9.7 Macroinvertebrate community composition

Author, affiliation: Orlando Lam-Gordillo (NIWA), Drew Lohrer (NIWA)

Citation for this chapter: Lam-Gordillo, O., Lohrer, D. (2024). Macroinvertebrate community composition. *In:* Lohrer, D., et al. *Information Stocktakes of Fifty-Five Environmental Attributes across Air, Soil, Terrestrial, Freshwater, Estuaries and Coastal Waters Domains*. Prepared by NIWA, Manaaki Whenua Landare Research, Cawthron Institute, and Environet Limited for the Ministry for the Environment. NIWA report no. 2024216HN (project MFE24203, June 2024). [https://environment.govt.nz/publications/information-stocktakes-of-fifty-five-environmental-attributes]

State of knowledge of the "Macroinvertebrate community composition" attribute: Good / established but incomplete – general agreement, but limited data/studies.

In general, the state of knowledge for the "Macroinvertebrate community composition" attribute is 'good/established but incomplete'. Macroinvertebrate communities are well studied internationally and nationally. There is good evidence and track record of their ecology, distribution, and use for depicting natural and anthropogenic disturbances (bioindicators). There are several New Zealand studies describing relationships between macroinvertebrate community composition and stressors such as nutrients and mud content in sediments [1-4]. However, assessments identifying multiple stressors affecting these communities are limited, information on tipping points is scarce, and the further consequences to ecosystem functioning and provision of ecological services is lacking. Despite the existence of relatively standardised national protocols for monitoring, collection, and identification of macroinvertebrate communities [5-8], there are regions in New Zealand where sampling and knowledge of macroinvertebrate communities is limited. This may impede New Zealand-wide comparisons and the implementation of national guidelines.

Part A—Attribute and method

A1. How does the attribute relate to ecological integrity or human health?

International and national research demonstrates that macroinvertebrate community composition is strongly related to ecological integrity (and somewhat indirectly to human health if/where macroinvertebrate community composition indicates toxic contamination). Macroinvertebrate communities are a key component of marine and estuarine ecosystems. These organisms are major providers of ecosystem functions and services in marine habitats. For example, they transfer energy and matter from lower to higher trophic levels as food sources for fish and birds, and modify soft-sediment habitats through biological processes such as ingestion, digestion, excretion, and bioturbation, which facilitates microbial recycling of nutrients, detoxification of pollutants, and organic matter remineralization [3, 9-12]. Macroinvertebrate communities are widely used as bioindicators of natural and anthropogenic disturbances and often used in estuarine monitoring programmes to assist assessments of ecosystem health due to their sensitivity to environmental change [8, 13-16]. Macroinvertebrate communities are found throughout estuarine, coastal, and open ocean benthic ecosystems across New Zealand, and are vital to the functioning of these ecosystems. For this reason, univariate and multivariate metrics developed from macroinvertebrate community data are potentially highly useful as measurable and comprehensible estuarine/coastal environmental attributes.

A2. What is the evidence of impact on (a) ecological integrity or (b) human health? What is the spatial extent and magnitude of degradation?

Healthy estuarine and coastal systems with high ecological integrity tend to have diverse macrobenthic communities characterised by high numbers of individuals and taxa. There are many different types of "healthy" macrobenthic communities. In contrast, highly stressed and disturbed estuarine and coastal systems support fewer macrofaunal individuals and taxa, and the range of possible community types tends to be smaller. A large body of international peer-reviewed literature suggests that relationships between macrofauna and stress/disturbance are moderately predictable, but that the relationships are not always linear or monotonic. For example, macrofaunal abundance can be high in "unhealthy" organically enriched sediments (e.g., when dominated by a few highly opportunistic taxa), whilst areas of "intermediate" disturbance (rather than lowest disturbance) may support the highest numbers of taxa [17]. It is also important to recognise that healthy estuarine and coastal benthic ecosystems are mosaics of patches that are subjected to a range of natural and anthropogenic disturbances that are occurring on varying spatial and temporal scales, so environmental context is crucial for interpreting macrofaunal community metrics.

In simplest terms, when estuarine/coastal macroinvertebrate communities are highly diverse (high numbers of taxa and high evenness in abundance across taxa), they are thought to be more resistant and resilient to stress and disturbance. This is because biodiversity underpins functional redundancy (having multiple species as back-ups if one or a few species are lost) and response diversity (having species with different sensitivities to the same set of stressors, which enables communities to maintain functionality). These concepts are central to ecological integrity.

Internationally and nationally, there is strong evidence of the impact of degraded macroinvertebrate communities on the ecological integrity of coastal ecosystems and indirect impacts on human health (internationally [14, 16, 18-21], nationally [2, 3, 8, 15, 22-32]). In New Zealand, past and ongoing anthropogenic pressures such as coastal development, conversion of natural habitats to land for agriculture and forestry, excessive fishing and resource extraction, industrialisation, and increasing nutrient and sediment inputs, in combination with overarching global stressors (e.g., increases in sea water temperature and sea level rise, increases in the frequency and intensity of heatwaves, changes in dissolved oxygen concentrations, and decreases in ocean pH) are deteriorating the health of macroinvertebrate communities [3, 9, 22, 23, 25, 27, 28, 30, 32-34]. These human-induced pressures can alter the composition and structure of macroinvertebrate communities, potentially limiting the provision of key ecosystem services in New Zealand coastal ecosystems. The anthropogenic impacts are evident at national scale, decreases in macroinvertebrate communities and reduction on ecosystem functions and services due to severe increases in sediment and nutrient inputs along coastal ecosystems has been extensively reported [2-4, 26, 27, 30, 32, 33, 35-39]. Some of the most rapid and spatially widespread shifts in macroinvertebrate community composition have been observed in Southland estuaries, such as New River and Jacob's River estuaries, where dairy intensification and nutrient/sediment loadings have impacted macrobenthic invertebrate communities across scales of tens to hundreds of hectares [40, 41].

A3. What has been the pace and trajectory of change in this attribute, and what do we expect in the future 10 - 30 years under the status quo? Are impacts reversible or irreversible (within a generation)?

It is highly likely that macroinvertebrate communities in New Zealand have been degraded since monitoring for macrofauna began ~50 years ago, and likely longer [42]. The causes of degradation are thought to stem primarily from elevated rates of sediment and nutrients discharged to coastal receiving environments. The influence of sediments and nutrients has been confirmed primarily from space-for-time substitution studies supplemented by experiments, rather than by monitoring (due to the poor spatial coverage and low frequency of macroinvertebrate community sampling in most New Zealand estuaries). Urban stormwater contaminants and stressors associated with ports and marinas (dredging, pollutants, anti-foulants) have also impacted macroinvertebrate communities in places.

New Zealand has naturally high rates of sediment loading due to high rainfall and steep catchments, although terrigenous sediment loading to estuaries is reportedly 10 to 100 times higher today than it was prior to European colonisation ~150 years ago [43, 44]. Degradation of macroinvertebrate communities in Southland estuaries appears to have accelerated during the last 20-25 years in association with land use change (e.g., dairy intensification).

Macrofauna themselves have the potential to recover quickly, with many species highly fecund and highly dispersive. However, the habitats that support macrofauna (for example, highly muddy and infilled arms of estuaries) may take longer than 10-30 years to recover due to the legacies of past stressors. If the loading of sediments and nutrients can be better controlled over the next 30 years, rising sea levels and increased flushing of estuaries with cleaner coastal seawater may facilitate the recovery of estuarine macrofaunal communities. However, other aspects of climate change (e.g., heat waves; increased storm intensity and sediment loading) may increasingly impact macroinvertebrate communities over the next thirty years.

A4-(i) What monitoring is currently done and how is it reported? (e.g., is there a standard, and how consistently is it used, who is monitoring for what purpose)? Is there a consensus on the most appropriate measurement method?

There is widespread routine monitoring and reporting on macroinvertebrate community data and metrics in New Zealand [5]. Monitoring is performed by Regional Councils or unitary authorities with two main objectives: (1) assess the ecological condition of estuaries across New Zealand, and (2) to enable temporal changes in condition to be consistently evaluated [5]. Monitoring is classified as either consent monitoring or State of Environment (SOE) monitoring. Consent monitoring is for specific assessment relating to a resource consent, while SOE monitoring has broader focus and is generally long-term and spread over wider geographical area. At national scale, the National Estuary Monitoring Protocol (NEMP [5]) or a modified version of the NEMP is used by 14/16 Regional Councils or unitary authorities, providing resource managers nationally with a scientifically defensible, cost-effective and standardised approach for monitoring the ecological status of estuaries in their region. A recent scoping review of the current NEMP reported that, of the 14 Regional Councils using the NEMP, only 1 council uses it unmodified, 9 councils use a variation of the NEMP, and 4 councils use alternative methods [45]. Recommendations to standardise and improve macroinvertebrate monitoring across New Zealand have been discussed, including on aspects such as the number of samples collected, preservation methods, and level of taxonomic identification required [45]. At present, there is no general consensus as to preferred macroinvertebrate indices and methods of analysis to be used, thus reporting is variable across the country. Recent posting of macroinvertebrate community data to the LAWA website and the development of National Benthic Health Models for mud and metals may facilitate more consistent reporting and interpretation at a national level. A review/update of the NEMP and development of health bands for estuarine indicators is due to be delivered by 30-June-2024.

A4-(ii) Are there any implementation issues such as accessing privately owned land to collect repeat samples for regulatory informing purposes?

The monitoring of macroinvertebrate communities requires field work. Sites are often able to be accessed from shore or by small boat, although there may be areas in estuaries that are only accessible via private land and therefore subject to the rights of landowners. Ancestral and sacred areas, such as areas near burial grounds, are likely to be off limits for environmental monitoring. It is always advisable to communicate with mana whenua to understand access issues. In general, clear communication, good relationships, and addressing concerns or impacts to landowners' property or operations is necessary. Formal access agreements may need to be established in some cases.

The monitoring of intertidal estuarine sites is dependent on the state of the tide, which can limit the access to sites at certain times of day and thus determine the timing of field work. Several health and safety indications also need to be considered for fieldwork. Use of boats and kayaks generally requires health & safety training and Worksafe qualifications. Sinking into deep mud or traversing channels on incoming tides can be fatally hazardous if this risk is not managed. Subtidal sites may be sampled by divers (a highly regulated activity) or by using grab devices; both generally involve boats and the management of health and safety risks.

A4-(iii) What are the costs associated with monitoring the attribute? This includes up-front costs to set up for monitoring (e.g., purchase of equipment) and on-going operational costs (e.g., analysis of samples).

Macroinvertebrate community monitoring is relatively expensive on a per-sample basis. The costs reflect the time and resources required to collect, preserve, store, sort, and identify the invertebrates in the samples to lowest practicable taxonomic resolution. Councils have commented on the high costs of macroinvertebrate monitoring and, as budgets have become tighter, many have had to reduce the scale of the monitoring programmes (fewer sites sampled and/or reduced frequency of sampling). Depending on the number of invertebrates present in a sample, and factors such as presence of seagrass material that can interfere with sorting, it can take many hours of staff time to process a single sample. Quality assurance / quality control protocols (i.e., sample checking) can add to the expense. The costs of monitoring particular sites generally becomes more predictable after they have been sampled a few times. Macroinvertebrate monitoring requires some up-front capital expenditures (boats, kayaks, GPS units, sieves) but many councils and research providers already have these items. The approximate cost to generate macroinvertebrate community data at one site on one occasion (12 replicates)—including fieldwork, materials, laboratory processing time and quality checking—is likely to be in the range of $3,000 to $10,000 (depending on the diversity/difficulty of the samples and the provider used).

A5. Are there examples of this being monitored by Iwi/Māori? If so, by who and how?

There are likely many examples of iwi and hapū representatives monitoring estuaries in New Zealand using a range of mātauranga Māori based and western science based methods. However, to the best of our knowledge, monitoring of macroinvertebrate community composition (comparable to what most Councils do) may not be regularly undertaken by iwi and hapū representatives in New Zealand.

Some of the species that comprise part of the macroinvertebrate community and that are of particular significance and interest to iwi and hapū (e.g., shellfish such as cockles, pipi, and mussels) are indeed monitored by kaitiaki in many parts of New Zealand. This includes the monitoring of pipi by Patuharakeke Te Iwi Trust on intertidal banks in outer Whangārei Harbour, the monitoring of cockles by Ngāti Whakehemo in intertidal soft-sediment habitats of Waihī Estuary, and the monitoring of subtidal mussel populations and beds by Ngāti Awa and the Te Ūpokorehe Resource Management Team in Ōhiwa harbour. Similarly, many hapū and iwi led have co-led and/or driven the assessment of shellfish and associated ecosystem, including Te Papatipu Rūnanga o Koukourarata, Te Papatipu Rūnaka o Ōraka-Aparima, Te Papatipu Rūnaka o Kāti Huirapa ki Puketeraki, and many more. In addition to this, there are many hapū and iwi led shellfish species and ecosystem assessments, e.g., including co-development of appropriate indicators of estuarine mahinga kai [70-72].

Standard methods that local kaitiaki and mana whenua can use to monitor shellfish are described in Ngā Waihotanga Iho (Estuary Monitoring ToolKit; [46]). Versions of Ngā Waihotanga Iho are available in both English and te reo. The degree of use and uptake of Ngā Waihotanga Iho by iwi and hapū, and the degree of method standardisation across New Zealand, is unclear.

A6. Are there known correlations or relationships between this attribute and other attribute(s), and what are the nature of these relationships?

Macroinvertebrate community composition is closely related to the sediment characteristics in coastal ecosystems [2, 23]. Macroinvertebrate communities vary based on the sediment's organic matter content and grain size, especially the sediment's mud content (proportion of particles <63 µm)[2, 29, 33, 47, 48]. Another attribute related to macroinvertebrate communities is the concentration of nutrients in sediment (e.g., total Nitrogen). Elevated concentrations of pore water nutrients may indicate eutrophic conditions, which are sometimes associated with nuisance macroalgal outbreaks, low bottom water oxygen, and decreased macroinvertebrates and functioning [1, 9, 30, 35, 36, 49]. It is increasingly recognised that macroinvertebrate communities are shaped by multiple environmental variables, including climatic, oceanic, freshwater, and local estuarine variables [25-27, 32]. As such, grouping the correlated attributes is not optimal; macroinvertebrate community composition is a robust indicator of ecological health that integrates or encompasses the influence of many other (especially sedimentary) attributes.

Part B—Current state and allocation options

B1. What is the current state of the attribute?

Macroinvertebrate community datasets provide a wealth of information that can be mined to understand the status and trends of estuarine and coastal sites. Our understanding of macroinvertebrate community composition is sufficiently good for it to be used as a national-scale indicator. Steps have already been taken in this direction (i.e., posting of macrofauna data on LAWA [50]; National Benthic Health Model development [8]). As described above, regular monitoring is carried out by most of the Regional Councils across the country using relatively standardised protocols. In addition to Councils, reports assessing macroinvertebrate communities are also commissioned by MfE, which has also contributed to describe the status of this attribute. The general consensus of experts in New Zealand is that macroinvertebrate communities are being impacted by excessive sediment and nutrient inputs to coastal receiving environments [2, 3, 26, 27, 29, 51].

Nevertheless, idiosyncratic/unexplained variation in macroinvertebrate community composition across sites and times, and poor correlations with individual stressors, can be frustrating to managers/kaitiaki seeking simplicity and clarity. Although we believe there is enough understanding for macroinvertebrate communities to be used as indicators of site health [e.g., 8, 26, 52, 53], more research on how to generalise and expand macrofauna based metrics to the national level may be required.

B2. Are there known natural reference states described for New Zealand that could inform management or allocation options?

One of the ways to understand natural or reference conditions is to seek and sample "pristine" locations. Given the widespread influence on humans on our land, oceans, and climate, finding "pristine" locations can be difficult. Secondly, there are many permutations and variations of "healthy"—which impedes the identification of a single clear reference state. Nevertheless, it is relatively easy to identify degraded states in this attribute, which can be compared to healthier locations nearby.

Several indices used internationally and in New Zealand utilise references states in some way. These include the Traits Based Index (TBI, [53]), the Benthic Health Models (BHM, [8]), the Estuary Trophic Index (ETI, [52]), and the Benthic Index of Biotic Integrity (B-IBI, [54]). At least one overseas index has been successfully adapted for use with New Zealand macroinvertebrate community data (e.g., AMBI [13, 55]). However, the ability of the various indices to track stressors and indicate health varies widely [56]. Some authors have tried combining indices to take advantage of each one's individual strengths [57]. However, testing and validation of indices outside of the regions where they were originally developed remains an issue.

B3. Are there any existing numeric or narrative bands described for this attribute? Are there any levels used in other jurisdictions that could inform bands? (e.g., US EPA, Biodiversity Convention, ANZECC, Regional Council set limit)

There are several metrics and numeric bands being used in New Zealand to describe the status of macroinvertebrates communities. The TBI provides an indicator of coastal ecological integrity and resiliency based on macroinvertebrate traits and abundance [53]. Scores > 0.4 are considered 'good', 0.3 – 0.4 are considered 'moderate', and <0.3 indicate 'poor' health and low functional redundancy. However, the use of this scoring system outside of Auckland and Waikato is not advised at this time, nor is the comparison of TBI scores across intertidal and subtidal habitats [58].

Bands for the AMBI have been used in New Zealand to categorise site health [e.g., [59]. However, the appropriateness of the banding system that was developed overseas [healthy to unhealthy reported as 'Very low' 0.0-1.2; 'Low' 1.2-3.3; 'Fair' 3.3-5.0; 'High' 5.0-6.0; and 'Very High' >6.0 to azoic] has not to our knowledge been checked or validated. The original Borja et al. publication [13] had eight benthic community health categories.

Numeric bands for BHMs are also available. Local BHMs developed for Auckland and Bay of Plenty estuaries were expanded to the national level in a 2020 [8]. Although the methods underpinning the BHMs are complex (based on multivariate canonical analysis of principle coordinates), results can be summarised relatively simply in five equally sized categories [Level of impact from lowest to highest: 'Very Low' 1.0 to <2.0, 'Low' 2.0 to >3.0, 'Moderate' 3.0 to <4.0, 'High' 4.0 to <5.0, and 'Very high' ≥5.0]. The National BHMs have been shown to perform well in two estuary types (e.g., tidal lagoons and shallow river valleys) and across five to six regions of New Zealand (Mud BHM: Abel, Banks, Chalmers, Portland, Raglan and Northeastern; Metals BHM: Abel, Southeastern, Portland, Raglan and Northeastern)[8]. Councils appear supportive of the use of the National BHM models, with further testing and refinement urged as more data become available.

B4. Are there any known thresholds or tipping points that relate to specific effects on ecological integrity or human health?

There is general consensus that >25% mud content in intertidal sediments will lead to decreases of macroinvertebrate community integrity [2, 11, 23, 25, 26, 29, 60]. Recent research suggests that this threshold may be slightly higher in the subtidal zone [61]. It has been suggested that >4-5% of organic content in sediment is associated with degraded macroinvertebrate communities [2, 24, 25]. Thresholds for over environmental variables, such as Chlorophyll *a* and coastal sea surface temperatures are not well delimited at this time.

B5. Are there lag times and legacy effects? What are the nature of these and how do they impact state and trend assessment? Furthermore, are there any naturally occurring processes, including long-term cycles, that may influence the state and trend assessments?

Macroinvertebrate communities respond to changes in environmental conditions (individuals immigrate or emigrate, alter their reproduction, or die). These responses are unlikely to be immediately detectable (i.e., they are temporally lagged). Temporally lagged responses of macroinvertebrate communities may occur depending on life history traits (i.e., timing of reproduction, hatch, or settlement) and ecological thresholds (i.e., how tolerant individual species and communities are to particular levels of stress). A recent study suggested that responses of macroinvertebrate communities were site-dependent and lagged in relation to oceanic, climatic, freshwater, and local environmental conditions [32].

At broader scales, legacies of past loadings—particularly the infilling and substantial expansion of mangroves and muddy habitats in our estuaries—may be masking or interfering with the detection of responses to newly loaded contaminants. There is substantial ecological theory on the topic of alternative stable states and hysteresis [62-65]. This work suggests that although elevated loading of catchment contaminants has led to estuarine degradation, we cannot expect estuaries to immediately respond to catchment contaminant reductions. This type of lag is much longer than the ecological lags described above. Although the existence of lagged responses is unequivocal, exploring the significance of lags is difficult.

B6. What tikanga Māori and mātauranga Māori could inform bands or allocation options? How? For example, by contributing to defining minimally disturbed conditions, or unacceptable degradation.

To our knowledge, tikanga Māori and mātauranga Māori has not been considered during the development of macroinvertebrate health/integrity bands. Macroinvertebrate communities, specifically kaimoana (e.g., seafood), are highly valuable for Māori as crucial economic and cultural resources. Tikanga Māori and mātauranga Māori should be included in decision making and band/threshold definition where possible.

Part C—Management levers and context

C1. What is the relationship between the state of the environment and stresses on that state? Can this relationship be quantified?

The relationship between macroinvertebrate communities and environmental stressors is reasonably well understood. There are several studies showing the effects of single and multiple stressors on the macroinvertebrate communities [1, 2, 9, 24, 25, 27, 30-33]. In most of the cases the relationships are quantified and showed deterioration of macroinvertebrate communities with increased environmental stressors. The BHM also quantified the relationship between macroinvertebrate communities and specific stressors such as mud content and metals concentration in sediment [8].

C2. Are there interventions/mechanisms being used to affect this attribute? What evidence is there to show that they are/are not being implemented and being effective?

Macroinvertebrate community composition has utility as an attribute because it is an integrative and responsive indicator of trends in estuarine/coastal ecological integrity. However, most catchment and estuarine interventions are not targeted at improving "macroinvertebrate communities" *per se*. Macroinvertebrates are generally small and cryptic (and tend to be overlooked by non-specialists), but their critical roles in food webs and maintaining a range of life-supporting ecosystem functions demonstrate their importance. Interventions that could affect macroinvertebrate communities include land/freshwater management practices aimed at reducing sediment and nutrient loads into coastal receiving environments. Recent updates to the National Policy Statement for Freshwater Management (NPSFM) are directed as such. Evidence that land/freshwater interventions are leading to improvements in macroinvertebrate community health states has not emerged yet (probably due to the presence of lags and legacy effects mentioned above).

C2-(i). Local government driven

Many councils and local/regional authorities have taken steps to control sediment and nutrient inputs, which should result in the eventual improvement of macroinvertebrate communities. However, current sediment and nutrients inputs to coastal ecosystems are likely still high and driven by external events (such as recent Cyclone Gabrielle). Some of the Jobs for Nature initiatives (while Central government driven) are being implemented locally, but outcomes for macroinvertebrates and estuarine health are not yet known. Some of the 'local' initiatives are being undertaken on relatively large scales (e.g., the $100m Kaipara Moana Remediation project; [66]).

C2-(ii). Central government driven

Central government developed the Essential Freshwater Package to improve and maintain sustainable outcomes from freshwater management and updates the NPSFM approximately every three years. The Jobs for Nature programme (administered by five central government agencies) has directed hundreds of millions of dollars towards riparian planting in catchments to prevent sediments and nutrients from entering freshwater and coastal receiving environments downstream. Obviously, the Jobs for Nature funding was not targeted at improving the macroinvertebrate community composition attribute, though the attribute may be useful at tracking the successes of individual catchment interventions (with the caveat that there will be temporal lags and legacy effects). Several central government agencies are commissioning work on the effects of catchment contaminants in estuarine/coastal ecosystems and/or have strategies for catchment contaminant load reductions, but specific actions may not be widely implemented yet.

C2-(iii). Iwi/hapū driven

Iwi and hapū have been heavily involved in Jobs for Nature projects across New Zealand. Iwi and hapū are also leading estuarine restoration initiatives in partnership with Councils, CRIs, as part of the National Science Challenge programmes from various universities, and with other research institutes/providers [67, 68]. Iwi and hapū have implemented Customary Management Areas tools, including temporary closures in coastal areas to protect shellfish resources (e.g., scallops, pipi). Again, however, we do not know of any iwi/hapū driven initiatives that were specifically designed to improve the condition of macroinvertebrate communities in coastal ecosystems of New Zealand.

C2-(iv). NGO, community driven

As above, although there are many estuary/coast orientated community groups, we know of no initiatives that are being specifically designed to improve the condition of macroinvertebrate communities in coastal ecosystems of New Zealand. Revive Our Gulf is a broad partnership designed to restore mussel reefs in the Hauraki Gulf [69], and there are similar shellfish restoration initiatives at the Top of the South Island.

C2-(v). Internationally driven

To the best of our knowledge there are no initiatives to improve the condition of estuarine/coastal ecosystems or macroinvertebrate communities in coastal ecosystems of New Zealand driven by international entities. However, the Department of Conservation and other agencies set many of their policy goals to align with Convention of Biological Diversity targets, e.g., CBD Aichi Target 11 (biodiversity and ecosystem services are conserved using effective area-based conservation measures integrated into wider landscapes and seascapes).

Part D—Impact analysis

D1. What would be the environmental/human health impacts of not managing this attribute?

Degradation of macroinvertebrate communities would affect the ecological integrity of coastal ecosystems, as explained in previous sections. Ignoring the management of macroinvertebrate communities could lead to reductions in the delivery of key ecosystem functions and services. For example, less diverse and abundant macroinvertebrate communities may result in reduced organic matter cycling and nutrient removal, essential processes for healthy estuarine/coastal ecosystems. It will also result in less food available for higher trophic levels, and reduced pollutant detoxification capacity. Macroinvertebrates are often habitat-defining species (e.g., “cockle bed”, “tube-worm mat”, “crab burrow habitat”) that modify the environment and facilitate/inhibit other organisms—

signifying their fundamental roles in estuarine/coastal ecosystems. Not managing macroinvertebrate communities could impact Māori, particularly with estuarine/coastal kaimoana supporting whānau nutritionally, economically, and culturally.

D2. Where and on who would the economic impacts likely be felt? (e.g., Horticulture in Hawke's Bay, Electricity generation, Housing availability and supply in Auckland)

Impacts of macroinvertebrate community degradation may manifest as increased eutrophication, reduced food available for fish and birds, increased deposition of sediment, and altered sediment geochemistry/oxygenation [2, 3, 30, 31, 33, 35, 48]. Furthermore, coastal ecosystems will be impacted economically by reduced shellfish fisheries and less cultural activities (e.g., tourism).

D3. How will this attribute be affected by climate change? What will that require in terms of management response to mitigate this?

Increasing temperatures, frequency of storms, and heatwaves (i.e., climate change), are expected to exacerbate the pressures/stressors on macroinvertebrate communities [25, 27, 32]. For example, increasing temperatures and storms will result in increased nutrient and sediment inputs, anoxia, and eutrophication events. Sea level rise is also expected to alter the proportions of intertidal and subtidal habitats, and there is a great likelihood that estuarine morphology (the positions of tidal creeks and sediment accretion/deposition zones) may change. Management actions should focus on limiting sediment and nutrient inputs from terrestrial sources entering to coastal ecosystems, to reduce the risk of worsening the condition of the macroinvertebrate communities. Although carbon reduction strategies and a reduced reliance on fossil fuels is essential, the ability of New Zealand to influence overall climate change trajectories may be small.

References:

1. Douglas, E.J., et al., Macrofaunal Functional Diversity Provides Resilience to Nutrient Enrichment in Coastal Sediments. Ecosystems, 2017. 20(7): p. 1324-1336.

2. Lohrer, A.M., et al., Terrestrially derived sediment: response of marine macrobenthic communities to thin terrigenous deposits. . Marine Ecology Progress Series, 2004(273): p. 121-138.

3. Thrush, S.F., et al., Changes in the location of biodiversity-ecosystem function hot spots across the seafloor landscape with increasing sediment nutrient loading. Proc Biol Sci, 2017. 284(1852).

4. Thrush, S.F., et al., The many uses and values of estuarine ecosystems. Ecosystem services in New Zealand–conditions and trends. Manaaki Whenua Press, Lincoln, New Zealand, 2013: p. 226-237.

5. Robertson, B.M., et al., Estuarine Environmental Assessment and Monitoring: A National Protocol. Prepared for supporting Councils and the Ministry for the Environment, Sustainable Management Fund., 2002. Contract No. 5096.: p. 93-159.

6. Hewitt, J.E., Hailes. S.F., and B.L. Greenfield, Protocol for Processing, Identification and Quality Assurance of New Zealand Marine Benthic Invertebrate Samples. Prepared for Northland Regional Council, 2014: p. 36.

7. Greenfield, B.L., et al., Protocol for the Identification of Benthic Estuarine and Marine Macroinvertebrates: A guide for parataxonomists. C-SIG Coastal Taxonomic Resource Tool. 40 p + appendix., 2023.

8. Clark, D.E., et al., The development of a national approach to monitoring estuarine health based on multivariate analysis. Marine Pollution Bulletin, 2020. 150: p. 110602.

9. Douglas, E.J., A.M. Lohrer, and C.A. Pilditch, Biodiversity breakpoints along stress gradients in estuaries and associated shifts in ecosystem interactions. Scientific Reports, 2019. 9(1): p. 17567.

10. Lam-Gordillo, O., et al., Restoration of benthic macrofauna promotes biogeochemical remediation of hostile sediments; An in situ transplantation experiment in a eutrophic estuarine-hypersaline lagoon system. Sci Total Environ, 2022. 833: p. 155201.

11. Lohrer, A., S. Thrush, and M. Gibbs, Bioturbators enhance ecosystem function through complex biogeochemical interactions. . Nature, 2004. 431: p. 1092–1095.

12. Kristensen, E., et al., What is bioturbation? The need for a precise definition for fauna in aquatic sciences. Marine Ecology Progress Series, 2012. 446: p. 285-302.

13. Borja, A., J. Franco, and V. Pérez, A marine biotic index to establish the ecological quality of soft-bottom benthos with European estuarine and coastal environments. Marine Pollution Bulletin, 2000. 40: p. 1100–1114.

14. Veríssimo, H., et al., Assessment of the subtidal macrobenthic community functioning of a temperate estuary following environmental restoration. Ecological Indicators, 2012. 23: p. 312-322.

15. Drylie, T.P., Marine ecology state and trends in Tāmaki Makaurau / Auckland to 2019. State of the environment reporting. . Auckland Council technical report. TR2021/09, 2021.

16. Lam-Gordillo, O., et al., Loss of benthic macrofauna functional traits correlates with changes in sediment biogeochemistry along an extreme salinity gradient in the Coorong lagoon, Australia. Mar Pollut Bull, 2022. 174: p. 113202.

17. Pearson, T.H. and R. Rosenberg, Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. Oceanogr. Mar. Biol. Annu. Rev., 1978. 16: p. 229-311.

18. Villnäs, A., et al., Changes in macrofaunal biological traits across estuarine gradients: implications for the coastal nutrient filter. Marine Ecology Progress Series, 2019. 622: p. 31-48.

19. van der Linden, P., et al., Spatial and temporal response of multiple trait-based indices to natural- and anthropogenic seafloor disturbance (effluents). Ecological Indicators, 2016. 69: p. 617-628.

20. Villnas, A., et al., Consequences of increasing hypoxic disturbance on benthic communities and ecosystem functioning. PLoS One, 2012. 7(10): p. e44920.

21. Cloern, J.E., et al., Human activities and climate variability drive fast-paced change across the world's estuarine–coastal ecosystems. Globlal Change Biology, 2016. 22: p. 513–529.

22. de Juan, S., et al., Cumulative degradation in estuaries: contribution of individual species to community recovery. Marine Ecology Progress Series, 2014. 510: p. 25-38.

23. Douglas, E.J., et al., Changing intra- and interspecific interactions across sedimentary and environmental stress gradients. Ecosphere, 2023. 14: p. e4373.

24. Ellis, J., et al., Effects of terrigenous sedimentation on mangrove physiology and associated macrobenthic communities. Marine Ecology Progress Series, 2004. 270: p. 714-82.

25. Ellis, J.I., et al., Multiple stressor effects on marine infauna: responses of estuarine taxa and functional traits to sedimentation, nutrient and metal loading. Scientific Reports, 2017. 7(12013).

26. Ellis, J.I., et al., Assessing ecological community health in coastal estuarine systems impacted by multiple stressors. Journal of Experimental Marine Biology and Ecology, 2015. 473: p. 176-187.

27. Hewitt, J.E., J.I. Ellis, and S.F. Thrush, Multiple stressors, nonlinear effects and the implications of climate change impacts on marine coastal ecosystems. Global Change Biology, 2016. 22: p. 2665-2675.

28. Lohrer, A.M., et al., Ecosystem functioning in a disturbance-recovery context: Contribution of macrofauna to primary production and nutrient release on intertidal sandflats. Journal of Experimental Marine Biology and Ecology, 2010. 390(1): p. 6-13.

29. Thrush, S.F., et al., Muddy waters: elevating sediment input to coastal and estuarine habitats. Frontiers in Ecology and the Environment, 2004. 2(6): p. 299-306.

30. Thrush, S.F., et al., Cumulative stressors reduce the self-regulating capacity of coastal ecosystems. Ecological Applications, 2021. 31(1): p. e02223.

31. Clark, D.E., et al., Influence of land-derived stressors and environmental variability on compositional turnover and diversity of estuarine benthic communities. Marine Ecology Progress Series, 2021. 666: p. 1-18.

32. Lam-Gordillo, O., et al., Climatic, oceanic, freshwater, and local environmental drivers of New Zealand estuarine macroinvertebrates. Marine Environmental Research, 2024. 197: p. 106472.

33. Douglas, E.J., et al., Sedimentary Environment Influences Ecosystem Response to Nutrient Enrichment. Estuaries and Coasts, 2018. 41: p. 1994–2008.

34. Hicks, M., et al., Updated sediment load estimator for New Zealand. Prepared for the Ministry for the Environment. NIWA Client Report 2018341CH., 2019: p. 190.

35. Hillman, J.R., et al., Loss of Large Animals Differentially Influences Nutrient Fluxes Across a Heterogeneous Marine Intertidal Soft-Sediment Ecosystem. Ecosystems, 2020. 24(2): p. 272-283.

36. Lohrer, A.M., et al., Detecting shifts in ecosystem functioning: the decoupling of fundamental relationships with increased pollutant stress on sandflats. Mar Pollut Bull, 2012. 64(12): p. 2761-9.

37. Thrush, S.F., et al., The effects of habitat loss, fragmentation, and community homogenization on resilience in estuaries. Ecological Applications, 2008. 18(1): p. 12-21.

38. Thrush, S.F., et al., Multiple stressor effects identified from species abundance distributions: Interactions between urban contaminants and species habitat relationships. Journal of Experimental Marine Biology and Ecology, 2008. 366: p. 160-168.

39. Thrush, S.F., et al., Experimenting with ecosystem interaction networks in search of threshold potentials in real-world marine ecosystems. Ecology, 2014. 95(6): p. 1451-1457.

40. Robertson, B.M., et al., Condition of Southland’s shallow, intertidal dominated estuaries in relation to eutrophication and sedimentation: Output 1: Data analysis and technical assessment - habitat mapping, vulnerability assessment and monitoring recommendations related to issues of eutrophication and sedimentation. Report prepared by Wriggle Coastal Management for Environment Southland: 172., 2017.

41. Lohrer, D., et al., Findings from an interdisciplinary workshop on Jacobs River Estuary. Prepared by NIWA for Dairy NZ Ltd., NIWA Client Report No: 2020148HN (project DNZ20205), 2020.

42. Jones, H.F.E., et al., Historical data provides context for recent monitoring and demonstrates 100 years of declining estuarine health. New Zealand Journal of Marine and Freshwater Research, 2022. 56(3): p. 371-388.

43. Hunt, S., Summary of historic sedimentation measurements in the Waikato region and formulation of a historic baseline sedimentation rates. WRC Tech. Report 2019/08., 2019.

44. Hume, T.M. and J.G. Gibb, Determining historical sedimentation rates in some New Zealand estuaries. Journal of the Royal Society of New Zealand., 1987. 17(1): p. 1-7.

45. Roberts, K.L. and L.M. Stevens, Scoping review to update the National Estuary Monitoring Protocol. . Salt Ecology Report 115, prepared for Ministry for the Environment, June 2023. 30p., 2023.

46. Ngā Waihotanga Iho: Estuary Monitoring ToolKit. https://niwa.co.nz/te-kuwaha/tools-and-resources/ng%C4%81-waihotanga-iho-the-estuary-monitoring-toolkit.

47. Lohrer, A.M., et al., Deposition of terrigenous sediment on subtidal marine macrobenthos: response of two contrasting community types. Marine Ecology Progress Series, 2006. 307: p. 115-125.

48. Pratt, D.R., et al., Changes in Ecosystem Function Across Sedimentary Gradients in Estuaries. Ecosystems, 2013. 17(1): p. 182-194.

49. O'Meara, T., et al., Rapid organic matter assay of organic matter degradation across depth gradients within marine sediments. Methods in Ecology and Evolution, 2017. 9(2): p. 245-253.

50. LAWA, *https://www.lawa.org.nz/explore-data/estuaries/*.

51. Lohrer, A., et al., Exposure of estuarine sites to freshwater contaminants: Influences on estuarine macrofauna time-series data. . Prepared for Ministry for the Environment. NIWA report no. 2023089HN, MFE22204. , 2023: p. 80.

52. Zeldis, J.R. and D.R. Plew, Predicting and Scoring Estuary Ecological Health Using a Bayesian Belief Network. Frontiers in Marine Science, 2022. 9.

53. Rodil, I.F., et al., Tracking environmental stress gradients using three biotic integrity indices: Advantages of a locally-developed traits-based approach. Ecological Indicators, 2013. 34: p. 560-570.

54. Weisberg, S.B., et al., An estuarine benthic index of biotic integrity (B-IBI) for Chesapeake Bay. Estuaries, 1997. 20(1): p. 149-158.

55. Robertson, B.P., J.P.A. Gardner, and C. Savage, Macrobenthic–mud relations strengthen the foundation for benthic index development: A case study from shallow, temperate New Zealand estuaries. Ecological Indicators, 2015. 58: p. 161-174.

56. Berthelsen, A., et al., Relationships between biotic indices, multiple stressors and natural variability in New Zealand estuaries. Ecological Indicators, 2018. 85: p. 634-643.

57. Hewitt, J.E., D. Lohrer, and M. Townsend, Health of Estuarine Soft-sediment Habitats: continued testing and refinement of State of the Environment indicators. Prepared by NIWA for Auckland Council. Auckland Council Technical Report 2012/12. , 2012.

58. Lohrer, D., Health metrics and thresholds. Letter report to the Ministry for the Environment (16-July-2023; NIWA Project MFE22204). 15 pp., 2023.

59. Robertson, B.M. and L.M. Stevens, Jacobs River Estuary Fine Scale Monitoring 2010/11. Report prepared by Wriggle Coastal Management for Environment Southland. 34p., 2011.

60. Jones, H.F.E., et al., Sedimentary Environment Influences the Effect of an Infaunal Suspension Feeding Bivalve on Estuarine Ecosystem Function. PLOS ONE, 2011. 6(10): p. e27065.

61. Lohrer, D., et al., Narratives to inform land-to-sea management based on estuarine and coastal benthic data. Prepared for Ministry for the Environment. NIWA report 2023213HN (project MFE22204), 19 p., 2023.

62. van de Koppel, J., et al., DO ALTERNATE STABLE STATES OCCUR IN NATURAL ECOSYSTEMS? EVIDENCE FROM A TIDAL FLAT. Ecology, 2001. 82(12): p. 3449-3461.

63. Mac Nally, R., C. Albano, and E. Fleishman, A scrutiny of the evidence for pressure-induced state shifts in estuarine and nearshore ecosystems. Austral Ecology, 2014. 39(8): p. 898-906.

64. Capon, S.J., et al., Regime shifts, thresholds and multiple stable states in freshwater ecosystems; a critical appraisal of the evidence. Science of The Total Environment, 2015. 534: p. 122-130.

65. McGlathery, K.J., et al., Nonlinear Dynamics and Alternative Stable States in Shallow Coastal Systems. Oceanography, 2013. 26(3): p. 220-231.

66. Kaipara, *https://kmr.org.nz/*.

67. Waihi, *https://www.waihi-estuary.iwi.nz/*.

68. Tehoiere, *https://www.tehoiere.org.nz/*.

69. Hauraki, *https://www.reviveourgulf.org.nz/*.

70. Kainamu-Murchie, A., Williams, A., Kitson, J., Dallas, R., Ratana, K., May. K. (2021) Ngā taonga waimātaitai: Co-developing appropriate tangata whenua approaches to improve estuarine mahinga kai management. Technical report for Te Rūnaka o Ōraka-Aparima. 55p

71. Te Rūnanga o Ngāi Tahu (2004) State of the Takiwā. Cultural Monitoring and Reporting on the Health of our Environment. Te Rūnanga o Ngāi Tahu, Christchurch, New Zealand: 28.

72. Pauling, C. (2005) State of the Takiwā. Cultural Monitoring and Reporting on the Health of our Environment. Te Āhuatanga o Te Waiau, Te Rūnanga o Ngāi Tahu, ESR, EnviroLink, Ministry for the Environment, Manaaki Whenua, Environment Southland. Aotearoa, New Zealand: 88.