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Standardising the assessment of Functional Integrity in benthic ecosystems

Silvia de Juan [a,*], Judi Hewitt [a], Simon Thrush [a,b], Debbie Freeman [c]

[a] National Institute of Water and Atmospheric Research, P.O. Box 11-115, Hamilton, New Zealand
[b] Institute of Marine Sciences, Auckland University, New Zealand
[c] Department of Conservation, P.O. Box 10-420, Wellington 6143, New Zealand

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ABSTRACT

Ecological integrity is an overarching concept that integrates multiple properties of ecosystems, including structure, function and resilience to external change. We explore the links between ecological integrity and structural surrogates for ecosystem functioning to develop a cost-effective assessment of Functional Integrity for marine habitats based on biological traits, abundance and heterogeneity, focused on the visible components of the seafloor, i.e., epibenthic flora and fauna and seabed biogenic habitat features. The assessment was based on diversity and redundancy of functional traits of the identified benthic components, supplemented by estimates of spatial heterogeneity (habitat transitions) and vertical habitat complexity. This approach was developed using video data collected in different years with different sampling strategies in two locations: Kawau Bay in North Island of New Zealand, and Port Pegasus in Stewart Island, off South Island of New Zealand; this last location was a priori expected to be nearly—pristine. Despite variability in sampling techniques and environmental settings, the approach proved effective and evidenced higher measures of Functional Integrity in the Port Pegasus location. This study introduces a first step to measure ecological integrity by successfully converting video data to surrogates of Functional Integrity, in a way expected to be habitat independent.

1. Introduction

Globally, the main objective of many different frameworks and policies is to maintain or develop a good environmental status for marine ecosystems (Cochrane et al., 2010; SCBC, 2010), also referred as ecological integrity (EI). However, EI is not always either conceptually or operationally defined, as it is a high-level concept perhaps better understood by its absence rather than its presence. The term EI encompasses ecosystem structure, function and resilience (Özkundackzi et al., 2014) and it refers to the necessity to safeguard the self-organising capacity of ecosystems (Burkhard et al., 2011). But an important feature is that there is no "fossilisation" of the current or past state of the biota. Rather, there is recognition that following human modification and environmental change, the configuration of indigenous communities at a location may be quite different from that of the past (Lee et al., 2005). The elements of EI allow for natural successional change and trophic cascades and acknowledge that compositional shifts can occur in environments that have been modified by human activities.

Over the past decades, there has been an explosion of indices of "ecosystem health", "good environmental status" or "ecological integrity" (note the ambiguous definitions), partly due to the need for new tools for assessing the status of marine waters, which is required by regulations like the US Clean Water Act and EU Water and Marine strategy framework directives. But there is a general lack of scientific consensus and most indices are very location — and stress — specific (Borja et al., 2008; Hale and Heltshe, 2008; Van Hoey et al., 2010). Many ecosystem indicators developed for coastal areas are based on dividing macrobenthic species into the previously defined ecological groups in relation to the stressor, and examine the relative decrease of sensitive species confronted with a stressor, (e.g., Dauvin, 2007; Rosenberg et al., 2004; Simboura et al., 2005). But one of the problems with many of the indices currently available is that they focus on structure and deal less explicitly with function and thus may be poor surrogates for EI.

For ecological function at present there is a gap in terms of what we can easily and routinely measure and what these values mean (Birchenough et al., 2012). Directly measuring aspects of function is possible but in many cases it is more likely to be supported as a research question rather than a routine long-term monitoring tool, due to costs. The functional traits of the organisms and their interactions determine the functioning and stability of communities and ecosystems (Loreau et al., 2001) and thus potentially offer useful surrogate variables for ecosystem function. In this context, changes in functional components provide information about how communities respond to environmental stress and, in some contexts of their resilience (Lavorel and

* Corresponding author.
E-mail address: sdejuanmohan@gmail.com (S. de Juan).

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Garnier, 2002). The scientific challenge now is to scale up these studies to assess their relevance for monitoring diverse marine ecosystems as well as their dynamics and response to different sources of stress.

A thorough EI assessment implies broad spatial knowledge of habitats, biological communities and human uses, and this is currently lacking from most marine regions world-wide (Douvere, 2008). As ecosystems are interconnected, comprehensive monitoring and evaluating criteria are needed for measuring EI at regional levels (Reza and Abdullah, 2011). Recent advances in acoustic and visual survey techniques offer a great potential to support demands for frequent monitoring of seabed habitats at a range of spatial scales (Freitas et al., 2011; Lo Iacono et al., 2008). Remotely operated vehicles (ROVs) have proven to be very efficient for quantifying the distribution, structure, abundance and health status of benthic organisms in a variety of ecosystems (Teixidó et al., 2011; Thrush et al., 2001). Amongst several advantages, these techniques imply non-intrusive sampling (Teixidó et al., 2011) and record "real" images of the seafloor that are more likely to capture the behaviours of larger mobile species that are missed by other sampling methods. These techniques would allow direct estimations of epibenthic flora and fauna densities, as well as the identification of habitat structures (Hewitt et al., 2004), that can be linked to elements of EI.

Indicators of EI aim to summarise copious, complex, scientific information in a simple, condensed, and comprehensible way. One method of simplifying the "complexity of ecosystems", that integrates well with functional trait analysis, is to target selected ecosystem components. Our proposed method is based on the functional components and functional diversity of benthic communities (hereon we refer to Functional Integrity, FI). Targeting of benthic organisms has many advantages: they are relatively non-mobile and therefore useful for studying the local disturbance; some of these species are long-lived and would represent historical disturbance; many of the larger organisms can be monitored remotely over large scales (100 m to km) by video; and there is extensive literature on their distribution in specific environments and on changes related to various stresses. We propose to further reduce complexity by targeting "visible" benthic organisms (flora and fauna), recognising that while infauna and flora serve many important functions, in many systems visible components can represent the "invisible ones", i.e., inconspicuous fauna not registered by video (Reiss et al., 2010; Thrush et al., 2001). Based on the known relationships between ecosystem components (infauna–epibenthos and epibenthos–water column, though biotic interactions or nutrient fluxes, etc., e.g., Hewitt et al., 2006; Lohrer et al., 2005; Turner et al., 1997), we propose surveying visible epibenthos at large scales with remote sensing techniques and converting this information to aspects important for ecosystem functioning. Importantly, we propose using a biological traits methodology that should be independent of habitat type or regional species pool (Bremner et al., 2003; Hewitt et al., 2008). In the development of this proposed assessment framework, we introduce a case study of two subtidal coastal ecosystems in New Zealand that are expected to differ in terms of the integrity of benthic ecosystems. The objective of using the two locations was to determine whether the proposed approach could be used in two different environments where different video surveying methods were used. We evaluate the results against the prediction that functional diversity decreases with increasing environmental stress (Mouillot et al., 2006), and discuss its potential further application at broader scales.

1.1. Study approach

The strategy we propose is based on obtaining different metrics linked with the functional diversity of benthic systems that compose an essential subset of some elements of EI (Fig. 1) and that will guide the assessment of the broad concept of FI. The FI metrics proposed encompass the spatial heterogeneity, i.e., the variety and arrangement of biogenic habitats, defined by dominant engineering species (e.g., kelp forests, sponge gardens or shrimp burrow dominated habitats), in both 2 and 3 dimensions and then assess functional composition and diversity based on biological traits related to ecosystem functioning.

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 biological traits that relate to various aspects of how a community or ecosystem functions (e.g., dispersal, recovery, trophic dynamics, nutrient fluxes) (Bremner et al., 2003; de Juan et al., 2007; Villnäs et al., 2011) (Table 1). The Biological Traits Analysis (BTA) is well rooted in ecological theory (Statzner et al., 2001) and it fulfils most of the requirements of a good bio-monitoring tool: i) it enables priori predictions of the ecological responses of communities to environmental conditions (e.g., de Juan et al., 2009; Townsend and Hildrew, 1994); ii) biological traits are indirectly related with ecological functions, for example, reproductive frequency and body size are related to secondary production (Queirós et al., 2006; Statzner et al., 2001), such that analyses of biological traits are now frequently regarded as surrogates of functionality; iii) it allows the distinction amongst different types of human

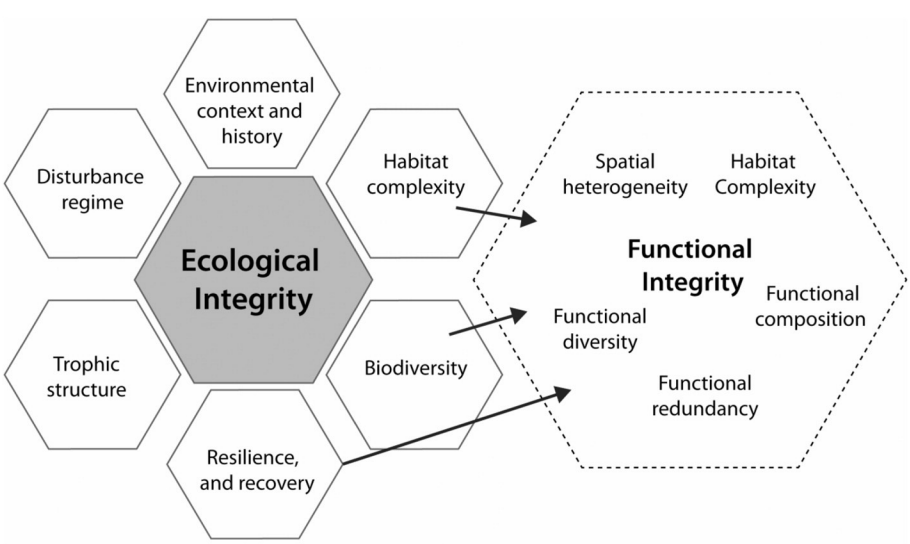

Fig. 1. General features of marine ecosystems to include in an ecological integrity assessment and overlap with Functional Integrity assessment: spatial heterogeneity and habitat complexity are components of broader elements of habitat structure and complexity; functional diversity and composition inform of the biological diversity of systems; and functional redundancy, together with diversity and composition, determine the resilience and recovery of ecosystems.

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Table 1
Trait categories most frequently used in assessing the functional components of communities related to ecological integrity. The traits are sorted by the degree of information generally known: Y = generally known, I = sometimes known, and 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

disturbances (e.g., pollution: Baird and Van den Brink, 2007; waste water treatment effects: Charvet et al., 1998; increased salinity: Piscart et al., 2006; fishing disturbance: de Juan et al., 2007; and iv) as the same traits are expressed in different species, the biological trait composition is spatially more stable than taxonomic composition across regional species pools (Hewitt et al., 2008).

In order to assess FI, the traits selected need to be related to a diverse range of functions and ecosystem components. Also it must be recognised that there are other aspects needed to provide insight into function. For example, landscape scale information on the connectivity of habitats, patch structure or the degree of bare space, provide broader scale parameterisation and provide insight into function (Hewitt et al., 2005; Thrush et al., 2006). We propose a hierarchy of monitoring variables ranging from the general to the specific, i.e., from habitat features to functional traits of the organisms to the number of taxa groups representing different functional traits.

1.1.1. Spatial habitat heterogeneity

Variations in density, the size and placement of dense patches of biological components (e.g., suspension feeding bivalve beds, rhodolith beds, and seagrasses) will all have significant effects on ecosystem function (Hewitt et al., 2004). Additionally, the site heterogeneity is important 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).

1.1.2. Habitat structural complexity

Habitat features are often generated by the resident biota, especially in soft sediments (Zajac, 2003) and they have been proposed that 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 will be linked to biodiversity and function (e.g., Bolam et al., 2002). Moreover, 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), and going beyond epifauna to include some assessment of the infauna. We suggest calculating a measure of habitat complexity based on size and complexity of growth form of sedentary species and sediment micro-topography.

1.1.3. Functional diversity

We propose a standard set of traits for benthic flora and fauna to be used across all habitat types, based on those used in a variety of studies (Table 2). The aim of using a standard set of traits is to objectively assess FI across locations, on the basis of functional diversity and redundancy, rather than on comparing the structure and composition of communities. This trait classification is subsequently translated, through multivariate analysis, into a quantitative functional composition characterisation of the community, including within and between location similarities, with a subsequent estimation of functional diversity and redundancy of functional traits.

2. Material and methods

2.1. Study areas and video sampling

Two study cases were compared: Kawau Bay in North Island of New Zealand and Port Pegasus, Stewart Island, south of the South Island of New Zealand (Fig. 2). Locations sampled ranged between 10 and 30 m deep and were mainly composed of a variety of soft-sediment biogenic habitats in Kawau Bay and a mix of soft-sediments and hard substrates in the Port Pegasus location. Port Pegasus, despite its name, is no longer a commercial port, and while historically was an area of high use (in the 1800s and early 1900s), is now considered one of the most pristine of New Zealand's coastal locations, due primarily to its remoteness and catchment forested in native vegetation (Department of Conservation, 2013). Kawau Bay, while not a particularly degraded environment, is located within the Hauraki Gulf near Auckland, the largest New Zealand city (population > 1 million). Of the two locations, we expect higher Functional Integrity in Port Pegasus than in Kawau Bay.

Five locations were surveyed in Kawau Bay: Bigbay, Iris Shoal, Mayne, Motuora Island and Pembles Island. These locations were surveyed with a towed video in summer of 1999; three 1 km towed-video transects were done per location using two high-resolution colour CCD video cameras with independent light sources and scaling lasers (further details of the study can be found in (Hewitt et al., 2004). Eight locations were surveyed in Port Pegasus in winter of 2012: Disappointment, Inside Pearl, Knob, Noble Island, North Arm, Pigeon House, South Arm and Sylvan Cove. Two video survey techniques were used: 1) a diver filming with a video along a 100 m transect, and 2) a boat drifted drop-camera of varying length. No significant differences in community composition were detected between samples from the same location obtained with different methods; therefore, data from these two sampling techniques was merged for the Port Pegasus data. In order to standardise all data, the 1 km video transects in Kawau Bay

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

Traits	Megafauna	Flora
Position/living habitat	Epibenthic, attached, infauna (endobenthic)	Epibenthic, attached
Growth form	Crustose/encrusting, globose/cushion, arborescent, tubiculous, bed forming, erect, vermiform, turbinate, stellate, bivalvia, articulate, pisciform, burrow-dweller	Foliose, laminar, arborescent
Flexibility	Soft, rigid, calcified	Soft, 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

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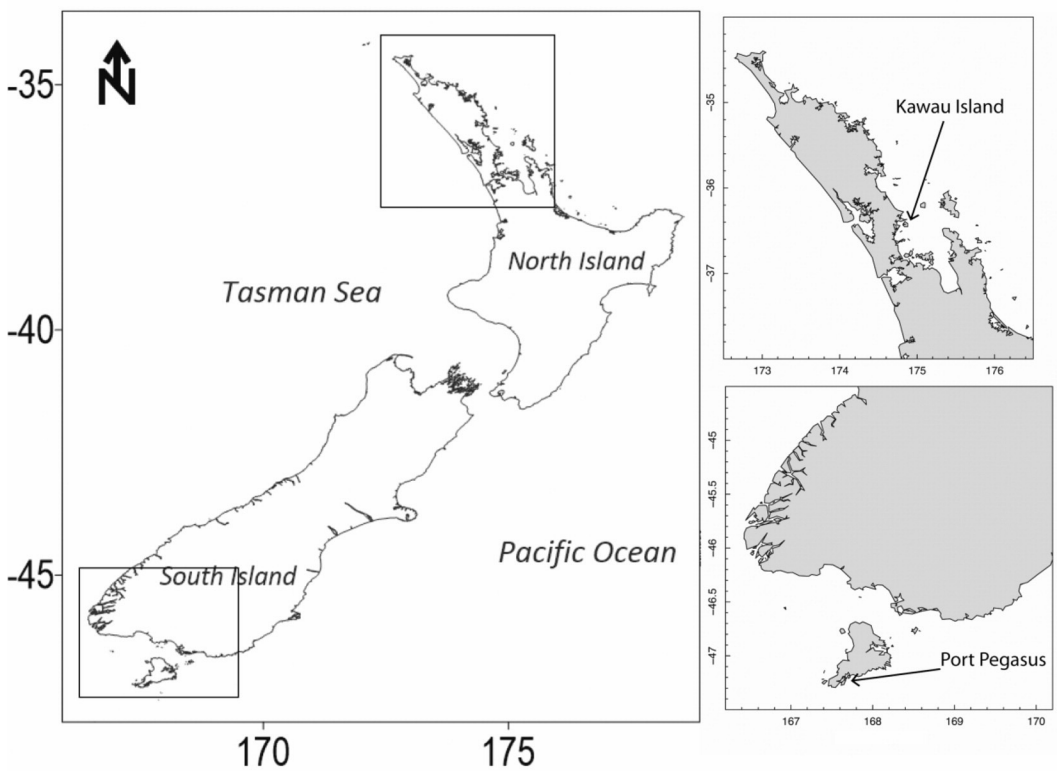

Fig. 2. Map of New Zealand with the location of the two study areas: Kawau Bay in the North Island and Port Pegasus on Stewart Island, south of New Zealand South Island.

were randomly split into 100 m samples, obtaining 5–6 samples per location.

2.2. Spatial heterogeneity index

Habitat types were defined based on the dominant biological component (e.g., tube mat, mixed sponge-bivalve bed, kelp canopy, bare sand, bioturbated mud) that was assigned after viewing the full video footage. Species differences within these broad categories were only used to define different habitats if they resulted in obviously visual differences in density, height protruding above seafloor, or form. For example, scallop and horse mussels are both bivalves yet they can easily be identified as different habitat types based on complexity. Similarly, differences in growth form between 2 canopy forming kelps would result in them being treated as different habitats along a transect. This would also be the case with the same species of kelp if the growth height and density changed markedly. The spatial heterogeneity was defined as the number of habitat transitions in each video sample and for Kawau Bay it was standardised as the average of transitions per 100 m in each location.

2.3. Habitat complexity index

A vertical habitat complexity index was developed based on growth forms and sediment micro-topography that enhanced the vertical relief of the basal substrate (form complexity see Table 3). These definitions were originally developed for the soft-sediment habitats from Kawau Bay; the one extra growth form that had to be added for the hard substrate was "foliose". This was added because it was felt that it was important to differentiate between the more flexible foliose algae and the more rigid branching algal and sponge species. These were ranked depending on how intricately they were branched, their likely spatial extent (2-dimensional extent of a single unit) and their rigidity (Table 3), based on expert's judgement. This rank was then weighted by the vertical size of the habitat-former (small, medium, large, multiplied by 1, 2 and 3 respectively). Small was defined as <15 cm, medium as 15–50 cm and large as >50 cm. The obtained score was then multiplied by the relative occurrence of the corresponding components observed across each sample. Sample results were then averaged for each location. Analysis of the sensitivity of results to the initial ranks was conducted, namely around mounds, burrows and crustose ranks, however, any differences appeared removed once the size weighting

Table 3
Scores assigned to the growth form and micro-topography to assess the form complexity. Branching, spatial extent and rigidity were added to create form complexity. Vertical habitat complexity was then determined by weighting form complexity scores by vertical size and occurrence.

Growth form and sediment microtopographic features	Branching	Spatial extent	Rigidity	Form complexity
Arborescent	3	1	1	5
Foliose	3	1		4
Erect colonial or bed-forming	1	2	1	4
Erect other	1	1	1	3
Single tubes	1		1	2
Crustose	1			1
Mounds	1	2		3
Burrows	1	1		2

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was applied. In the Port Pegasus data these analyses were conducted on full transects, for the Kawau Bay data, they were conducted on detailed analyses of randomly selected 100 m sections. Relative occurrence was defined as 0 (not observed), 1 (observed occasionally), 2 (common, found multiple times or for extended minutes of footage) and 3 (abundant, widespread and dominant).

2.4. Functional traits

Images were inspected to assess relative occurrence of visual benthic organisms and seabed structures, occasionally to a species level but more generally to a taxonomic/biotic group (e.g., branching sponges, large arborescent kelp, turfing algae, crustose coralline, large burrows). The trait categories listed in Table 2 were assigned to all groups identified in the samples. Fuzzy coding was used to allow the species to vary in the degree in which it exhibited affinity to a specific trait within a category (Chevenet et al., 1994). This fuzzy probability matrix was multiplied by relative occurrence to obtain a trait abundance matrix for subsequent analysis. The trait occurrence matrix was analysed with multivariate statistics to test for variability in the functional trait composition across sites within a location using: non-metric multidimensional scaling ordination based on Bray–Curtis similarity, Analysis of Similarities and the Similarity Percentages (MDS, ANOSIM and SIMPER procedures with PRIMER, Clarke and Warwick, 1994). The occurrence matrix was square-root transformed prior to multivariate analysis.

The number of functional traits, evenness and Shannon–Wiener diversity of traits observed in each sample were also calculated. Functional redundancy was assessed by high trait richness (i.e., abundant traits: traits with at least three distinct biotic group possessing this same functional trait) and high frequency of occurrence (i.e., common traits: traits with relative occurrence > 2 at some stage along each transect).

Overall differences between areas and between locations within each area for each of these metrics were assessed with non-parametric Kruskal–Wallis tests. No multiple comparisons were performed within areas due to the un-balanced design and the relatively low n (d.f. = 7 for Port Pegasus and d.f. = 4 for Kawau Bay). Therefore, comparisons between locations within an area were qualitatively assessed after significant Kruskal–Wallis test.

3. Results

3.1. Spatial heterogeneity

In Kawau Bay, the average spatial heterogeneity (number of transitions per sample) ranged from 0.9 in Motuora Island to 0.4 in Mayne, but no significant differences were detected. In Port Pegasus, the average number of habitat transitions per sample was lowest in South Arm, 1.2, and highest in Knob, 3.3 ($p < 0.01$). The overall spatial heterogeneity was highest in Port Pegasus area, $p < 0.001$ (Table 4).

3.2. Habitat complexity

In Kawau Bay, the habitat complexity was highest in Iris Shoal, which was dominated by bivalve beds (*Atrina zelandica*) and sponges, and lowest in Motuora Island (a mix of bare sand/mud and *Atrina*) and Mayne (dominated by scallop beds) ($p < 0.05$). In Port Pegasus, the habitat complexity was highest in Noble Island, which was predominantly rocky reef substrate, and lowest in North Arm location ($p < 0.01$), a mix of soft-sediment and rocky reef substrates. The overall habitat complexity was higher in Port Pegasus ($p < 0.001$), although North Arm and Sylvan Cove values were similar to those from Kawau Bay (Table 4).

Table 4
Metrics obtained from the video samples in Kawau Bay (5 locations) and Port Pegasus (8 locations): spatial heterogeneity (SH) as the average of transitions standardised to 100 m transect; habitat complexity (HC) as the size weighted average occurrence of complexity scores (Table 3); "overall" is the average of locations in each area ± standard deviation.

Kawau Bay	SH	HC
Bigbay	0.8	77
Iris Shoal	0.6	90
Mayne	0.4	62
Motuora	0.9	57
Pembles	0.73	71
Overall	0.69 ± 0.19	71.4 ± 13
Port Pegasus	**SH**	**HC**
Disappointment	1.3	113
Inside Pearl	1.9	101
Knob	3.3	107
Noble Island	2.8	149
North Arm	2.6	63.7
Pigeon House	2.3	117
South Arm	1.2	118
Sylvan cove	1.8	90
Overall	2.4 ± 0.78	106.4 ± 30.5

3.3. Functional traits

In Kawau Bay, the locations differed in relative abundance of functional traits (ANOSIM test: 0.54, $p < 0.01$). All pair-wise comparisons of locations were significantly dissimilar, with the exception of Bigbay, Mayne and Motuora Island, where no significant differences were detected between these 3 locations (Fig. 3a). In Port Pegasus, no location was distinctly different from any other in the ordination space (ANOSIM test: 0.3, $p < 0.01$; Fig. 3b) and Inside Pearl was not significantly dissimilar to any other site, while Disappointment, Knob, Pigeon and Sylvan cove were dissimilar to only half of the sites. The within-location similarity was high at all locations in both Kawau Bay and Port Pegasus (Bray–Curtis similarity >80%).

Number of traits and Shannon index in Kawau Bay were highest in Iris Shoal and lowest in Pembles Island and Motuora Island; the other 2 locations had similar values (Table 5); significant differences were only detected for the Shannon index, $p = 0.03$. In Port Pegasus, the number of traits and the Shannon index were higher in Disappointment and North Arm, and lower in Knob and Noble Island (number of traits: $p = 0.04$; Shannon index: $p = 0.01$). Evenness was similar across locations within each area, but was higher on average in Kawau Bay ($p < 0.001$). The average number of traits across locations was also higher in Kawau Bay ($p > 0.001$), while no significant differences were detected for the Shannon index between locations (Table 5).

In Port Pegasus, the number of abundant traits (>2 biotic groups with that trait) that occurred on a transect was fairly consistent across locations (Table 5), ranging from 10 (Noble Island) to 15 traits (South Arm), and averaging 13 across locations. In Kawau Bay the number of abundant traits was lower, averaging 11 across locations, ranging from 9 in Mayne to 13 in Iris Shoal (Table 5). In Port Pegasus, the number of common traits (traits with relative occurrence > 2) varied little across locations, ranging from 19 (Inside Pearl) to 24 (North Arm, Pigeon House and South Arm). In Kawau Bay the number of common traits had an average across locations of 18.9, ranging between 23 in Irish Shoal to 16 in Mayne (Table 5). Neither of these two measures of functional redundancy was significantly different between sites within a location, but weak significant differences were detected between locations ($p = 0.031$ for common traits and 0.026 for abundant traits).

4. Discussion

In this study we propose an approach to assess the Functional Integrity of marine benthic ecosystems based on the estimation of the

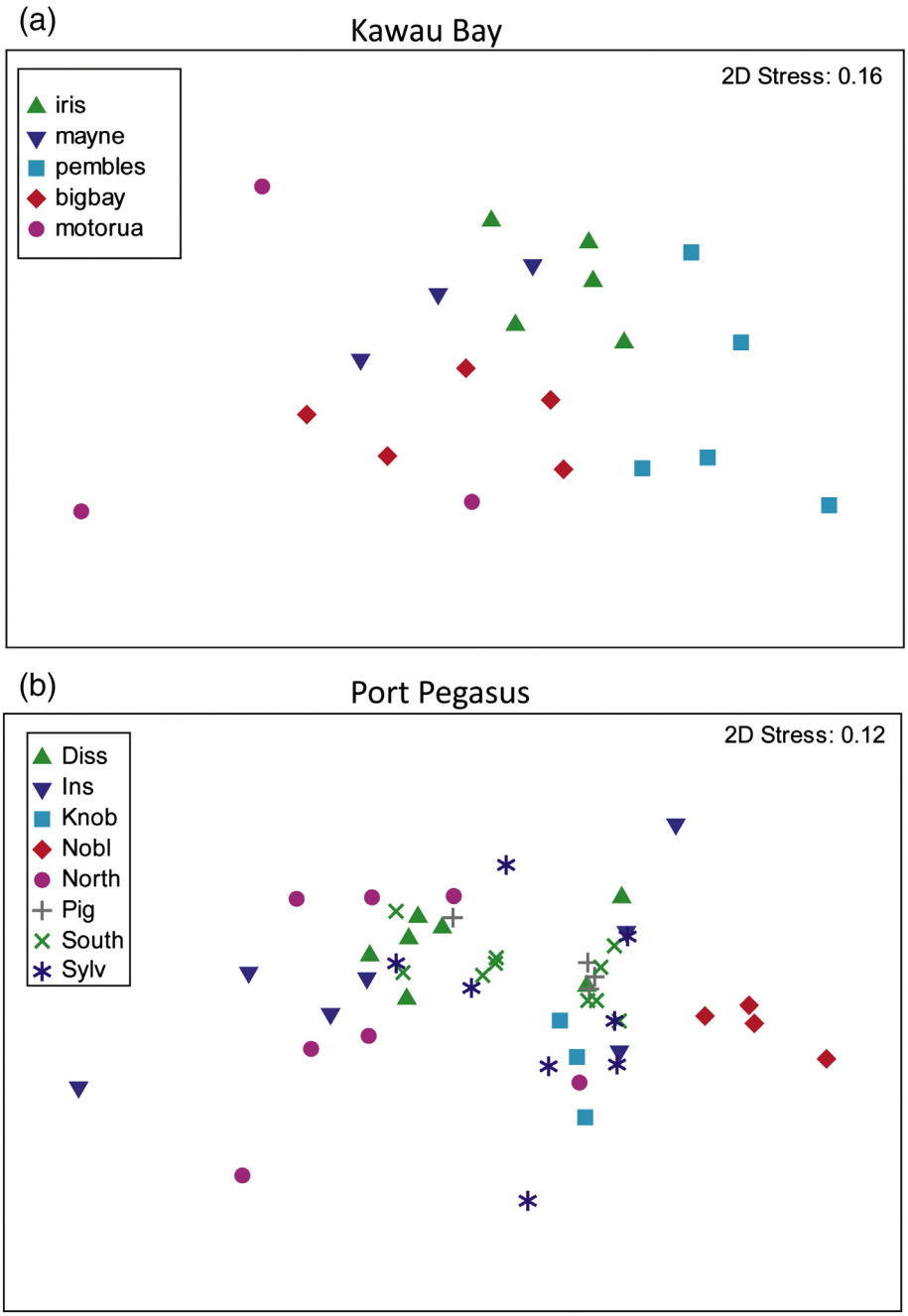

Fig. 3. Non-metric multi-dimensional scaling ordination of the relative abundance of functional traits in the two study sites: Kawau bay (a) and Port Pegasus (b). Samples from the 5 locations in Kawau Bay and 8 locations in Port Pegasus are represented by symbols.

relative occurrence of functional components of the seafloor, identified by video surveys that comprises a range of metrics rather than a single one (Fig. 4). The approach starts with a general characterisation of the seafloor structure and increases in complexity to estimate indices of functional diversity and redundancy; the approach aims to be flexible as it could be re-adjusted as larger sets of data become available. However, we see our Functional Integrity metrics as only one step. Functional Integrity indices need to be considered within the environmental setting (specific stressors and historical context), and their temporal dynamics need to be established. Knowledge on the existing sources of stress is important, as, for example, habitat transitions could be linked to stress gradients conditioning observed patterns. If any specific stressors were considered to be of importance, we would expect that trait categories assumed to respond to those stressors would be explicitly included and assessed separately.

Our approach was applied in two areas that varied in the degree of soft versus hard substrates and that were surveyed in different years with different video techniques. In accordance with predictions of lower functional diversity and seabed structural complexity with stress (Thrush and Dayton, 2002), the average FI metrics in Port Pegasus, where anthropogenic stress was expected to be lowest, were consistently higher than those in Kawau Bay. Kawau Bay was characterised by bare soft-sediments alternating with biogenic habitats and the habitat complexity and spatial heterogeneity were considerably lower than in Port Pegasus, characterised by a mix of rocky reef and soft sediment. Port Pegasus appeared to be a region supporting a diverse array of functional traits at relatively small scales, with a large number of traits with both high occurrence and large number of organisms with the same trait (22 and 13 traits respectively out of an overall of 38 traits) that would contribute to resilience facing external stress. While encouraged

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Table 5
Functional diversity in Kawau Bay (5 locations) and Port Pegasus (8 locations): number of traits (S), Shannon index (H) and evenness (J); "abundant traits" is the number of traits represented by >2 taxonomic/biotic groups in each transect; "common traits" is the number of traits with relative occurrence >2 along a transect; "overall" is the average of location in each area ± standard deviation.

Kawau Bay	S	H	J	Common traits	Abundant traits
Bigbay	32.3	3.4	0.98	17	11
Iris Shoal	34.2	3.5	0.99	23	13
Mayne	32	3.4	0.98	16	9
Motuora	30	3.3	0.98	19	10
Pembles	29.6	3.3	0.98	19	10
Overall	31.7 ± 3.6	3.4 ± 0.1	0.98 ± 0.01	18.9 ± 5.6	11 ± 2.5
Port Pegasus	**S**	**H**	**J**	**Common traits**	**Abundant traits**
Disappointment	30.7	3.13	0.92	22	13
Inside Pearl	28.1	3.01	0.9	19	14
Knob	25	2.95	0.92	20	12
Noble Island	24.8	2.93	0.91	22	10
North Arm	30.1	3.14	0.92	24	13
Pigeon House	27.8	3.05	0.92	24	14
South Arm	29	3.10	0.92	24	15
Sylvan cove	28.4	3.08	0.93	23	14
Overall	27.7 ± 2.8	3.03 ± 0.01	0.91 ± 0.01	22.3 ± 1.9	13 ± 1.6

by these results we see an important next step that is to use the methodology in a series of locations that represent a stronger gradient in anthropogenic disturbance and thus potentially in Functional Integrity.

In New Zealand, the concept of ecological integrity is being used by a government agency, the Department of Conservation, to monitor and report on the status of conservation lands, utilising a suite of indicators

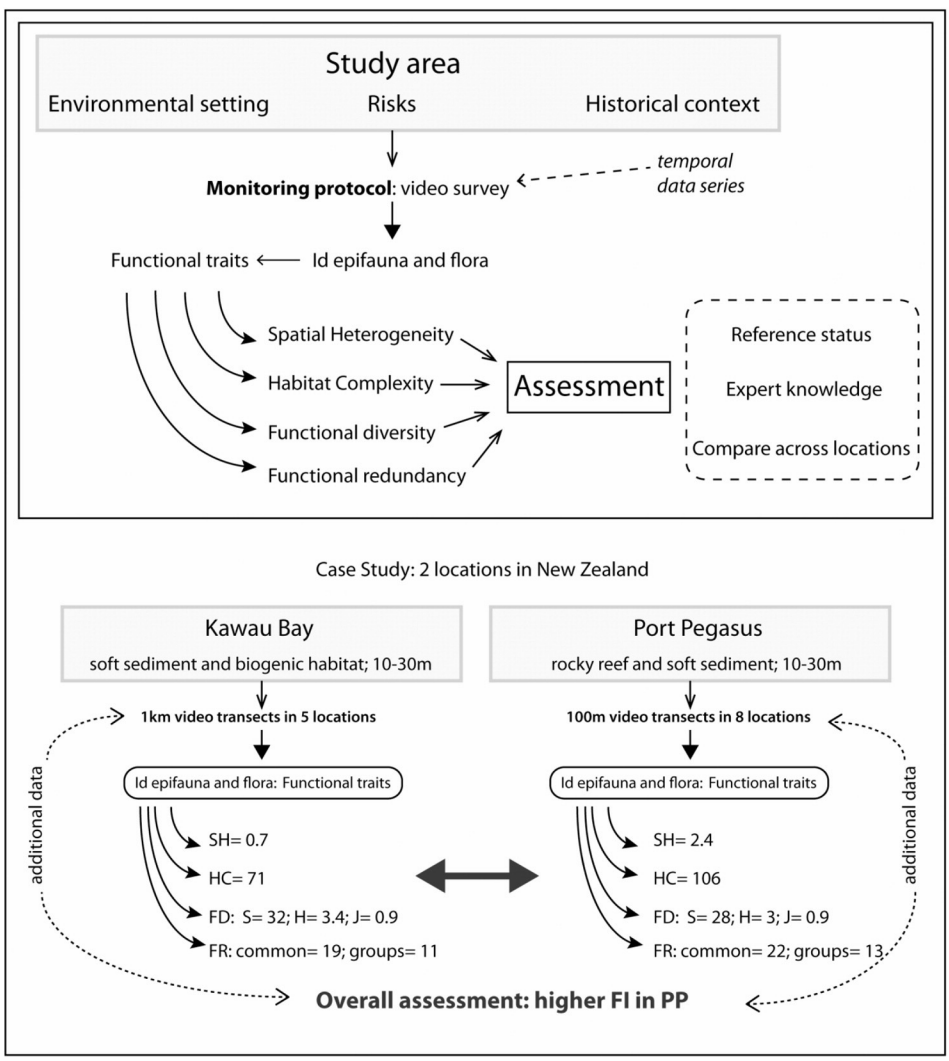

Fig. 4. Proposed protocol for the assessment of Functional Integrity based on video surveys and example of the assessment in Kawau Bay and Port Pegasus, based on the overall comparison of the FI metrics, and on the assumption that the assessment would be improved as more data are obtained, aiming to match the protocol proposed.

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of ecological integrity, including the representation of functional groups, species demography and distribution and abundance of invasive species (Bellingham et al., 2013). A systematic monitoring system is currently being employed, allowing a broad-scale assessment of the status of New Zealand's native terrestrial species and ecosystems (Department of Conservation, 2013). This system was considered to allow not only national and regional reporting of status and trend in EI, but also contribute to the evaluation of the effectiveness of conservation management and policy, provide an early warning system and inform the prioritisation for resource allocation (Bellingham et al., 2013). A similar monitoring and reporting framework for the marine environment is currently in development and will be an invaluable opportunity for extending and testing our approach.

With the implementation of our methodology to the Kawau Bay and Port Pegasus data, we could identify two major problems, namely sampling extent and defining habitat transitions, which we would further explore when conducting future sampling. Preferably, video transects should cover similar lengths at a standardised width, but this is rarely practical due to variations in waves, currents and visibility. Recording geo-location, so that estimations of surveyed areas could be made, would at least allow relative differences related to sampling extent to be examined and possibly overcome. Another area requiring standardisation is the definition of habitat transitions. While we found it easier to define a transition had occurred rather than to define what a habitat was, or the exact GIS location of the transition, it was essentially a subjective process. Having developed the method, it would now be appropriate to define the relative change, in terms of density, form and size, required to be considered as a transition. Our score ranking of form complexity could also be improved with more sampling and analysis. It appears reasonably robust with respect to habitats as we only had to incorporate one new form (namely foliose) in moving from the soft-sediment habitats, in which the scoring was developed, to the soft-hard substrate mix of habitats in Port Pegasus. We did test the sensitivity of the estimate of habitat complexity to some aspects, namely assigning mounds as higher than burrows and assigning crustose as the lowest value, and found that these minor changes were smoothed over by the size weighting. However, incorporating opinions of others working in different areas and trialling across a range of systems will undoubtedly result in improvements.

Our data suggests that locations may be differentially ranked using different measures. Locations 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, similar to the broader concept of ecological integrity, it should be considered as a multifaceted concept. This is not surprising, as we would expect that integrity, and especially resilience, would be maintained by multiple processes; however, it does raise the issue of how these measures should be integrated. Due to the limitations imposed by the differences in survey protocol, and only having 2 areas sampled expected to exhibit different levels of ecosystem degradation, we merely do simple comparisons of metrics. However, an array of sophisticated statistical techniques is currently available; this could range from simple Decision Trees to Structural Equation Models that link changes in indicator response to environmental variability (Thrush et al., 2012). Simultaneously evaluating multiple-metrics, all related to some degree with the FI, may aid the assessment of the ecological status of a location, where not all metrics point to the same status. Also, the approach might be improved by differently scoring the metrics, on the basis of their tighter link to the FI of a location. But in order to use these techniques, a number of steps are required, the first of which we suggest is the adoption of a standard protocol as discussed above. Other steps include sampling both in more degraded areas and in a variety of habitat types, and the collection of environmental and historical context data (Fig. 4).

The approach we propose 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 and trophic links), although generally links between fluxes and large visible epifauna and flora and links between the diversity of infauna and epibenthos have been demonstrated (e.g., Lohrer et al., 2005; Thrush et al., 2014). While it is accepted that large epifauna are key components of hard substrates, often driving the distribution of other species, epifauna and flora are often also key species in soft substrate systems, influencing the infauna by providing habitats, affecting productivity and modifying environmental factors. In areas without large epibenthos, the presence of key functional infauna can often be seen by modifications to the sediment microtopography (burrows, holes, mounds) (Hewitt et al., 2004). However, this needs to be robustly tested in any habitat type where this approach could be used to evaluate FI. It maybe that a hierarchical sampling method that nests collection of infaunal data within the broader scale video data will not only build robustness but also form a link to past surveys focused on grabs and cores. Moreover, the metrics we proposed are cost-efficient but in order to escalate to ecological integrity, 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 (Fig. 1) (Özkundackzi et al., 2014). Many of these could also be assessed from video images, but would require development of some standard methodologies. Such standardisation of methodology and the collation of measures of the various components of ecological integrity, being the FI metrics an essential part of this process, would allow a comprehensive assessment of the status and trends in marine species and habitats, including for the purposes of monitoring and reporting on the effectiveness of conservation management and prioritising resources (Bellingham et al., 2013).

5. Conclusions

Ecosystem integrity assessments need to capture the structure, function and resilience of natural systems, emphasising the need to incorporate function and ecosystem performance information into the development of indicators. Here we have demonstrated a successful method for converting video data to functional traits data, and other important aspects of seafloor functioning. What we propose is a framework that allows for, and encourages, detailed measurements of the functional components of the seabed, as an essential constituent of EI, and supports continuous knowledge generation considered appropriate to management. We also stress the importance of building on previous monitoring and maintenance of time series in developing an appropriate framework for an ecological integrity monitoring strategy. Combining the indicators that represent Functional Integrity in a meaningful way 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 research at regional scales focussed on ecological knowledge and possible management actions. Currently, monitoring of the ecosystem status at regional scales based on video surveys of the seafloor is probably the most cost-efficient approach; as demonstrated in our case study, functional components of the seafloor can be obtained from video images and used to assess the Functional Integrity of the area. The way forward should focus on combining this approach with other measures obtained from video images and that link with the integrity of the ecosystems.

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