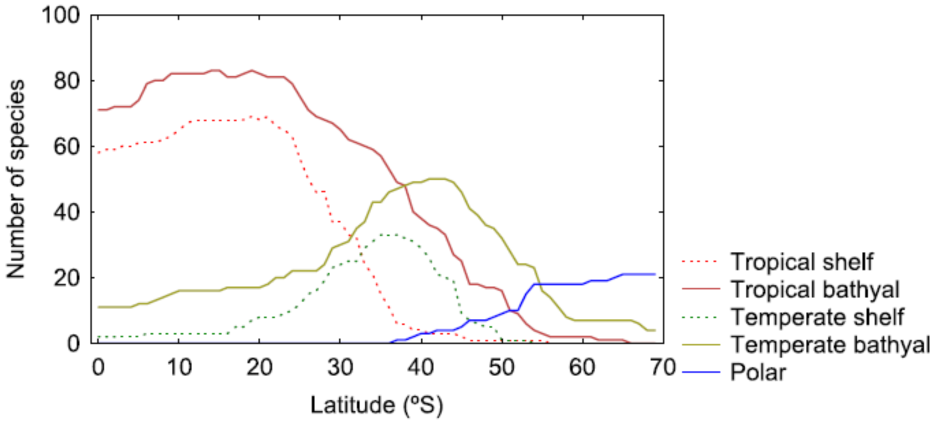


Figure 8-12: Number of species in the seafloor assemblage classes for each degree of latitude (source: O'Hara et al. 2011).

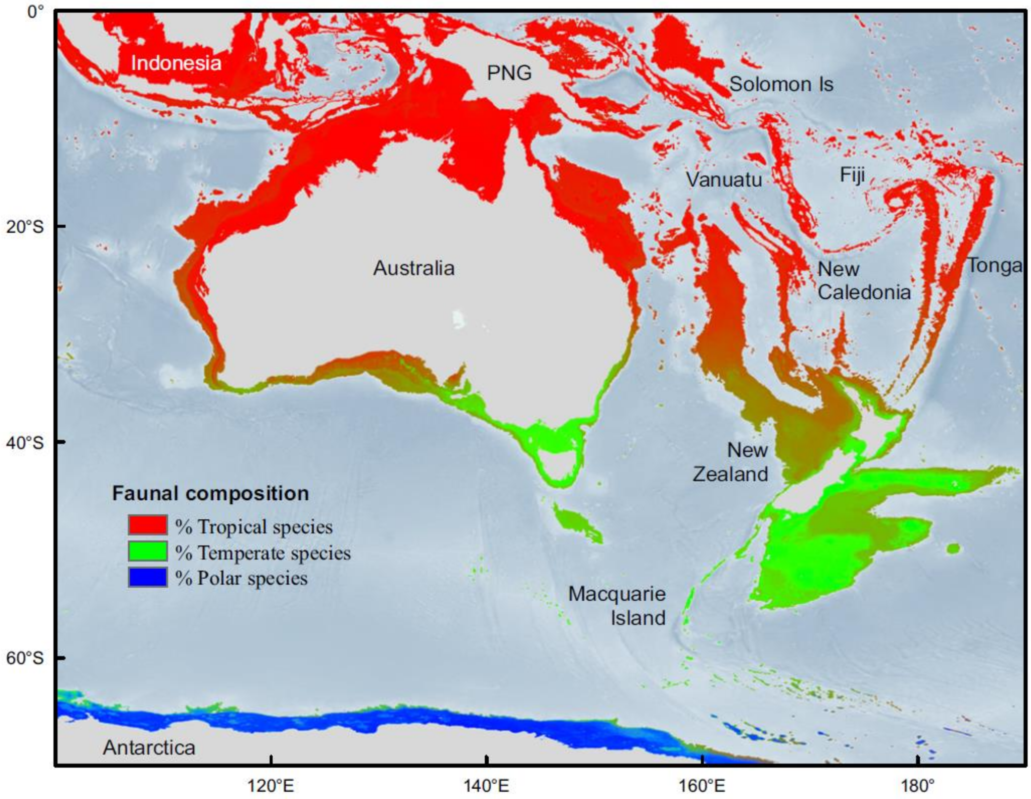


Figure 8-13: A visualisation of the spatial patterns of predicted distribution of species turnover, formed by assigning the proportion of tropical, temperate, and polar species from the maximum entropy modelling to the red, green, and blue bands of an RGB colour image (source: O'Hara et al. 2011).

The type of presence-only approach used by O'Hara et al. (2011) utilises a randomly generated set of pseudo absences or "background" data to model the relationship between fauna and environment, which is a useful attribute of such models because true absence data are often rare in marine species datasets (particular those for deep-sea taxa). If true absence data, and abundance data, are available they can be used in these types of habitat suitability/species distribution models. The overall performance of these types of models can be assessed using background or true absence data, as well as the spatial error or uncertainty associated with the final map prediction. The overall validity of the model can also be determined by first running the model with part of the available dataset withheld, against which to check the accuracy of the model. Ideally, the validity of the model would be assessed by independent ground-truthing directed by the model outputs. As already noted above, being able to attach statistical uncertainty and validity to a model is useful when using the output maps for conservation and management planning.

The final example of numerical classifications detailed in this section demonstrates the use of mapping model uncertainty when predicting the spatial distribution of biotic assemblages. Hill et al. (2017) used a recently developed approach for marine applications, called Regions of Common Profile (RCP), to quantify and map the distribution of demersal fish assemblages on the Kerguelen Plateau (Southern Ocean). The RCP approach is a multi-species model approach that can overcome many limitations of traditional distance-based approaches (i.e., such as clustering techniques and GDM that use similarity metrics calculated between pairs of sites, and uncertainty in the final classification can appropriately be quantified from the one model). Formally, the RCP approach is a multivariate adaptation of a mixture-of-experts model, and it differs from standard mixture models because the mixing proportions vary with covariate data (Foster et al., 2003). Hill et al. (2017) used RCP to simultaneously group or classify sites with a similar composition of species (the site's species profile) and describe the patterns of variation in these assemblages using environmental data (15 variables describing sea surface and seafloor conditions), allowing the prediction of assemblages or RCP classes across the study region. This 1-stage classify and predict approach has advantages over 2-stage processes that for example, first classify and then predict into geographic space where only environmental data exist, or predict and then classify (Ferrier and Guisan, 2006, but see also Baselga and Araújo, 2010).

Hill et al. (2017) used a forward selection procedure to select environmental variables and the number of classes or RCPs simultaneously. The best model was achieved by the combination of environmental variables and number of RCPs that minimised the Bayesian Information Criteria (BIC), and no further improvement in BIC occurred between selection steps. Seven classes or RCPs were identified that were defined by depth, surface temperature and chlorophyll a (the relationship of which can be visualised, Figure 8-14), and these were mapped onto the study area (Figure 8-15a). Hill et al. (2017) quantified a number of different model uncertainty parameters, including the probability or likelihood that a map cell belongs to a RCP class (Figure 8-15b). This probability map allowed the authors to consider the source of this uncertainty, and potentially devise strategies for improving their model. For example, they noted that assemblage classes are less certain at the intermediate depths of the Kerguelan Plateau and near class boundaries, and postulated that this uncertainty may reflect: (i) the relatively sparse sampling between the islands; (ii) that the available environmental variables may not entirely capture biological differences at this location on the plateau; (iii) the difficulty in defining hard boundaries for assemblages within a continuum of multispecies responses, e.g., at the transition between deep and shallow fauna; and/or (iv) the temporal mismatch between the environmental variables (which were climatologies) and data from the biological sampling.

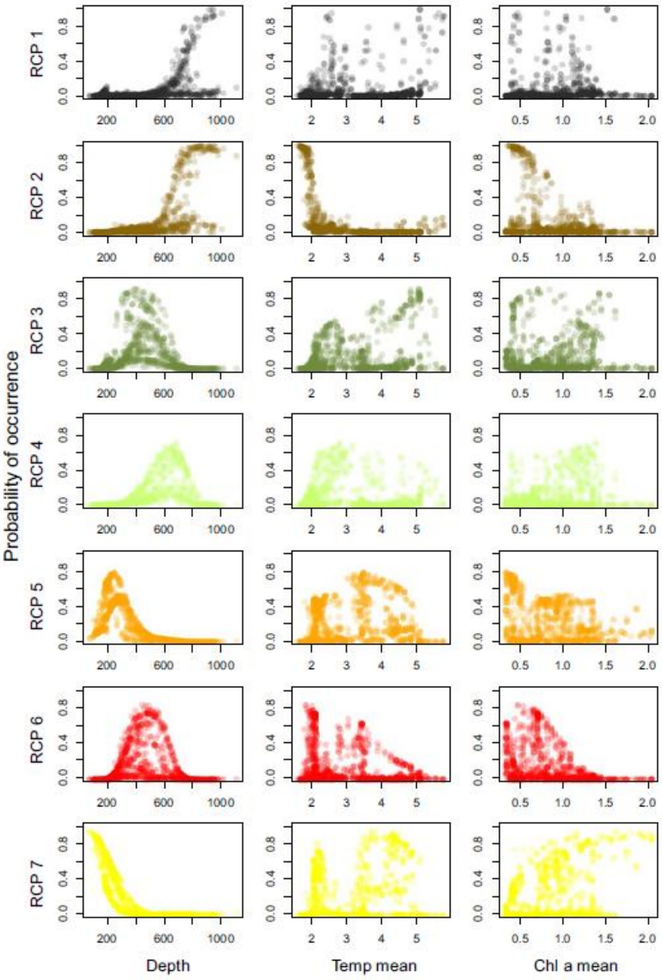


Figure 8-14: Response of each Regions of Common Profile (RCP) to depth, surface temperature yearly mean and chlorophyll-a yearly mean. Plots were generated by predicting RCP membership for each trawl site based only on its environmental covariates (source: Hill et al. 2017).

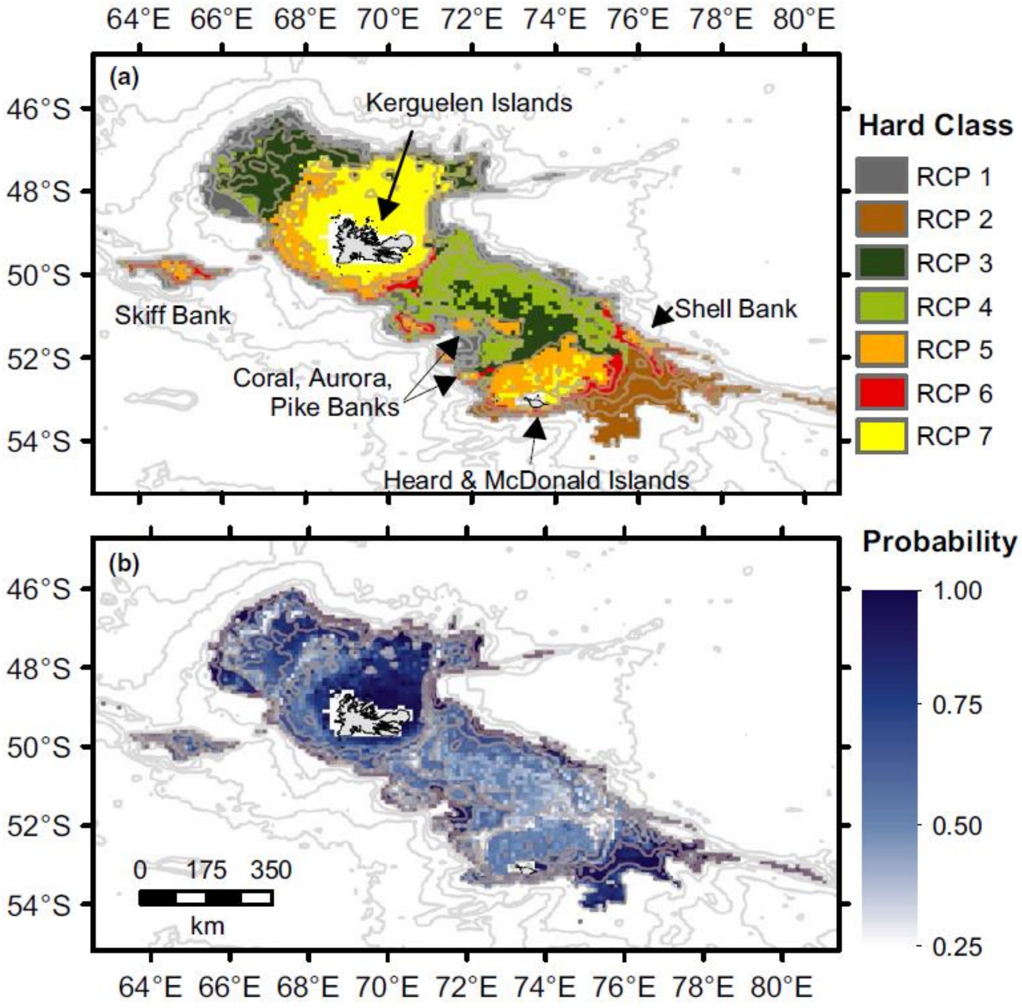


Figure 8-15: Maps of the (a) classes predicted by the Regions of Common Profile (RCP) model, and (b) the probability or likelihood that a map cell belongs to the class (source: Hill et al. 2017).

Because the RCP model predicted the occurrence of individual species across the study area as well as the species composition of sites, it was possible for Hill et al. (2017) to determine the proportion of endemic and cosmopolitan species. This information was coupled with the results of the modelling that determined the extent of the representativeness of the demersal fish assemblages (a group of organisms Hill et al. (2017) considered to be key component of the local ecosystem) within the Heard Island and McDonald Island Marine Reserve in the study area. These outputs demonstrate convincingly the usefulness of RCP and other similar simultaneous modelling techniques for defining and prioritizing areas for conservation, targeting monitoring efforts, and managing human activity.

8.4 Recent regional examples of the application of numerical approaches

A relatively recent, and perhaps the best current example of the advantages of numerical approaches is provided by the classification and other analyses conducted for the continental shelf area of the Great Barrier Reef World Heritage Area (Pitcher et al. 2007). Pitcher et al. (2007) used biological and environmental data from multiple surveys that used a range of sampling gears, together with different numerical approaches (predictive models for species, clustering of the data outputs of these models as well as raw data for species and substratum habitats) to classify the shelf area for different taxonomic groups and habitat types. The selection of appropriate environmental variables for the bio-physical modelling was rigorously evaluated, and field validation was carried out to refine the results of the various analyses. Metrics for biodiversity (e.g., biomass, species richness, rarity, uniqueness etc) of the shelf area were derived from the raw and modelled data and used for the evaluation of the effectiveness of current MPAs, and to provide recommendations for future spatial management planning and monitoring. Pitcher et al. (2007) also used the information they generated from the species modelling together with data for bottom trawling effort to estimate the exposure of seabed fauna to fishing disturbance (estimates of proportion caught), and the likely extent of past effects of trawling on the benthos (negative or positive changes in biomass) over the shelf area. Indicators based on qualitative recovery ranks of the species used in the models were used to assess the relative risk with respect to trawling. These assessments demonstrate how it is possible to use data inputs for numerical classifications to assess the quality of habitats, which could influence the choice of habitats to represent in an MPA network. In this example the impact of fishing disturbance on habitat quality could be assessed, but it is also possible given appropriate data to similarly assess the potential influence of land-derived sedimentation or nutrients on habitat quality.

In a later publication, data from the Great Barrier Reef Heritage Area analyses were compared with similar data from other areas of the world to determine the wider effectiveness of using environmental variables and a Gradient Forest (GF) approach to map patterns of biodiversity (Pitcher et al. 2012). From this study, Pitcher et al. (2012) concluded that importantly GF enabled the combination of data from disparate datasets, information from the gradient response curves can be used to transform environmental variables so that they can be used to predict and map patterns of biodiversity better than uninformed variables, and they considered that the approach is particularly useful for supporting spatial management planning in the marine environment.

Current work by NIWA[1] is building on the research of Pitcher et al. (2012) to produce classifications at finer spatial resolutions using the GF approach. GF is highly flexible and can incorporate a large number of environmental predictors (irrespective of co-linearity) as well as accounting for complex non-linear interactions without overfitting the models. GF is also particularly well suited to the analysis of large datasets, whose size can be limiting in other methods. Nonetheless, GF can provide relevant trends in species turnover with relatively low data availability.

As an example of the utility of this approach for the New Zealand context, a 30-group classification was produced for demersal fish within the Extended Continental Shelf (Stephenson et al., in submission) and predicted spatially at a 1 km^2 grid resolution (Figure 8-16). This 30-group demersal fish classification, based on aggregated species turnover functions from GF models for 253 species, proved to be highly effective in summarising both variation in fish assemblage composition (assessed using independent samples). The number of classes within a classification can easily be modified in

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[1] This research is funded primarily by ongoing research within the NIWA SSIF-funded Coasts & Oceans project “Biodiversity connectivity and measures of health” with links to aligned research in the Sustainable Seas National Science Challenge project “Spatially explicit decision support tools” (July 2016 – June 2019).

numerical classifications. For this example, a classification at a 30-group level was adopted to facilitate visualisation. However, testing of correlations between environmental and biological similarity at higher levels of classification detail (up to 100 groups) indicate that these levels can provide even greater discrimination of compositional differences and species turnover than those presented by the 30-classes. Thus, similarly to BOMEC classification, the hierarchical nature of the classification supports its use at varying levels of classification detail, which is advantageous for conservation planning at differing spatial scales. Because the classification is based on GF models of species turnover functions across environmental gradients, it can accurately reflect differences in species composition spatially, e.g., across depth gradients. This attribute of the underlying GF models means that a single classification can reflect the dynamic environments in inshore areas with a greater number of classes compared to fewer classes in the more homogenous offshore areas. That is, this approach to classification excludes the need for separate classifications between coastal and marine classifications.

Further work is currently underway to extend the classification to include other taxonomic and ecological groups (e.g., shallow water macro-algae and benthic invertebrates), and thus to represent more broadly the benthic communities associated with coastal and marine habitats. The production of single-taxon classifications for each of the other groups will also be possible, and can be used for management purposes that require a more taxon-specific approach. The efficiency of GF modelling opens the possibility of combining a broad-scale New Zealand-wide classification (e.g., 4-16 classes), and nested analyses constructed for regional subsets of the data; if the mix of factors controlling species turnover varies region by region, then this may provide better representation of regional features not well described by a broader New Zealand-wide analysis. However, consideration should be given to the quality and quantity of data available within each region, for these will have implications for the spatial validity of the classification. More data generally results in a better fitting model, but model performance is also dependent on the biology of the modelled organism. For example, evidence from other studies indicates that species with limited geographic range and environmental tolerances are generally better modelled than those with greater ranges (Thomson et al., 2014, Morán-Ordóñez et al., 2017, Guisan et al., 2013) because widespread species are less likely to have sharp easily identifiable environmental thresholds that clearly delineate their most suitable environmental conditions (Morán-Ordóñez et al., 2017). Finally, the environmental predictors used in the recent classification by Stephenson et al. (in review) are available at 250 m grid resolution which could allow classification at these fine scales (particularly useful for inshore areas where there is there is greater heterogeneity in assemblages over smaller spatial scales than further offshore).

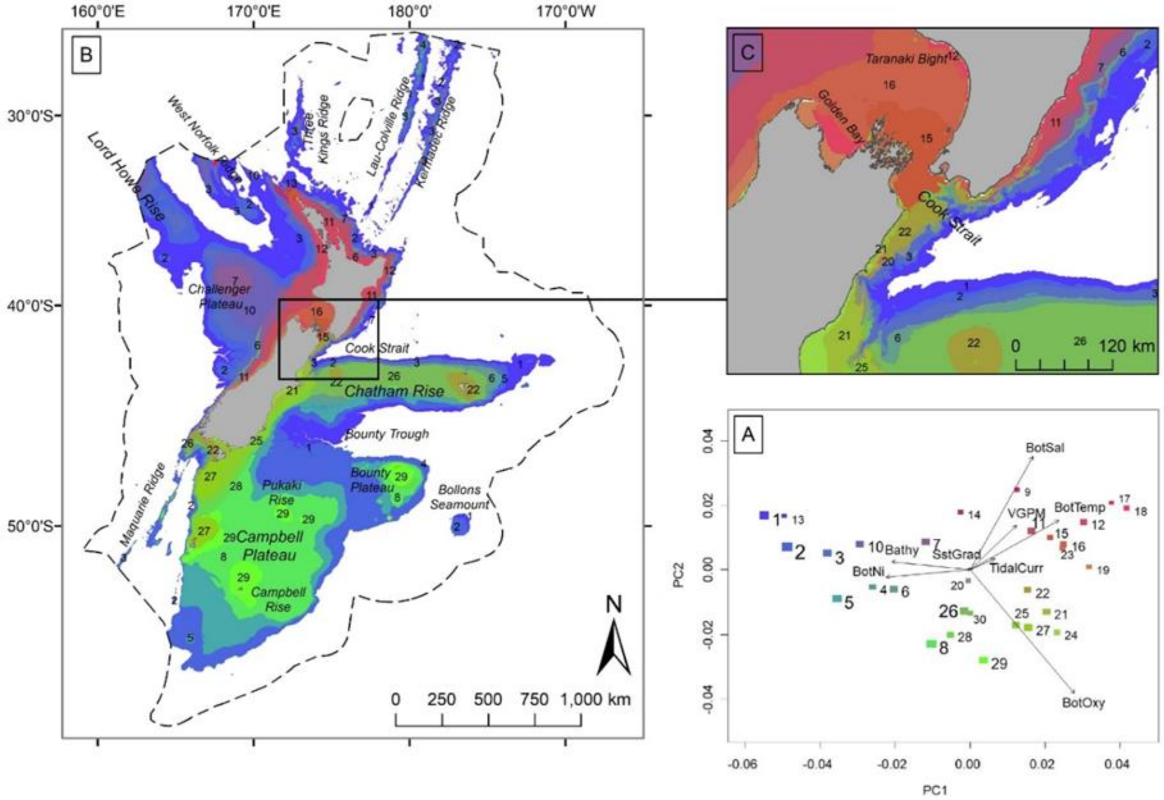

Figure 8-16: Distributions in PCA and geographic space of 30 demersal fish groups defined by classification of environmental predictors for the seas within the New Zealand Extended Continental Shelf to a depth of 2000 m after transformation using a Gradient Forest model fitted to data from 13,917 research trawls. Colours are based on the first three axes of a PCA analysis applied to the group means for each of the transformed predictor variables, so that similarities/differences in colour correspond broadly to intergroup similarities/differences in the transformed environmental space. A) Distributions of groups in PCA space, with vectors indicating correlations with the eight most important environmental predictors and symbol/font size indicating the relative size of the group area; B) Geographic distributions of groups across New Zealand's Extended Continental Shelf (dashed line); C) Geographic distribution of groups at finer scales, centred on Cook Strait.

9 Ancillary concepts

Concern about the damage caused to marine habitats and biodiversity, and the desire to prevent adverse future disturbance from human activities such as bottom fishing, has resulted in the identification of natural features that require particular protection based on their relative importance to the function of the marine ecosystem. These features include what are called Ecologically or Biologically Significant Areas (EBSAs; CBD 2009) and Vulnerable Marine Ecosystems (VMEs; FAO 2009) which are internationally recognised concepts, as well as variants that are recognised at the level of a nation state (e.g., "Key Ecological Features" of Australia; Falkner et al. 2009). In New Zealand, identification of "Key Ecological Areas" has been identified as a potentially useful component of future MPA design processes (ISAG pers. comm.), while at the local level regional councils are required to identify "sites of significant ecological value" as part of their planning process to protect biodiversity (e.g., Lundquist and Smith 2014). There is a growing demand to map the occurrence of these features and incorporate them in marine spatial planning, both in national and international waters, and therefore there are obvious benefits if a classification scheme or component of the underlying analysis could readily identify them.

VMEs, EBSAs and the like are identified according to a set of criteria. Many of these criteria are the same or similar across the majority of these feature types, and include uniqueness or rarity, habitat fragility and functional significance. There are also criteria that are more restricted in their application to particular concepts of a feature (e.g., naturalness) (Table 9-1) (Ardron et al. 2014). Examples of EBSAs, VMEs and related features include seamounts in productive waters that may provide habitat for certain life stages of some organisms, and coral reefs and other structural biogenic habitat that support high levels of biodiversity (Lundquist et al. 2017).

Table 9-1: Comparison of VME and EBSA criteria (source: Ardron et al. 2014).

FAO VME		**CCAMLR VME**		**CBD EBSA**	
Criterion	**Definition**	**Criterion**	**Definition**	**Criterion**	**Definition**
Uniqueness or rarity	An area of ecosystem that is unique or contains rare species whose loss could not be compensated for by similar areas or ecosystems. These include: habitats that contain endemic species; habitats of rare, threatened or endangered species that occur only in discrete areas; or nurseries or discrete feeding, breeding, or spawning areas.	Rare or unique populations	Species that create dense, isolated populations	Uniqueness or rarity	Contains either (i) unique, rare (occurs only in few locations) or endemic species, populations or communities, and/or (ii) unique, rare or distinct habitats or ecosystems; and/or (iii) unique or unusual geomorphological or oceanographic features
Functional significance of the habitat	Discrete areas or habitats that are necessary for the survival, function, spawning/reproduction or recovery of fish stocks, particular life-history stages (e.g. nursery grounds or rearing areas), or of rare, threatened or endangered marine species.	Habitat-forming	The degree to which they create habitat that could be used by other organisms	Special importance for life history stages of species importance for threatened, endangered or declining species and/or habitats	Area containing habitat for the survival and recovery of endangered, threatened, declining species or area with significant assemblages of such species
Fragility	An ecosystem that is highly susceptible to degradation by anthropogenic activities	Fragility	The potential for damage or mortality resulting from physical disturbance from bottom fishing gear	Vulnerability, Fragility, Sensitivity, or Slow recovery	Contain a relatively high proportion of habitats, biotopes or species that are highly susceptible to degradation or depletion by human activity or by natural events.

FAO VME		**CCAMLR VME**		**CBD EBSA**	
Life-history traits of component species that make recovery difficult	Ecosystems that are characterised by populations or assemblages of species with one or more of the following characteristics: slow growth; low age of maturity; low or unpredictable recruitment; or long lived	Slow growth Larval dispersal, potential longevity Lack of adult mortality	Organisms which grow slowly will take a longer time to attain a large size or reproductive maturity [Limited dispersal] influences the ability of a species to recolonise impacted areas Estimate of maximum longevity for the members of the taxon Lack of Mortality [adds] some degree of vulnerability and decreases resilience because as adults those organisms cannot redistribute themselves in response to a direct disturbance	Vulnerability, Fragility, Sensitivity, or Slow recovery	Slow recovery
Structural complexity	An ecosystem that is characterised by complex physical structures created by significant concentrations of biotic and abiotic features. In these ecosystems, ecological processes are usually highly dependent on these structured systems. Further, such ecosystems often have high diversity, which is dependent on the structuring organisms.	Habitat-forming	See above	*No explicit comparable criterion*	
No explicit comparable criterion		*No explicit comparable criterion*		Biological productivity	Area containing species, populations or communities with comparatively higher natural biological productivity
No explicit comparable criterion		*No explicit comparable criterion*		Biological diversity	Contains comparatively higher diversity of ecosystems, habitats, communities, or species, or has higher genetic diversity
No explicit comparable criterion		*No explicit comparable criterion*		Naturalness	A comparatively higher degree of naturalness as a result of the lack of or low level of human-induced disturbance or degradation

Efforts are currently underway to identify VMEs and EBSAs in areas beyond national jurisdiction, and various frameworks and methodologies have been developed to assist in these processes (e.g., for EBSAs - Taranato et al. 2012, Ardron et al. 2014, Clark et al. 2014, Dunn et al. 2014, Dunstan et al. 2016). None of these approaches proposes the use of an existing classification to identify VMEs or EBSAs in a stand-alone capacity, but either classifications, a step in a classification process, or the underlying data can be used to help identify these features. Gregr et al. (2012) in their review of existing marine habitat and ecosystem classifications to identify EBSAs and potentially similar "Important Marine Features" and "Biologically Sensitive Areas" in Canadian waters, concluded that while no single classification currently provides a way to identify such features, it was possible to produce a methodology whereby an existing classification could be used to assist in the identification EBSAs and their ilk. Real-world examples for the use of thematic classifications include Clark et al. (2014) who using the deep-sea biogeographic classification of Watling et al. (2013) to help identify seamount EBSAs in the Pacific Ocean. Yamakita et al. (2015) provide an example of how numerical classification approaches can identify kelp forest ecosystem EBSAs along the coast of Japan. The data input, analysis, and output flexibility inherent in numerical classification methodologies is likely to more useful overall for identifying EBSAs, VMEs and related features, and incorporating them into the process of spatial management planning in international or national waters.

10 Discussion

This review, and a previous gap analysis (DOC and MFish 2011), have identified a range of structural, performance, inadequacy, and fundamental issues that make New Zealand's CMHEC not fit-for-purpose. In brief, the current CMHEC is unable to provide a seamless classification from estuary to the deepest ocean depths, be applied consistently at different spatial scales, and provide the fullest possible information about biotic habitats and their biodiversity. The CMHEC has not been tested widely to confirm that its aim to identify habitats as surrogates for "biological pattern" is met. To be fit-for-purpose, a classification for New Zealand's coastal and offshore habitats would now also need to be flexible enough to inform the designation of the categories of MPA that may allow for protection of certain species and habitats. Ideally, information used to derive the classification would also be useful for identifying EBSAs, VMEs, and other similar key ecological features.

10.1 Fit-for-purpose

Before considering the results of this review further, it is worth returning to the concept of fit-for-purpose, covered in Section 6. There is no single and best way to classify coastal and marine habitats. Classifications should be specific to conservation and management objectives, i.e., the classification approach adopted or developed needs to be able to address the question(s) being asked. Data availability can also influence the type of classification that can be used. However, the availability of appropriate data should not necessarily influence the choice of a classification. It is better to choose the most appropriate classification methodology, and thereafter prioritise obtaining those data needed to use such an approach.

In Section 6, the general attributes of a fit-for-purpose classification for New Zealand coastal and marine habitats were outlined. A relevant classification must be able to: identify habitats at a range of spatial scales; account for variability in data availability and quality within and across spatial scales; be modifiable; be mappable; and easily interpreted and understood by a range of stakeholders and end-users including the general public. These attributes are related to the basic need of a classification to be able to inform the development of a representative network of MPAs across New Zealand's marine environment. It is relatively straightforward to identify suitable approaches to produce a classification with these attributes and to satisfy that objective. However, also in Section 6, several other attributes were listed which ideally would be included in a future classification. These attributes (see pages 30-31) can be represented by a classification, but the extent to which they can be included in any one classification will be depend on the type of approach used.

10.2 Which classification approach to use?

There are two main types of classification, thematic and numerical, each with pros and cons. This review has already described in detail several classifications of both types, and highlighted the benefits and drawbacks of the two approaches. There are classifications that involve both classification approaches (so-called 'mixed' classifications), but these are usually based primarily on a thematic approach (e.g., EUNIS). By default, these mixed classifications incorporate the pros and cons of both types of approach, without any overall benefit resulting from their combination, and potentially introducing an added element of confusion to the user. Below we summarise the pros and cons of thematic and numerical classifications, and make specific reference to both the general and ideal attributes of a fit-for-purpose classification in the New Zealand context. We also consider options for replacing the CMHEC, before finally making a recommendation for future steps towards achieving the goal of having a fit-for-purpose classification.

10.2.1 Thematic classifications

Thematic classifications, such as the CMHEC, have been a popular choice for conservation management - in part because of their conceptual simplicity and relative ease of application, which means they are also easily interpreted and understood by end-users. Such classifications also lend themselves to including both pelagic and benthic components of coastal and marine habitats, and can usually be readily aligned with estuarine and/or terrestrial classifications. Of the major thematic classifications of coastal and marine habitats reviewed here, the USA's CMEC is probably the best developed and could be a suitable candidate for replacing or modifying New Zealand's CMHEC. The CMEC was developed through an extensive design and review process, it is flexible to modification, and supported by databases and tools that allow it to be used in the best possible manner. The federal organisation that manages the classification has committed to supporting the use of CMEC in other countries. Thus, should New Zealand adopt the use of CMEC, the level of underpinning support means that New Zealand can devolve overall responsibility for the management of the classification's continuation and applicability. However, New Zealand conservation and management agencies would need to commit to actively engage long-term in the structured feedback process to help maintain, modify and improve CMEC. For example, to practically use CMEC would require immediate additional work around what components, classes and subclasses should be used to create appropriate habitat maps for New Zealand MPA planning. This work would include considering how to include habitats that are missing from the current CHMEC, as well as how to align coastal and marine habitats with New Zealand's current estuarine and terrestrial classification schemes (or ignore and replace such schemes). Thus, adopting the CMEC is just the first step in a potentially time-costly process to create modifications that allow CMEC to be fully suitable for local use. There are obvious risks associated with this process being out of full New Zealand control.

10.2.2 Numerical classifications

There are disadvantages of using thematic classifications such as CMEC, some of which are overcome by the advantages of using numerical classification approaches. Numerical classifications are usually built using biological as well as physical data, and thus represent a direct statistical linkage between environment and biotic assemblages. This linkage is often absent in thematic classifications – at least at higher classification levels typically based on physical data alone (at lower levels, classifications such as EUNIS include numerical approaches and do include linked biological data). The underlying models for numerical classifications can be validated (by internal model validation methods), and testing the accuracy of these in the field (external model validation) is relatively straightforward (though potentially expensive). Uncertainty in the underlying models and thus the classification can also be expressed, including spatially. With the availability of new data, the underlying numerical methodology means that the classifications can be easily and quickly be re-run, and thus this type of classification lends itself to continual improvement. Numerical classification approaches are flexible to using different types of data and at different scales, something that is more troublesome for thematic classification approaches. Numerical classifications can also be readily applied at different spatial scales. Numerical classifications can be built by modelling species distributions (e.g., using GDM and GF), which means the individual species distribution models underpinning the classification can also be extracted and used to identify sites for those MPAs designed to protect particular species of concern. In addition, when numerical classifications are derived from the inclusion of biological datasets as well as physical data, they can be used to not only identify distinct biotic assemblages and their habitats but also derive useful biodiversity metrics (species richness, areas of high species turnover, rarity, ecological function, etc). Such metrics can be used for describing more fully the representative habitats identified by the classification to be included in a MPA network, and they can also be used for associated monitoring and assessment purposes (e.g., examining the effectiveness of the protection measures, e.g., Leathwick et al. 2008). Derived outputs or component parts of numerical classifications are particularly well-suited for supporting efforts to identify EBSAs, VMEs, and key ecological features, and assess the potential impacts of human disturbance (and thereby provide a measure of habitat condition). Because the models that are used to build numerical classifications involve a predictive component, they can be used to assess how biodiversity might respond to future climate change, and thus MPA planning can also take these potential changes into account. All these features of numerical classifications are listed as ideal attributes for a fit-for-purpose classification approach for New Zealand's coastal and marine habitats.

10.3 Are numerical classifications the way forward?

Numerical classifications don't have to be maintained as such, but underlying data layers need to be managed and improved over time, and classifications periodically re-run to check that MPAs which have been implemented based on their results are still appropriate. However, this regular updating forces a positive aspect of MPA design – that MPA networks should be flexible to being altered in order to ensure and improve their effectiveness over time. The usefulness of numerical classifications for conservation and management has been acknowledged in New Zealand through their use to create the MEC, BOMEC, and Ross Sea bioregionalisation. Thus, New Zealand now has a considerable body of experience in creating numerical classifications of the marine environment and applying them in management contexts. Furthermore, the directions for future numerical classification developments, as well as lessons learnt about their limitations, are understood (Sharp et al. 2007, Stephenson et al. in review). Numerical classifications do have some disadvantages. If numerical classifications use only physical data, then they will share the same disadvantage of thematic classifications also based on physical data. That is, classifications based solely on environmental data are highly unlikely to be able to represent biological variability at small spatial scales (<10 km). Even when numerical classifications include biological data, they can struggle to classify areas that are heterogenous at small spatial scales, which may be important for MPA design. The amount of data required to build robust numerical classifications can be large, decision-making around which data to include/exclude can be subjective, and our understanding of which data layers are the most useful for building numerical classifications is still developing as well as how to deal best with the varying spatial scales at which such data are usually available. One of the main disadvantages of numerical classifications is that not only are the methodologies not readily understood by non-scientists, but the results (i.e., classes) are also not always intuitively understood by environmental managers and public. That is, the identified classes don't always lend themselves to obvious names or description that conform to people's perceptions of a habitat or biotic assemblage. This lack of association with a personal viewpoint can be a significant issue when engaging in stakeholder consultation to identify MPAs and design networks. This issue can be overcome if appropriate explanation documents/webinars, other web-based products, and application-tools are constructed and made freely available on the internet. Such provisions will allow non-scientists (environmental managers, public) to have the means to understand the underlying assumptions and limitations inherent in the approach to classification, as well as be able to interact with and apply the classification themselves. While this level of support and integration has been provided for thematic classifications (e.g., CMEC) it has not been done for numerical classifications in the public domain (that we know of). However, such support for the up-take of a numerical classification is possible, but significant funding and other government agency investment would be required to provide this type of essential underpinning infrastructure. That said, issues around the acceptance of classifications by stakeholders and their subsequent use in spatial management planning are not restricted to numerical classifications, and how understandable they may be. Experience to date in New Zealand has shown that the overall and specific attributes of the spatial management process in which classifications are presented is key to the success of MPA implementation. That is, it is important that the process is inclusive and participatory, that time is taken to build trust and establish shared understanding among stakeholders, there is adequate resourcing, and that multiple values and information types are included in MPA decision making (Davies et al. 2018).

11 Recommendations

It is important that before a decision is made on which type of classification to adopt for future use in MPA planning and potentially other marine management applications in New Zealand, that the objectives for the use of the classification are fully and clearly articulated. The more refinement that can be placed around these objectives, the easier it will be for the relevant government agencies (DOC, MfE, MPI) to identify the best classification approach to adopt and maintain. It is possible that when these objectives have been finalised, that more than one classification will need to be developed. At this juncture, our recommendation is for the development of numerical classifications for the coastal and marine habitats of New Zealand. In general, numerical classifications are more flexible in their construction and use, and the underlying data layers and approaches are also more readily amenable for identifying Key Ecological Areas and specific-specific protection measures. Furthermore, considerable expertise and experience in developing numerical classifications already exists in New Zealand. However, it will be essential that, once built, sufficient resourcing will be provided to support the on-going maintenance and application of the classifications. The latter was not provided for previous classifications of New Zealand's coastal and marine environments, and is one of the reasons that the CHMEC was not ultimately fit-for-purpose, that the MEC has been underutilised, and the development of a regional MEC and a final BOMEC did not progress beyond examples.

12 Acknowledgements

We gratefully acknowledge the input and collegiality of the ISAG members during the carrying out of this review. In particular, Debbie Freeman, Shane Geange, Greig Funnell, Clinton Duffy (DOC), Pierre Tieller, Constance Nutsford, Tim Riding (MfE), Richard Ford, and Ben Sharp (MPI). Special thanks are owed to Debbie Freeman for her support and patience.

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