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Functional traits as indicators of ecological integrity

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Executive summary

The Department of Conservation requires tools for assessing the ecological integrity of coastal subtidal marine habitats and wants methods and indices developed to facilitate monitoring and reporting. The objective of the project was to provide a comprehensive review of the concept of functional traits and then to develop a traits-based functional approach to the analysis of video imagery.

Functional traits that could be determined from video were derived from international literature and tested using video data collected by DOC in Port Pegasus, Stewart Island. Six broad functional categories were used (living position, growth form, body flexibility, mobility, feeding mode, and size; these represent traits that are important for vulnerability, resilience, recovery as well as aspects of ecosystem functioning). The categories are broad enough to accommodate functional traits of species living in additional physical habitats (e.g., intertidal, deep sea) that would be encountered during future surveys in New Zealand.

A Biological Traits Analysis (BTA) supplemented by estimates of spatial heterogeneity (habitat transitions) and vertical habitat complexity was used to determine functional integrity. BTA fulfil most of the requirements of a good bio-monitoring tool, being well rooted in ecological theory, demonstrated to show responses to changes in environmental conditions and human disturbances and stability across regional species pools and time, and are directly and indirectly related to ecological functions and ecosystem goods and services.

The analysis undertaken found Port Pegasus to be a region supporting a diverse array of functional traits at relatively small scales (the transect level). None of the areas sampled are markedly different to each other in composition with the exception of Noble Island and Anchorage which differ from the rest in terms of biotic groups and functional composition. Interestingly, areas with the lowest spatial heterogeneity and/or vertical habitat complexity had highest diversity of traits and vice versa. This finding adds to the complexity of determining a single index of functional integrity and suggests that functional integrity should, similar to ecological integrity, be considered as a multifaceted concept.

This study demonstrates the first step to an index of ecological integrity by successfully converting video data to functional traits data, in a way expected to be habitat independent. Ongoing analysis by NIWA's Coasts and Ocean Centre on already available data and new data will be used to confirm the habitat independence.

The video from Port Pegasus was of excellent quality, in part due to clear water in Port Pegasus, but also due to good camera gear and boat operation. However, the analysis we could do was limited by 3 factors that could be rectified in future surveys:

- Limited epifaunal taxonomic resolution.
- Lack of scaling lights or information on the length of the dropcam transects.
- Lack of knowledge of sampling design, e.g., why some areas had more transects taken than others, whether the number of transects was stratified by size of area or degree of habitat heterogeneity (either observed on the transect or that of the physical environment).

1 Introduction and scope

New Zealand is a maritime nation, with an exclusive economic zone of roughly 4 million square kilometres. The habitats and the diversity of life in New Zealand waters provide a huge range of goods and services that are used and valued by New Zealanders for recreational, spiritual, economic and cultural pursuits. For this and other reasons, it is important to conserve marine biodiversity and ensure the sustainable delivery of marine ecosystem goods and services into the future. A critical step towards achieving these conservation goals is developing robust methods for assessing biodiversity and ecological integrity at broad scales in marine environments.

In a prior report to the Department of Conservation, Thrush et al. (2012) recommended the collection of video imagery from seafloor habitats. Although it is impossible to complete detailed taxonomic assessments of an area's biodiversity from video data alone, the different types of habitats present and the functional diversity of dominant life-forms therein can be ascertained from video imagery with relative ease. Thus it was also recommended that analyses of functional traits could be performed and used to develop indices of ecological integrity.

Following the development of the initial framework by Thrush et al. (2012), underwater video footage was collected from Port Pegasus, Stewart Island by Department of Conservation staff. This included hand-held camera footage collected by SCUBA divers (100 m transects at each of 7 sites) and 10 hours of towed / drift-camera imagery (approximately 50 transects of varying lengths). The quality of the video imagery was excellent due to the clear water, slow boat speed, and height of the camera above the seabed. Basic information on where the cameras were deployed and retrieved (i.e., approximate transect positions) was also available.

The aims of the present study were threefold. The first objective was to perform a review of the concept of functional traits and its potential utility as an indicator of ecological integrity in marine ecosystems. The second objective was to scrutinise the video data collected by DOC and to develop a technique to assess functional trait diversity. The third objective was to discuss the strengths and weaknesses of the approach, and make recommendations for future measurement of functional trait diversity as an indicator of ecological integrity in marine habitats.

2 Functional traits and ecological integrity

2.1 Ecological integrity

The requirement to protect marine habitats and biodiversity is articulated in regional, national and international guidelines and policies, e.g., Convention on Biological Diversity http://www.cbd.int/convention/text/. Globally, the main objective of different legal frameworks and policies is to maintain a good environmental or ecological status for marine waters, habitats and resources (also referred as ecological integrity, i.e., the necessity to safeguard the self-organizing capacity of ecosystems). Ecological integrity is thus a particularly useful concept for conservation management, allowing identification of threats to, and responses of, specific ecosystem components. This can be done at a variety of spatial scales (e.g., within a marine reserve, or between a network of reserves).

There are several definitions of ecological integrity (see Thrush et al. (2012) for a summary). The definition that DOC is using is that of Lee et a.l (2005) “The full potential of indigenous biotic and abiotic features, and natural processes, functioning in sustainable communities, habitats, and landscapes. Ecosystems have ecological integrity when all the indigenous plants and animals typical of a region are present, together with the key major ecosystem processes that sustain functional relationships between all these components, across all of the ecosystems represented in New Zealand”.

The concept of ecological integrity thus takes into account the structure, function and processes of marine ecosystems and integrates these properties with human uses in the area (see Table 2-1; Jørgensen et al. 2005; Müller and Burkhard 2007). Therefore, indicators for the assessment of ecological integrity have to reflect these processes and structures. Some of the existing indices (mainly developed under the framework of the WFD and MSFD) are based on biological traits of the marine species or on functional components of the ecosystems.

Table 2-1: Suggested attributes for indicators of ecological integrity.

UNESCO 2003	Scientifically sound meaning, representative of an important environmental aspect for society, valuable information, rapidly understandable meaning, provides information to answer important questions, assist decision making by being efficient and cost-effective.
Reza and Abdullaha 2010	Multi-scale, adjustable, relevant and helpful, sensitive to small variations in the stressor, simple, flexible, measurable, cost-effective, policy relevant, comprehensive.
Schallenberg et al. 2011	Simple in terms of sampling and analysis, geographic cover, represent nativeness and pristineness, sensitivity to pressure gradients, temporal variability, normalisation to reference conditions.
Thrush et al. 2012	Easily measured, sensitive to stressor on the system, respond to stress in a predictable manner, indicative of reversible changes, integrate across key gradients, under reasonable sampling scenarios do not behave erratically, relate to management goals, scientifically defensible.
Rice et al. 2012	Readily understood by decision makers, based on existing monitoring, reflect the actual state of ecosystems, representativeness, availability of historical data, specific to stress, ability to set reference points, sensitivity to the stressor.

As a consequence of anthropogenic activities, ecosystems may rapidly shift from desired to less desired states in terms of their capacity to generate ecosystem services (Folke et al. 2004); indicators of ecological integrity must identify these shifts. But ecological integrity cannot be exclusively defined as a pristine environmental status, but rather a status when human uses are sustainable (Rice et al. 2012). The definition of "sustainability" of human uses can be controversial, and baselines of reference status should be defined for each habitat or location prior to the implementation of monitoring programs. Moreover, as different types of ecosystem are interconnected, comprehensive monitoring and evaluating criteria are needed for measuring integrity at regional levels (Reza and Abdullaha 2010). To achieve this, broad spatial knowledge of the habitats and associated biological communities overlapped with human uses is essential and this is currently lacking from most regions world-wide (Pickrill and Todd 2003; Holmes et al. 2008; Stelzenmüller et al. 2010).

2.2 Presently used indicators for status and change

Indicators are the scientific response to governmental need for reliable and accurate information on a system's conditions (Heink and Kowarik 2010; Van Hoey et al. 2010). An indicator is, in essence, a formulation/calculation that is designed to summarise copious, complex, scientific information in a simple, condensed, comprehensible way. Accurate indicators can be extremely useful for tracking trends over time in response to stressors or management actions, and they can be very effective for communicating the significance of trends to non-scientists. There are more than 200 indicators that aim to describe marine ecosystem health (Rice 2003; Diaz et al. 2004; Borja et al. 2008; Van Hoey et al. 2010). These indicators, which can extend from the cellular to the community level, simplistically report on the state of the ecosystem, or a part of it, to a range of stakeholders with diverse interests and backgrounds (e.g., scientists, policy makers, the media, and the general public). Particularly in Europe in the context of the Water and Marine Framework Directives (established in 2000, Directive 2000/60/EC, and 2008, Directive 2008/56/E), there has been an explosion of indices of "ecosystem health", "good environmental status" or "ecological integrity" (note the ambiguous definitions). However, there is generally a lack of consensus and most indices are highly location- and stress-specific. Borja et al. (2009) underlined the increasing number of indices and lack of consensus.

Over the last years, the interest in using biotic indicators to assess marine environments has increased radically. This increasing interest is mostly due to the need for new tools for assessing the status of marine waters, which is required by regulations like the Clean Water Act and Water Framework Directive (Dauvin et al. 2010). Recently developed indices, such as AMBI, BENTIX, BQI and BOPA (Borja et al. 2000; Rosenberg et al. 2004; Simboura et al. 2005; Dauvin et al. 2007; Pinto et al. 2009), are based on dividing macrobenthic species into previously defined ecological groups in relation to the stressor, and then determining the respective proportion of the different groups in the macrobenthic communities. Most of these indices examine the relative decrease of sensitive species confronted with increasing organic matter in the sediment, or the increase of species that are resistant or indifferent to such increases (i.e., tolerant species). The final outcome of these indices is the integration of multivariate data into a single numeric score (or category) that can be interpreted by a non-specialist within a 'good' versus 'bad' continuum (Diaz et al. 2004). The main problem related with these indices is the inclusion of stress-tolerant species that may also be tolerant of natural stressors (see Table 2-1: the indicators must be tightly linked to the stressor and respond in a predictable manner). Most of the existing marine ecosystem indicators are based on benthic invertebrates although other indicators have also been described and related to biological traits of fish or trophic levels in the system (e.g., Rochet and Trenkel 2003; Salas et al. 2006). Diaz et al. (2004) listed all the indices proposed to that date (including marine and freshwater ecosystems), and for the 64 compiled indices, there was a tendency for investigators to embrace broadly similar goals and exploit comparable methods of metric assembly. He interpreted this as evidence for the duplication of methods in the published literature, rather than the independent coevolution of different indices, and to the fact that there is little acceptance of any specific metric by environmental managers or scientists. Almost 10 years after this review paper, things have not significantly changed. Borja et al. (2009) stated that the next decade needed to be characterized by a consolidation in indices and that these should go through a reliable validation process (e.g., trial through strong stress gradients).

Generally the indices proposed to date are related to specific sub-components of marine ecosystems (e.g., benthic species, fish, phytoplankton) and in response to single factors (e.g., organic enrichment, sewage, fishing, dredging). However, to measure ecological integrity the selected indicators should encompass various ecosystem attributes in a holistic way. For example, Rice et al. (2012) selected as indicators of sea-floor integrity: (i) type, abundance, biomass and extent of relevant biogenic substrate; (ii) extent of the seabed significantly affected by human activities for the different substrate types; (iii) presence of particularly sensitive and/or tolerant species; (iv) multi-metric indices assessing benthic community condition and functionality, such as species diversity and richness, proportion of opportunistic to sensitive species; (v) proportion of biomass or number of individuals in the macrobenthos above some specified length/size; and (vi) parameters describing the characteristics of the size spectrum of the benthic community. Reza and Abdullaha (2010) proposed a “Regional Index of Ecological Integrity” that included physical, chemical and biological components. Halpern et al. (2012) also proposed an “Index of the Ocean Condition”, but the over-generalisation necessary for such large scales reduced index accuracy and utility. A compromise between accuracy and generalisation must be achieved if indicators are to be used to manage our marine ecosystems. Adequate indicators must principally meet three criteria: they must be easily measured and recorded in a cost-efficient way, they must respond to stress in a predictable manner, and they must be clearly understood by stakeholders (Hiddink et al. 2006a; de Juan et al. 2009). However, ideally any indicator should also encompass the criteria included in Table 2-1. Thrush et al. (2012) recommended a number of indicators that could be used to indicate ecological integrity in marine systems. Some of these indicators centred around functioning of the benthic environment, based around functional traits of seafloor organisms and were to be cost-effectively measured by video.

2.3 Functional traits

Many studies have highlighted the complexity and unpredictability of ecological systems (Mouillot et al. 2006). One way to overcome this problem in order to assess ecological integrity is a simplification of communities through partitioning of species into a variety of guilds, functional groups or functional types (Simberloff and Dayan 1991; Mathieson et al. 2000; Jauffret and Lavorel 2003). The division of species into groups with shared behavioural traits (e.g., mobility), or which exploit a common resource base (e.g., feeding guilds), has for decades been used to analyse structure and function of biological assemblages (e.g., Fauchauld and Jumars 1979; Huey et al. 1984; Bonsdorff and Pearson 1999). However, the recent focus on the relationship between biodiversity and ecosystem functioning has increased the range of traits focussed on. Many authors have acknowledged the importance of functional traits, instead of species richness *per se*, for better understanding the role of biota in ecosystem structure and functioning (Dıaz and Cabido 2001; Loreau et al. 2001; Mouillot et al. 2006). Plants, as primary producers, represent the basal component of most ecosystems and were the focus of the first terrestrial studies, which observed that primary production exhibits a positive relationship with functional-group diversity (Loreau et al. 2001). Subsequent studies expanded to encompass other systems, including marine ecosystems. Currently, species classification into functional groups in marine systems is already well tested for fish (e.g., Dumay et al. 2004) and benthos (e.g., Pearson 2001; Mugnai et al. 2003).

A functional group is a set of taxa that share a range of similar attributes and have analogous effects on major ecosystem processes (Bonsdorff and Pearson 1999). This approach is based on the assumption that taxonomically unrelated organisms might have evolved similar biological adaptation, leading to functional similarity (Doledec and Statzner 1994; Usseglio-Polatera et al. 2000). As different species may perform similar functions, this approach can be used to compare among regions that may have large differences in species composition (Bremner 2008). It also is expected to overcome a number of problems associated with species data. For example, the observed taxonomic richness is an underestimation of the true taxonomic richness (i.e., real number of species in a sampling site) and the error involved will depend on sampling effort (Ugland et al. 2003; Bady et al. 2005). Moreover, the identification of individuals to the species level is time-consuming and is not an easy task for many taxonomic groups (e.g., due to small-size species or subtle morphological differences or their cryptic nature (Knowlton 1993); consequently, many taxonomic errors may occur in ecological assessment studies (Mouillot et al. 2006). Cochrane et al. (2012), by applying the functional components approach to benthic studies in the Barents Sea, observed that this approach was effective to overcome discrepancy in taxonomic names across studies. Therefore, this approach already covers some of the criteria specified for “ecological integrity” assessment (e.g., widely applicable and easy monitoring).

The underlying concept of functional diversity and functional traits is that functional structure of a community and its effects on other aspects of the ecosystem can be represented by a set of traits related to longevity, reproduction, behavioural and morphological characteristics of the species comprising the community. The traits and their interactions determine the functioning and stability of communities and ecosystems (Figure 2-1; Loreau et al. 2001), and provide information about how communities respond to environmental stress (Lavorel and Garnier 2002). Functional traits have already been used in many freshwater and marine ecosystems assessment studies, which usually focused on invertebrates (e.g., Townsend and Hildrew 1994; Usseglio-Polatera et al. 2000) for freshwater ecosystems; (Bremner et al. 2003) for marine ecosystems. One reason for this focus is that invertebrates, and particularly benthic invertebrates, are commonly selected for use in index construction due to their regular use in monitoring, the existing knowledge and databases and the good identification guides for their taxa and their close relationship with the local environment (Hewitt et al. 2004; Thrush et al. 2004; Hewitt et al. 2005).

In summary, a trait-based response-and-effects framework is a very valuable tool for incorporating community dynamics into predictions of environmental change (Suding and Goldstein 2008). This framework involves integrating two components:

1. how a community responds to change, and
2. how that changed community affects ecosystem processes.

But to achieve this, there is a need to incorporate previous knowledge on the ecosystem structure and functioning.

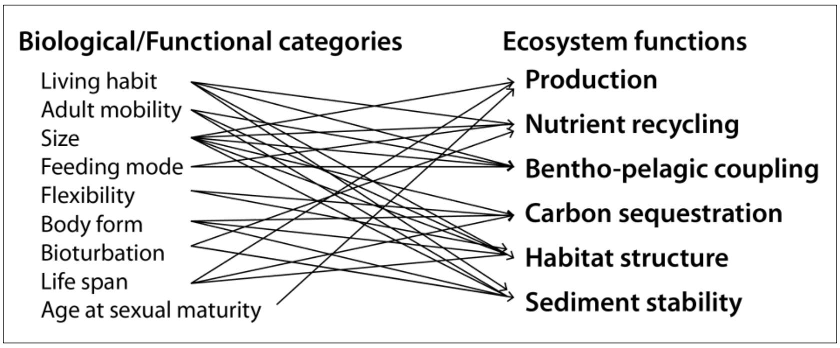

Figure 2-1: Examples of biological/functional traits linked with important ecosystem functions.

There has been some debate about the traits that should be used to assess functional diversity and structure (Petchey and Gaston 2002). Common to most studies is a characterization of the functional strategies of the common species in a community by identifying a relevant combination of functional traits (Mouillot et al. 2013)). In this context, one of the most widely used techniques to assess the functional components of ecosystems in a disturbance framework is the Biological Trait Analysis (BTA), which uses morphological, life history and behavioral species characteristics to indicate aspects of their ecological role and performance of ecosystem functions. As roles performed by organisms are important in regulating ecosystem processes, BTA approach is useful to assess these processes and related ecosystem services (Bremner et al. 2006). While the actual information is strictly more related to structure than function, it is generally accepted across a variety of ecosystems types that when the biological traits are carefully chosen they extrapolate to function. For example in marine systems, deep burrowing fauna increase the oxygen flow into the sediment and extend the total zone of denitrification, stimulating nutrient cycling as an important ecosystem service (Beaumont et al. 2007; Norling et al. 2007). Other traits such as body size and life span are related to ecosystem productivity (Jennings et al. 2001) and to food production services (Beaumont et al. 2007; see Figure 2-1), while dispersal information is related to the ability of organisms to recover from a disturbance.

2.4 Functional trait responses to marine stressors

The first ecosystem impact assessments were based on species composition (e.g., de Groot 1984; Thrush et al. 2004; Kaiser et al. 2006). This strategy increased our knowledge on community responses to anthropogenic impacts, but it provided limited information on ecosystem functioning and, additionally, the studies were often geographically limited (Bremner 2008). Moreover, community diversity or richness by itself is difficult to relate to an ecosystem disturbance level because the diversity–disturbance relationship remains controversial and its shape is often unimodal with the highest diversity being observed for medium levels of disturbance (Mackey and Currie 2001). Bustos-Baez and Frid (2003) examined the role of key species as indicators in contaminated sediments, and observed the lack of generality these species showed (i.e., inconsistency across sites included in a meta-analysis) and encouraged the inclusion of functional components of ecosystems for impact studies. Changes in the functional composition of communities represent the adaptations of the organisms to the environment and their response to stress (de Juan et al. 2007). Therefore, human impacts on the environment cause not only a general decline in diversity, but also functional shifts as sets of species with particular traits are replaced by other sets with different traits (Mouillot et al. 2013). In general, functional trait analysis has proved to be a useful approach, highlighting community responses which are difficult to detect using species approach (e.g., Frost et al. 1995; Bremner et al. 2003; Tillin et al. 2006; de Juan et al. 2007).

Pearson and Rosenberg (1978) demonstrated that the response of benthic fauna to organic enrichment depends on the biological traits of the organisms. Recent advances in the application of species biological traits to assess the functional structure of communities have provided an approach that responds rapidly and consistently across taxa and ecosystems to multiple disturbances. Importantly, trait based metrics may provide advanced warning of disturbance to ecosystems (Mouillot et al. 2013). Theoretically, we predict that functional diversity decreases with increasing environmental constraints or stress (Mouillot et al. 2006). When environmental constraints increase, coexisting species are more likely to be similar to one another as the species that make it through the environmental filters are likely to share many biological/ecological characteristics (Statzner et al. 2004). Using traits for quantifying species differences may yield functional diversity, which in turn allows a shift from the usual monitoring of species towards the monitoring of ecosystem function (Ghilarov 2000). Quantifying and predicting functional community structure within a context of increasing disturbance intensity and frequency is now required to anticipate the potential loss of ecosystem services that is indisputably associated with biodiversity erosion (Cardinale et al. 2012).

Several studies in the last decades have applied BTA for the study of marine ecosystems, including the evaluation of ecological status. For example, Bremner et al. (2006) used BTA to study variations in the functionality of benthic communities linked to environmental variability. Paganelli et al. (2012) studied the BTA of benthic communities over a gradient based on distance from the River Po. Frid et al. (2008) used BTA to aid in the designation and management of MPAs. de Juan et al. (2009) analysed changes in biological trait components due to trawling. Overall, results showed the potential for BTA to be used as ecosystem indicators in impact studies (see examples in Table 2-2).

2.5 Assessing functional diversity

Many indices have been proposed, tested and accepted by the scientific community as effective tools to evaluate ecological quality in marine environments (see Occhipinti-Ambrogi et al. 2005; Salas et al. 2006) for reviews; Table 2-3). Indices based on indicator species, such as AMBI (Borja et al. 2000), BENTIX (Simboura and Zenetos 2002), BQI (Rosenberg et al. 2004) and BOPA (Dauvin and Ruellet 2007) are based on the classification of benthic taxa into ecological groups according to their sensitivity/tolerance under varying degrees of disturbance. While some of these indicators have been widely used e.g., AMBI, M-AMBI, IBI (see Muxika et al. 2005; Callier et al. 2008; Bakalem et al. 2009), they are subjective to a certain extent, requiring classification of sites into different habitats (e.g., sandy cf muddy) or the decision as to what to do with species with unclassified sensitivities (e.g., remove or classify as nearest taxonomic level). In addition, disparities in the environmental classification by the different indices have been determined. At least some of these problems may be a reflection of differential response of species to stressors in different environments. They also can only be considered to be related to functioning in a very broad sense, that of considering individual species responses.

Table 2-2: Example of marine impact studies that focus on functional/biological traits. BTA is used to indicate that a combination of biological traits were considered, e.g., size, feeding mode.

Functional traits	Organisms	Impact	Region	Reference
Size	macrobenthos	Hypoxia	-	Diaz and Rosenberg 1995
Trophic groups	Macrobenthos	Pollution	Dublin Bay	Roth and Wilson 1998
Size	Megabenthos	Fishing	Bering Sea	McConnaughey et al. 2005
Size	Benthos	Bottom fishing	North Sea	Hiddink et al. 2006b; Robinson et al. 2010
BTA	Benthos	Bottom fishing	North Sea	Tillin et al. 2006
BTA	Megabenthos	Bottom fishing	Bay of Fundi	Kenchington et al. 2007
BTA	Coral ecosystems	Heavy metals	Indonesia	Rachello-Dolmen and Cleary 2007
BTA	Benthos	Bottom fishing	NW Mediterranean	de Juan et al. 2007; de Juan et al. 2009
Functional Diversity	Macrobenthos	Aggregate extraction	North Sea	Cooper et al. 2008
BTA (size and tolerance groups)	Macrobenthos	Organic enrichment	Italian coastal lagoons	Marchini et al. 2008
Bioturbators	Benthos	Trawling	Oslo fiord	Olsgard et al. 2008
Size and life span	Benthos	Trawling	North sea	Robinson and Frid 2008
BTA	Benthos	Organic enrichment	Baltic Sea	Villnäs and Norkko 2011
BTA	Benthos	Fishing	NW Mediterranean	Barberá et al. 2012
BTA	Macrobenthos	Eutrophication	Adriatic Sea	Paganelli et al. 2012

A few indicators have been developed based on functional characteristics (Table 2-3), but these have generally been developed to link strongly a response to a stressor(s). For example, between 2009-2012 a New Zealand functional index was developed to assess functionality of intertidal communities under stress from increasing sediment mud content and heavy metal contamination (Rodil et al. 2013).

With no standard indices already developed, using functional traits as an indicator of ecological integrity requires consideration of three important aspects, summarised below and then expanded on in the following sections. (1) Which functional traits to use? (2) Which techniques to use in the assignation of traits? (3) Which measure(s) of functional diversity to use?

Table 2-3: Example of indices[1] currently described for marine ecosystems. Only the last five examples are clearly focussed on functional traits, the others only separate species based on sensitivity to an impact or include total diversity. Country of region is not given were multiple countries are involved.

Indicator	Organisms	Attributes	Stress	Region	Reference
B-IBI	Benthos	Multi-metric, including sensitive taxa	Pollution	Chesapeake Bay, USA	Weisberg et al. 1997
BRI	Benthos	Sensitive taxa	Pollution	California, USA	Smith et al. 2001
BCI	Benthos	Tolerant taxa	Pollution	Gulf of Mexico, USA	Engle and Summers 1999
APBI	Benthos	Tolerant taxa	Pollution	Gulf of Maine, USA	Hale and Heltshe 2008
BOPA	Benthos	Opportunistic polychaeta amphipoda	Pollution	English Channel	Dauvin and Ruellet 2007
AMBI	Benthos	Sensitive vs. opportunist taxa	Organic enrichment	Developed in Iberian peninsula, adapted to many regions (e.g., Florida, USA)	Borja et al. 2000
BENTIX	Benthic macrofauna	Sensitive vs. opportunist taxa	Pollution	Mediterranean	Simboura and Zenetos 2002
IBI	Benthos	Sensitive vs. opportunist taxa	Pollution	Chesapeake Bay, USA	Diaz et al. 2003
Ecological evaluation index	Macrophytes	Sensitive vs. opportunist taxa	Stress	Coastal areas	Orfanidis et al. 2003
Ecofunctional quality index	Benthos	Diversity, biomass, sensitive taxa	Pollution, eutrophication	Italian coastal lagoons	Fano et al. 2003
Multivariate-Ambi	Benthos	Sensitive vs. opportunist taxa. biomass and diversity	Pollution	European coasts	Muxika et al. 2005
ITI	Benthos	Trophic categories	Changed environment	California, USA	Word 1978
FINE	Macrofauna	FD, biomass	Pollution	Mediterranean Coastal lagoons	Mistri et al. 2008
TBI	Macrofauna	BTA	Mud and heavy metals	Estuaries in the Auckland region (NZ)	Rodil et al. 2013
TDI	Benthos	BTA	Trawling	Mediterranean	de Juan and Demestre 2012
BF1 and BF2	Benthos	Size	Organic enrichment	Gulf of Mexico	Rakocinski 2012

2.5.1 Deciding on functional categories and traits to include

Biological traits can be grouped into functional categories (e.g., feeding type; adult movement; Table 2-4). These categories each contain a set of biological traits which each

[1] Table 2 in Diaz et al. (2004) lists all the indices proposed to that date; Salas et al. (2006) also list indices and compare them in estuaries/harbours of the Iberian peninsula; on page 17 of Borja and Tunberg (2011) there is a compilation of all indices for invertebrates (mostly related to pollution).

species will exhibit one or more of (e.g., feeding category: suspension; deposit, grazer, predator etc).

Table 2-4: List of biological traits for a number of categories frequently used for assessing the ecological integrity of marine benthic systems. Main source, MarLIN – Biological Traits Information.

Adult movement	Flexibility	Potential size	Feeding mode	Living habit	Position	Growth form
Swimming Crawling Burrow Sedentary Attached	None Low High (>45º) <u>Option: fragility</u> Fragile Intermediate Robust	Very small <1 cm Small 1-5 cm Medium 5-15 cm Large >15 cm	Suspension feeder Deposit feeder Predator Scavenger Grazer Omnivorous	Tube-dweller Burrow-dweller Free living Attached	Endobenthic Epibenthic	Crustose Globose Arborescent Vermiform Cushion Tubiculose Turbinate Stellata Bivalvia Articulate Pisciform Bed forming Erect
Age at sexual maturity	**Asexual reproduction**	**Life-span**	**Type of larvae**	**Regeneration potential**	**Reproduction frequency**	
< 1 yr >1 yr Late	Yes No	< 1 yr 1-2 yr 3-5yr >5 yr	Direct development Short planktonic (<1 week) Long planktonic (>1 week)	Yes No	Continuous 2+ events/yr 1 event/yr < 1 event/yr	

One constraint of the BTA approach is the need to simplify the set of biological traits used, as measures containing disparate bits of information quickly become complex. For example the MarLin data base includes over 50 traits and many of these are not easily assigned to most species, e.g., life-history traits (Costanza and Mageau 1999). A starting point is to list trait categories and their functional attributes (Table 2-5) and focus on the categories that cover a range of attributes and are reasonably well known. Then, if any specific stressors are considered to be of importance, to include trait categories that would be expected to respond to those stressors. For MPAs, characteristics known to be affected by removal of fish species or by bottom trawling such as flexibility/fragility, living habitat, position and growth form would be particularly important. Life-span and potential size are also known to be affected (Thrush et al. 1998) but unfortunately there is usually limited information available about these characteristics.

To determine the effect of stressors, the focus should be on traits that provide mechanistic explanations for stress responses or resultant ecosystem conditions (Statzner and Beche 2010). Conversely, to assess ecosystem integrity, traits selected need to be related to a diverse range of functions and ecosystem components. In order to best inform conservation management, a mix of traits that will respond in specific known ways to identified threats, that can be expected to show a response to management activities (e.g., the creation of a reserve) at an appropriate space and time scale, and that represent a range of functions and ecosystem components need to be selected.

Table 2-5: Trait categories most frequently used in assessing ecological integrity and the functional components they most relate to. The table is sorted by the degree of information generally known: Y = generally known, I= sometimes known, N = often unknown.

Trait category	Functional component	Information generally known
Adult movement	Recovery, resilience, vulnerability, fluxes	Y
Flexibility/fragility	Vulnerability	Y
Feeding mode	Trophic transfers, vulnerability, fluxes	Y
Living habit	Vulnerability, diversity, species interactions, fluxes	Y
Position	Vulnerability, diversity, species interactions, fluxes	Y
Growth form	Diversity, species interactions	Y
Life-span	Diversity, vulnerability	I
Potential size	Diversity, vulnerability, fluxes	I
Age at sexual maturity	Diversity, vulnerability	I
Asexual reproduction	Recovery	I
Type of larvae	Recovery	N
Regeneration potential	Recovery, resilience, vulnerability	N
Reproduction frequency	Recovery, resilience, vulnerability	N

2.5.2 Techniques for assigning functional traits

There are a couple of techniques available for assigning species to biological traits. Firstly, traits within a category are ranked, for example 5 ranks of increasing size or increasing mobility. Secondly, and more generally, fuzzy coding is used to allow the species to vary in the degree in which it exhibits affinity to a specific trait within a category (Chevenet et al. 1994). For example, a species may predominantly exhibit deposit feeding but may also occasionally filter feed, and thus be assigned a 3 for the trait deposit feeding, a 1 for filter feeding and a 0 for other feeding traits (Bremner et al. 2003). Another example of fuzzy coding is to use probabilities, e.g., the species has a 0.75 probability of deposit feeding, 0.25 of suspension feeding and 0 of any other feeding traits (Hewitt et al. 2008); in this case the sum for a species across all traits within a category is 1.

2.5.3 Measures of functional diversity

Assessing functional status *per se* initially focussed on simple estimates of functional diversity or richness, e.g., the number of functional traits observed. Since then, an increasing number of assessment methods have developed. These fall into 2 major groups, based on whether information on overall presence/absence, biomass or abundance of a functional trait is considered, or whether the number of species, their identity and their biomass (or abundance) are also considered.

Estimates of functional composition and dissimilarity calculated from distance measures and ordination techniques (Doledec and Chessel 1994; Bremner et al. 2003) fall into the first group. However, an increasing number of techniques have been developed to assess the number of species that represent each function and the way abundance or biomass is spread within and between functions (Petchey and Gaston 2002; Botta-Dukát 2005; Petchey and Gaston 2006; Mouillot et al. 2013). These methods have been developed mainly to incorporate ideas about functional redundancy: the more species that represent a function, the more likely it is to be maintained in the face of changing environmental conditions (the insurance hypothesis, Yachi and Loreau 1999). Below is a list of different measures that have been used.

- Within-trait measures such as abundance, number of taxa, Margalef's richness, Pielou's evenness, Shannon-Wiener diversity index (Hewitt et al. 2008). These are calculated on a single trait and can be easily compared between studies assuming that sample size and taxonomic resolution are similar. The abundance (or biomass) of a trait and the number of different organisms representing that trait are particularly important for the functioning and resilience of the trait. The greater the number of species representing a trait, the more likely they are to have different responses to stressors and to exhibit different temporal and spatial patterns, thus increasing the potential resilience of the trait. Evenness and Shannon-Wiener diversity are two ways of encapsulating both of these aspects in a single number and are frequently used, although neither of these have yet been directly associated with trait functioning.
- Between- and within-trait measures.
  - Functional divergence: the proportion of total abundance supported by species with the most extreme trait values within a community (Villeger et al. 2008).
  - Functional diversity: the summed branch lengths of the dendrogram constructed from functional differences (Petchey and Gaston 2002).
  - Functional diversity: the distribution of species and their abundances in the functional space of a given community (Laliberte and Legendre 2010).
  - Functional dispersion: the abundance-weighted deviation of species trait values from the centre of the functional space (Laliberte and Legendre 2010).
  - Functional evenness: the regularity of the distribution and relative abundance of species in functional space for a given community (Villeger et al. 2008).
  - Functional identity: the mean value of functional traits, weighted by abundance, across all species present in a given community (Garnier et al. 2004).
  - Functional richness: the volume of multidimensional space occupied by all species in a community within functional space (Cornwell et al. 2006).

- Functional entropy: Rao's quadratic entropy based on the relative abundances of species and the pairwise functional differences between species.(Botta-Dukát 2005).

These between and within-trait measures suffer from one major drawback for assessing status and monitoring; in order to be comparable between sites or times, all sites/time data must be analysed together. That is, if a new site or time was added, all the previously calculated indices would need to be recalculated and it would not be possible to predict whether this would result in minor or major changes to previously reported values. This is because the addition of new species or of sites with different community compositions alters the position of previous sites in the ordination space. For this reason, we do not use them within this report. However, they can be highly useful for focussed impact assessments where samples are collected either before and after or along a gradient of stress (Mouillot et al 2013) and are intended to be analysed and reported together.

2.6 Using video surveys

Another method of simplifying the "complexity of ecosystems", that integrates well with functional trait analysis is to target selected ecosystem components. Here we focus on the use of benthic organisms visible by video.

Targeting of benthic organisms (flora and fauna) has many advantages: they are relatively non-mobile and therefore useful for studying the local effects of perturbations; some species are long-lived and would represent historical disturbance; their taxonomy and their quantitative sampling is relatively easy; and there is extensive literature on their distribution in specific environments and on changes related to various stresses (Borja et al. 2008; Bremner 2008). Other attributes of BTA of benthic organisms that make them useful as indicators is that these organisms are relatively easy to classify into biological traits (e.g., in a previous work by de Juan and Demestre (2012) over 200 megabenthic species were able to be classified into BT with help of taxonomic guides, on-line data bases (e.g., MarLIN, WORMS, fishbase) and expert´s judgment).

While videoing the seafloor only allows us to focus on larger (usually >4 cm), epibenthic (visible) flora and fauna, it has the advantage of allowing a large spatial coverage and an extended overview of the habitat and faunal communities. Other advantages of video surveying include the recording of "real" images of the seafloor that are more likely to capture the behaviours of larger mobile species that are missed by other sampling methods, thus helping build the associated BTA database.

In particular, the targeting of larger epibenthic flora and fauna integrates well with many conservation initiatives. In many cases, conservation programs, guidelines and policies require the distribution and status of different habitat types to be described and monitored. Traditional monitoring of habitats and associated species using grabs and trawls is costly and labour intensive. Recent advances in acoustic and video techniques offer the ability to survey large areas in a rapid non-destructive way (e.g., Lo Iacono et al. 2008; Lambert et al. 2013). Videos have proven useful for quantifying the distribution, structure, abundance and health status of benthic organisms in a variety of ecosystems (e.g., Fosså et al. 2002; Kendall et al. 2005; Martín-García et al. 2013). Several studies have used video transects to evaluate the "health" of ecosystems, related to trawled areas (e.g., Collie et al. 2000a; Smith et al. 2001), effects of marine aggregate dredging (Cooper et al. 2008), MPA effects (Lindholm et al. 2004) or for the detection of vulnerable habitats (e.g., Jones and Lockhart 2011). Eyre and Maher (2011) generated maps of benthic ecosystem processes and overall functional value. These maps were used to identify ‘‘hot spots’’ of functional value that have high conservation value.

There are a number of studies of benthic communities by video transects that highlight the ecosystem components identified, e.g., Collie et al. (2000b) who observed that anemones, tubeworm, sponges, plant-like animals and hermit crabs were more abundant in undisturbed areas, whereas flounder, starfish and scallop were more abundant in trawled areas. Lindholm et al. (2004) performed video surveys to designate MPAs and identified habitat categories including featureless sand, rippled sand, sand with emergent fauna, gravelly sand without attached-erect fauna, gravelly sand with attached-erect fauna, bivalves, biogenic depression and sponges. Holmes et al. (2008) recorded the distribution of marine benthos with underwater video identifying macroalgae, rhodoliths, kelp, sessile invertebrates, sponges, ascidians and soft corals. In New Zealand, Hewitt et al. (2004) recorded different features along video transects, including fauna (bivalves, hydroids, sponges, tubeworms, ascidians, bryozoans, etc.,) flora (kelp, diatom mats, coralline algae), seafloor microtopography (burrows, holes, tracks, pits) and sediment characteristics (coarse particles, shell hash, sand, mud). Diversity (spatial variability) in all four of these elements has been shown to relate to diversity of infauna in soft-sediments (Thrush et al. 2001).

Not all traits are necessarily important in all habitat types (e.g., rocky substrates and soft-sediments); but at present we propose a standard set of traits to be used across all habitats (Table 2-6). The potential for these traits to reflect aspects of ecosystem functioning is given in Table 2-5, however, three specific aspects are mentioned here, the first two of which are only important in soft-sediments.

- The ability to stabilise sediment is particularly important in soft-sediment systems, decreasing bed erosion and increasing the potential for other organisms, more likely to be found associated with hard substrates, to settle and live.
- The converse of this - sediment destabilisation - is an important aspect of benthic-pelagic coupling in most regions of the world, but is particularly important where contaminants and terrestrial sediments accumulate, as sediment destabilisation increases their dispersion.
- Body size has specifically been included as observed size rather than potential size as it is generally easy to measure from video and amenable to inter-calibration procedures; it is comparable across taxa, guilds and sites (Mouillot et al. 2006), and, as a community feature, it is expected to vary along disturbance gradients, according to energetic and ecological constraints (Basset et al. 2004). Body size is an important trait determining, to a large extent, the type and the strength of ecological interactions to which individuals are subjected (De Roos et al. 2003). Changes of benthic community biomass under disturbed conditions are well documented in benthic ecology (Pearson and Rosenberg 1978).

Table 2-6: List of biological/functional traits of benthic communities that can be recorded in video surveys.

	Megafauna	Flora
Position/living habitat	Epibenthic, attached, infauna (endobenthic)	Epibenthic, attached
Growth form	Crustose/encrusting, globose/cushion, arborescent, tubiculose, bed forming, erect, vermiform, turbinate, stellate, bivalvia, articulate, pisciform, burrow-dweller	Foliose, laminar, arborescent
Flexibility	Soft body, rigid, calcified	Soft body, rigid
Mobility	Swimming, crawling, burrow, sedentary	Sedentary
Size	Small, medium, large	Small, medium, large
Feeding	Suspension feeder, deposit feeder, predator, scavenger, opportunistic, grazer	Primary producer
Sediment stabilisation	Stabiliser, destabiliser, no effect	Stabiliser, no effect

We also suggest including four other aspects that are not directly based on biological traits of a specific species but which are functionally important.

- Density is important in determining the contribution of a species to ecosystem function. Variations in density, the size and placement of dense patches of specific traits (e.g., suspension feeding bivalve beds, rhodolith beds, seagrasses etc.) will all have effects on ecosystem function (Hewitt et al. 2004) and can be identified from video images.
- Site heterogeneity is important method for summarising ecosystem diversity and can be assessed at a number of scales (e.g., patch size of individual habitats, patch fragmentation across a transect, site or region).
- Soft-sediment micro-topography is not only important when making an assessment of 3-dimensional habitat complexity, but also as a surrogate for bioturbation (Lohrer et al. 2004).
- Habitat features are often generated by the resident biota, especially in soft sediments (Zajac et al. 2003); therefore, if detailed information of functional traits in the community is lacking, data on habitat complexity may be sufficient to assess state and impact, based on the assumption that complexity (e.g., living habit, growth form, size) will be linked to biodiversity and function (e.g., Bolam et al. 2002; Lambert et al. 2013). At the least we suggest calculating a measure of habitat complexity (3-dimensionality) based on size and complexity of form of sedentary species and sediment micro-topography.

3 Case study in Port Pegasus

Video footage was collected from 10 general areas in Pegasus Bay (Anchorage, Noble Island, Pigeon, South Arm, Knobs, Disappointment, Sylvan Cove, Inside Pearl, Northern Arm and Twilight Cove). Multiple sites—transects where video footage was recorded—were sampled in 8 general areas, with different numbers of sites (and different collection methods) in different areas (Table 3-1). Data were collected either by a diver swimming a transect; a boat drifted drop-camera; or a ROV. Some sites were sampled by both a diver transect and a dropcam transect.

Table 3-1: Areas sampled by video in Port Pegasus. Different collection methods and time length of videos is given.

Area	Collection Method	Number of sites	Range of video length (mins)
Disappointment	Diver	2	9 to 15
	Dropcam	5	6 to 13
Sylvan Cove	Dropcam	8	6 to 12
Inside Pearl	Diver	2	10
	Dropcam	5	9 to 13
Knob	Diver	3	15 to 16
Anchorage	Dropcam	1	12
North Arm	Diver	1	19
	Dropcam	5	5 to 14
Noble Island	Dropcam	4	9 to 24
Pigeon	Diver	3	13
	Dropcam	1	31
South Arm	Dropcam	9	5 to 21
Twilight Cove	ROV	1	10

3.1 Method development

3.1.1 Spatial heterogeneity

A first step in designing an effective video survey is to evaluate the habitat heterogeneity of the region in question (Van Hoey et al. 2010). In most transects, several distinct transitions between dominant organisms (habitat transitions) could be observed. Estimates of spatial heterogeneity were produced both as the total number of habitat transitions over the area and also as an average of the number of video transects run. Initially, an average per video minute was also included in an attempt to remove differences based on transect length. However, the conversion from minutes to length was decided not to be justified as speed of the video was variable.

3.1.2 Functional traits

Initially it had been intended to identify to at least family level the visible organisms. However, while this was possible for the flora and some of the fauna (e.g., sponges, anemones), it wasn't for others, (e.g., ophuroids, holuthurians). For this reason, organisms were assigned to 24 taxonomic units (henceforth called biotic groups (Table 3-2)) that could easily be associated functional traits. Once these biotic groups were described, video footage was viewed a second time to assess the relative abundance of these groups along the transects: 0 = absent; 1 = present at one point along the transect; 2 = common, found multiple times or for extended minutes of footage; 3 = abundant, widespread and dominant. Relative abundance was assessed using this semi-quantitative (four-point) scale as the field of view of the video was generally unknown and inestimable.

Following this, the biotic group information was converted into functional traits data, as per Table 2-6, but including bioturbation characteristics. Small was defined as <15cm, medium as 15 -50cm and large as >50 cm.

This functional trait data was used to calculate the number of functional traits, richness, evenness and Shannon-wiener diversity observed along each transect, within each area and from all sampling in Port Pegasus. Differences in functional composition were assessed visually using non-metric multi-dimensional scaling of Bray-Curtis dissimilarities. Finally, the within-trait abundance and number of biotic groups representing the trait along each transect, within each area and overall were calculated.

Table 3-2: The biotic groups identified in video footage from Pegasus Bay.

Biotic groups	Description
Foliose (algae)	Red algae **Adamsiella*, *Gracilaria*, *Rhodymenia* and *Delesseria*) and the brown alga *Spatoglossum*
Filamentous (algae)	Red algae that are fluffier and highly branched compared with the above, including *Asparagopsis*, *Polysiphonia* and *Plocamium*
Turfing (algae)	*Corallina*
Ulva (algae)	Soft and leafy (green); seen drifting in the water column
Crustose Coralline (algal paint)	Encrusting (red) algae
Kelp (algae)	Large brown kelps (*Ecklonia*, *Carpophyllum*, *Landsburgia* and *Macrocystis)*
Caulerpa (algae)	A highly branched green algae
Ophiuroidea (fauna)	Species of Ophiuroidea (brittle stars), mostly *Ophiopsammus maculata*
Cerianthus (fauna)	A tube dwelling anemone.
Holothurian (fauna)	Unidentified species of Holothuroidea (sea cucumbers)
Asteroidea (fauna)	Unidentified species of Asteroidea (sea stars)
Scallops (fauna)	*Pecten novaezealandiae* of approximately legal collection size
Sponges-A (fauna)	Tall sponges that extent into the water column
Sponges-F (fauna)	Sponges with a lower profile, minimal extension into the water column
Ascidians tall (fauna)	New species of *Hypsistozoa* (M. Page), a colonial ascidian
Ascidians shorter (fauna)	This includes *Hypsistozoa fasmeriana* (Michaelsen, 1924) (M. Page)
Mounds (fauna)	Assumed to be thalassinidean shrimp or similar from the conspicuous burrowing "mounds". Organisms not observed.
Holes (fauna)	Similar to the above but with an absence of mounds. Organisms not observed.
Black corals (fauna)	Gorgonian fan type corals, ranging in size.
Nested mussel (fauna)	Variably sized clumps of mussels (*Modiolarca impacta*) attached to hard structures and forming cocoon-like byssal nets.
Horse mussels (fauna)	*Atrina zelandica*
Kina (fauna)	*Evechinus chloroticus*
Fish (fauna)	Multiple species present (either benthic or in the water column)

3.1.3 Habitat complexity

An index of habitat complexity was developed based on sedentary growth-forms, sizes and abundances of the various biotic groups in Pegasus Bay. Growth forms and sediment micro-topography that enhanced the vertical relief of the basal substrate were converted to a numeric rank (Table 3-3) depending on how intricately branched they were, their likely spatial extent (a mussel bed has a greater spatial extent than a single horse mussel) and their temporal life (e.g., burrows and mounds last less time than tubes and the lack of rigidity of foliose forms make them less likely to support other organisms than other branched forms). Complexity of form was then weighted by the size of the organism (small, medium, large, multiplied by 1, 2 and 3 respectively) to produce a score. Although theoretically possible, not all of the growth forms listed in Table 3-3 were observed in each of the three size categories. This score was then multiplied by the relative abundance on each transect and then averaged (and summed) across each area.

Table 3-3: Habitat complexity coding. Based on converting sedentary 'Growth Form' traits to a relative numeric representing complexity of form which was then weighted by 'Size'.

Growth-Form	Form complexity
Arborescent	5
Erect	4
Globose-cushion	4
Foliose	4
Bed-forming	4
Tubiculose	2
Crustose	1
Mounds	3
Burrows	2

3.2 Results

3.2.1 Differences between collection methods

A within-area pairwise comparison of sites that had been sampled by both diver and dropcam found no significant differences in the number of transitions observed. Similarly, the biotic group composition showed no marked difference between the diver and dropcam transects at either Inside Pearl or North Arm (Figure 3-1), although there were some differences at Disappointment and Pigeon. However, at Pigeon none of the transects were repeats at a site, and at Disappointment the substrate types differed between the diver transects (a mixture of soft and hard substrates) and dropcam transects (soft substrates).

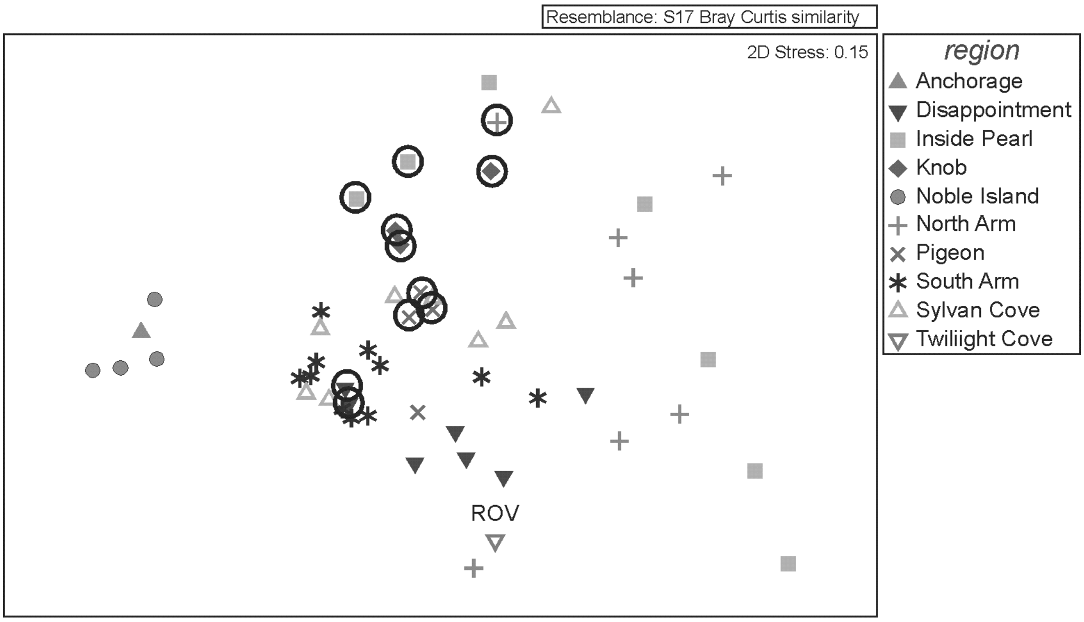

Figure 3-1: Non-metric multi-dimensional scaling ordination of relative abundance of biotic groupings observed on every transect. Points closest together are most similar, ringed symbols are those that were sampled by diver.

As there were no strong differences between method types, further results will not differentiate between methods.

3.2.2 Spatial heterogeneity

The total number of habitat transitions observed per area was related to the number of transects assessed per area. However, the Pearson's r correlation was poor (0.53) if the two areas sampled with just 1 transect (Twilight Cove, Anchorage) were excluded from the analysis. Total number of transitions ranged from 0 at Twilight Cove to 18 at North Arm (Table 3-4). Average number of transitions per transect varied from 0 (Twilight Cove) to 3.3 (Knob) and 3 (Anchorage).

Table 3-4: Number of habitat transitions observed for each area. Average number per transect, and total number of transitions observed in the area from all transects.

Area	Average	Total
Anchorage	3	3
Disappointment	1.3	9
Inside Pearl	1.9	13
Knob	3.3	10
Noble Island	2.75	11
North Arm	2.57	18
Pigeon	2.25	9
South Arm	1.17	14
Sylvan Cove	1.75	14
Twilight Cove	0	0

3.2.3 Habitat complexity

Conversely the total habitat complexity was strongly driven by number of transects taken in an area (Pearson's r = 0.94). Least habitat complexity was observed at Twilight Cove and North Arm, while highest average habitat complexity was observed at Anchorage with Noble Island next (Table 3-5).

Table 3-5: Habitat complexity observed for each area. Average complexity per transect, and total complexity measured over all transects in an area.

Area	Average	Total
Anchorage	160.0	160.0
Disappointment	113.4	794.0
Inside Pearl	101.0	707.0
Knob	106.7	320.0
Noble Island	149.0	596.0
North Arm	63.7	446.0
Pigeon	117.0	468.0
South Arm	117.5	1410.0
Sylvan Cove	89.8	718.0
Twilight Cove	66.0	66.0

3.2.4 Overall functional composition and diversity

No areas were distinctly different from all others in the ordination space (Figure 3-2), although Anchorage and Noble Island formed a cluster separate to the others.

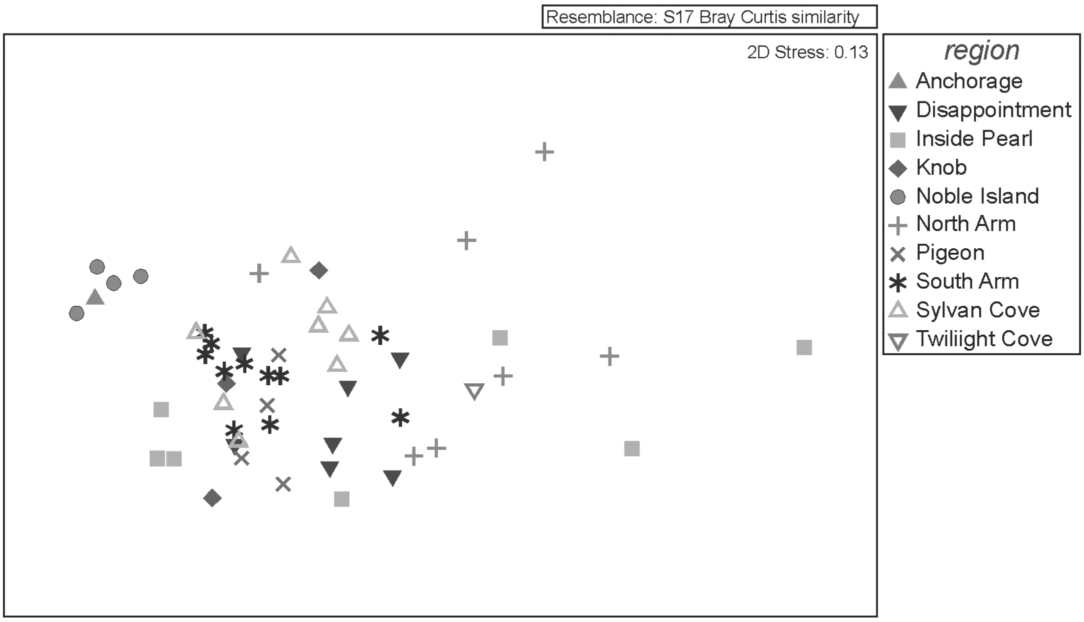

Figure 3-2: Non-metric multi-dimensional scaling ordination of relative abundance of functional traits observed on every transect. Points closest together are most similar.

The number of traits observed and the Shannon-Wiener diversity was lowest at Noble Island, Anchorage and Knob and highest at Twilight Cove. The evenness of the abundance of traits was high in all areas ($\geq$0.90).

Table 3-6: Functional trait diversity as number of traits, evenness and Shannon-Wiener diversity index (SW).

Area	Number	Evenness	SW
Anchorage	25.0	0.91	2.92
Disappointment	30.7	0.92	3.13
Inside Pearl	28.1	0.90	3.01
Knob	25.0	0.92	2.95
Noble Island	24.8	0.91	2.93
North Arm	30.1	0.92	3.14
Pigeon	27.8	0.92	3.05
South Arm	29.0	0.92	3.10
Sylvan Cove	28.4	0.92	3.08
Twilight Cove	34.0	0.93	3.27
Overall	28.6	0.92	3.07

Traits that were observed on every transect were:

- Growth forms- arborescent, crustose, erect, foliose, pisciform, vermiform.
- Position- attached, epifauna, pelagic.
- Flexibility- calcified, rigid, soft.
- Mobility- crawling, sedentary, swimming.
- Size- medium, small.

- Feeding mode- predator/scavengers, primary producers, suspension feeders.

3.2.5 Within-trait diversity

The number of traits for which an average of at least three biotic groups with that trait occurred on a transect was fairly consistent across areas (Table 3-7), ranging from 10 (Noble Island and Anchorage) – 15 traits (South Arm). Traits that were always represented by more than two biotic groups were attached, sedentary, small, medium, primary producers and suspension feeders and all traits from the flexibility category. Of the growth form traits, arborescent and erect forms were most likely to occur in more than two biotic groups per transect (Table 3-8).

Table 3-7: Summary of functional trait redundancy within areas. Number of traits per area represented by an average per transect of > 2 biotic groups (groups) or were common (relative abundance >2). Note that summing relative abundance (0, 1, 2, 3) of different biotic groups with the same trait results in trait relative abundance varying from 0 to 21.

Area	Groups	Common
Anchorage	10	20
Disappointment	13	22
Inside Pearl	14	19
Knob	12	20
Noble Island	10	22
North Arm	13	24
Pigeon	14	24
South Arm	15	24
Sylvan Cove	14	23
Twilight Cove	12	24

The relative abundance of traits was assessed by considering traits that were common (relative abundance > 2) at some stage along each transect. The number of these traits varied little across areas (Table 3-7), ranging from 19 (Inside Pearl) to 24 (North Arm, Pigeon, South Arm and Twilight Cove). Traits that were always dominant were erect, attached, epifauna, calcified, rigid, soft, crawling, sedentary, medium, small, predator, primary producers and suspension feeders (Table 3-8).

Both columns in Table 3-8 can be interpreted as representing the degree of functional redundancy and thus resilience as functional redundancy can be achieved through high abundance (i.e., large expanses of a trait) and through high trait richness (i.e., lots of distinct biotic group possessing the same functional traits). However, these two measures are likely to respond to stressors in different ways.

Table 3-8: Redundancy of individual traits across Port Pegasus. Number of traits per area represented by an average per transect of more than 2 biotic groups (groups) or were common (relative abundance >2).

Category	Trait	Groups	Common
Feeding	Deposit	0	0
	Grazer	0	2
	Omnivore	0	5
	Predator	8	10
	Primary	10	10
	Suspension	10	10
Flexibility	Calcified	10	10
	Rigid	10	10
	Soft	10	10
Growth form	Arborescent	6	9
	Bedform	0	0
	Bivalvia	0	1
	Burrows	0	2
	Crustose	0	9
	Erect	5	10
	Foliose	0	7
	Gastropods	0	0
	Globulose	0	2
	Mounds	0	2
	Pisciform	0	7
	Stellate	0	8
	Tubiculose	0	0
	Veriform	0	5
Mobility	Burrowing	0	4
	Crawling	8	10
	Sedentary	10	10
	Swimming	0	7
Position	Attached	10	10
	Epifauna	5	10
	Surface and top 2cm	5	8
	Infaunal	0	4
	Pelagic	0	7
Size	Large	0	3
	Medium	10	10
	Small	10	10

4 Conclusions and recommendations.

4.1 Port Pegasus

Port Pegasus appears to be a region supporting a diverse array of functional traits at relatively small scales (the transect level). None of the areas sampled are markedly different to each other in composition with the exception of Noble Island and Anchorage which differ from the rest in terms of biotic groups and functional composition. Twilight Cove is much less spatially heterogeneous in terms of habitat transitions, with low vertical complexity, but does exhibit the highest average number of traits and Shannon-Wiener diversity along a transect.

As expected, within-transect spatial heterogeneity did not correlate with vertical habitat complexity. Interestingly, areas with the lowest spatial heterogeneity and/or vertical habitat complexity, had high diversity of traits and vice versa, suggesting that diversity is maintained through a variety of processes. This finding adds to the complexity of determining a single index of functional integrity and suggests that functional integrity should, similar to ecological integrity, be considered as a multifaceted concept.

4.2 Functional integrity and BTA

Here we have demonstrated a successful method for converting video data to functional traits data, and the high information content of BTA.

BTA approaches fulfil most of the requirements of a good bio-monitoring tool. First, they are well rooted in ecological theory. Second, they enable *a priori* predictions of the ecological responses of communities to environmental conditions. Third, biological traits are indirectly related with ecological functions (e.g., feeding habits, reproductive frequency and body size are related to secondary production and respiration/metabolism). Fourth, multiple trait-based approaches have allowed the distinction among different types of human disturbances. Fifth, as the same traits are expressed in different species, the biological trait composition is spatially more stable than taxonomic composition across sites. Finally, indicators based on functional traits might also facilitate the evaluation of losses in terms of goods and services (by linking species composition to ecosystem services through functional traits (Beaumont et al. 2007; Townsend et al. 2011).

It is important to remember that the functional integrity approach used here is based on elements visible from a video survey of the seafloor. Thus, there are a number of functional components that are not well covered (e.g., nutrient fluxes, microphytobenthos and infaunal productivity, trophic links and potential for recovery from disturbance) although generally links between fluxes and large visible epifauna and flora have been demonstrated. Furthermore, the number of species representing functional traits is likely to be an underestimate as cryptic species such as infauna and sub-canopy species are either not sampled at all or are not well sampled.

4.3 Recommendations for future surveys

The video from Port Pegasus was of excellent quality, in part due to clear water in Port Pegasus, but also due to good camera gear and boat operation. However, the analysis we could do was limited by three factors that could be rectified in future surveys:

- Limited epifaunal taxonomic resolution, this could be increased in future by taking specimens for further taxonomic work.
- Lack of scaling lights or information on the length of the dropcam transects. Technology now exists to stream data from the boat's GPS and depth sounder and have it recorded onto the video footage. Although start and end times/positions were available for each transect sampled in this study, and notes were present, the lack of GPS coordinates and the difficulty of assessing the field of view (which varied with the height of the camera above the bottom) meant that using a fully quantitative approach was not possible. While the semi-quantitative biotic group based approach to quantifying the abundance of functional traits in Port Pegasus worked relatively well, lack of scale information probably contributed to variability and certainly affected the estimates of habitat heterogeneity (i.e., average number of habitat transitions per transect).
- Lack of knowledge as to why some areas had more transects taken than others, and whether the number of transects collected in an area was stratified by size of area or degree of habitat heterogeneity (either observed on the transect or that of the physical environment). Frequently, a first step in assessing ecosystem health is to evaluate the habitat heterogeneity of the region in question (Van Hoey et al. 2010). Particularly, the design of video surveys must take into account the habitat heterogeneity in the area and cover this heterogeneity so that it can be incorporated in the assessment of the ecosystem integrity through the BTA.

Collection of other measures of the ecosystem status should also be done, e.g., water turbidity, presence of non-native species, sediment deposition on the epibenthic fauna and flora. All these can be assessed from video images, but would require development of some standard methodologies. A standard methodology for determining habitat transitions is also necessary, but this can only be achieved when there is sufficient data available from a number of different areas and habitats.

4.4 Recommendations for future research

Creating a standard indicator of functional integrity requires a number of further steps.

- Development of standard methodologies as discussed in the previous paragraph.
- Determining to what extent the indicators used in this study are habitat independent. Some data which could be used to make this determination is available within DOC, NIWA, MPI and some regional councils. A project supported by NIWAs Coasts and Oceans Centre core funding is being developed that would include some assessment of this.
- Determining the sensitivity of the indicators to ecosystem state, natural temporal variability and stressors considered to be of particular importance requires surveys across a range of different ecosystem states—from largely un-impacted, like those in Port Pegasus, to highly stressed.
- Determining the natural temporal variability of the indicators. Currently, it is accepted that it takes multiple time series of metrics, and associated monitoring, to interpret the ecosystem status in a meaningful management context (Reza and Abdullaha 2010; Rice et al. 2012).

- These steps can be used to select the most useful of the measures, but Link et al. (2002) observed that a suite of metrics was required to accurately characterize the ecosystem status and, conversely, that focusing on only a few metrics (e.g., functional diversity, functional integrity or biological traits) can be misleading (see also Dauvin et al. 2007; Pinto et al. 2009; Ranasinghe et al. 2009). Combining the indicators that represent functional integrity to a single number or even a graphical representation of functional status requires development and validation of a method for doing this. Moving further from this to combine indicators of different components of ecological integrity will require a sustained national research direction focussed on ecological knowledge and possible management actions.

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