CAWTHRON

Monitoring Waste Water Discharges in Areas Of Freshwater-Estuarine Transition

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Hawkes Bay Regional Council

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1. INTRODUCTION

This report is part of an Envirolink small advice grant to provide Hawkes Bay Regional Council with advice on approaches to the biomonitoring of discharges within the lowland streams/rivers that are affected by saltwater. Specifically, there has been increasing concern around the ability to appropriately assess the environmental condition of estuarine areas downstream of wastewater discharges and where traditional biomonitoring indices like the Macroinvertebrate Community Index (MCI) lose their efficacy. The focus of this report, therefore, is on the pros, cons and limitations of these traditional indices, with some additional discussion on alternative approaches and references to supporting guidance documents when standard biomonitoring is deemed insufficient.

1.1. Background on biomonitoring

In flowing, freshwater environments, macroinvertebrate community indices are among the most widely used biomonitoring tools used to assess water quality and to detect the impacts of point- and non-point source discharges of pollutants in aquatic environments.

Macroinvertebrate community indices are widely used in monitoring of freshwater quality internationally (Rosenberg & Resh, 1993) and have several advantages for monitoring the effects of non-point source pollution. Macroinvertebrates are ubiquitous; they are found in most aquatic environments and include a wide range of species that vary in their responses to stressors. In addition, many species have limited mobility which allows for some spatial consideration of the effects of stressors. The relatively long life-cycles of many macroinvertebrates (months to years) mean that variations in community composition in response to a stressor will persist for several weeks to months (Rosenberg & Resh, 1993). However, macroinvertebrates also pose some disadvantages for biomonitoring, including the relatively high level of sampling that is often required to attain adequate precision, complications in the interpretation of biotic indices resulting from seasonal variation in community composition and individuals drifting from upstream reaches or tributaries, taxonomic uncertainty for some groups and the fact that invertebrates may not respond to all stressors, or may be affected by other factors (Rosenberg & Resh, 1993).

2. BIOTIC INDICES

2.1. The macroinvertebrate community index and related indices

In New Zealand, the most widely used macroinvertebrate index is the macroinvertebrate community index (MCI, Stark 1985) and related indices (semi-quantitative MCI (SQMCI), quantitative MCI (QMCI) and the corresponding soft-bottomed versions – Stark & Maxted 2007). The MCI family of indices was originally developed to assess the effects of organic enrichment by sampling in the riffles of stony streams (Stark 1985), although it has subsequently been found to be appropriate when assessing nutrient enrichment and sedimentation in stony and soft-bottomed streams (Stark & Maxted 2007). As with most macroinvertebrate-based biotic indices, the foundation of the MCI family of indices is the taxon scores that are used to calculate the index. These taxon scores are based on the tolerance of individual taxa to the pollutant of interest and in the case of the MCI, taxa with high scores (7-10) are intolerant of enrichment (such as mayflies, stoneflies and caddis flies) while tolerant taxa (such as many true flies, worms, snails) have low scores (<3). Usually these scores are derived using professional judgement, multivariate or iterative techniques (e.g. Chessman, Growns & Kotlash, 1997) from macroinvertebrate community data collected from rivers that cover a gradient of the pollutant in question. When calculating MCI scores for various sites, the identities of the different species comprising the community are not important, but rather the balance of species that are tolerant of enrichment versus those that are not.

2.2. Limitations of the MCI

The MCI family of indices were developed for the assessment of nutrient enrichment and sedimentation in gravel/cobble bed (MCI, SQMCI, QMCI) and soft-bottomed streams (the soft-bottomed MCI, SQMCI and QMCI) (Stark & Maxted 2007). These indices have not been evaluated for other habitat types, such as lakes, ponds, wetlands, large non-wadeable rivers or hot springs, or for stressors other than enrichment/sedimentation and, consequently, their performance is not assured when applied beyond the systems in which they were developed.

Given the types of systems that the MCI and its derivatives were developed for, Stark & Maxted (2007) recommend that sampling in stony streams should be limited to water depths of 0.1-0.4 m, current velocities of 0.2-1.2 m s^{-1} and median substrate of 60-140 mm, where possible.

Because the MCI was developed for freshwater stream systems, no attempt was made to take account of the salinity preferences of individual taxa before assessing the relationship with the stressor gradient. For many taxa, salinity may be the key driver of their distribution and abundance, with a secondary response to other variables (such as enrichment). If the MCI is to be applied in areas influenced by saltwater, it is suggested that the appropriateness of tolerance scores used should be verified, as recommended by Stark & Maxted (2007).

2.3. Factors complicating the interpretation of the MCI

Interpretation of biotic indices can be complicated by factors other than the pollution or stressor gradient being investigated. One such factor that complicates the interpretation of the MCI is the longitudinal change in community composition resulting from changes in morphology (channel shape, gradient, sediment characteristics), water temperature and vegetation from the headwaters to lowland sections. An even steeper gradient of change in macroinvertebrate community structure occurs where the lowland reaches of rivers enter tidally-affected sections and become brackish. Such changes may be expected to result in differences in MCI scores between sites, even in the absence of any impact of the discharge being monitored. Therefore, the application of MCI-like indices in areas where such strong longitudinal gradients are likely requires special care.

3. APPROACHES TO OVERCOME CONFOUNDING FACTORS IN BIOMONITORING

3.1. Reference sites

MCI scores taken at a single location are often interpreted against ‘water quality classes’, such as those of Stark & Maxted (2007). Such an approach is usually taken in State of the Environment (SoE) monitoring where the general water quality in a catchment is of interest, rather than the effect of a specific discharge or activity. Such a single-site approach provides information on the cumulative effects of all activities upstream of that location on water quality.

In the case of biomonitoring being applied to a specific discharge and/or activity, it is first necessary to determine what the background water/habitat quality is upstream of the discharge. This is generally done by establishing a reference (or control) site upstream of the discharge. The impact of the discharge/activity of interest is then interpreted relative to this control site (Figure 1). The selection and location of the reference site is probably the most important part of designing a sampling regime to detect the effect of a discharge. There is nothing that can be done after the fact to overcome a confounded study design resulting from a poorly selected reference site.

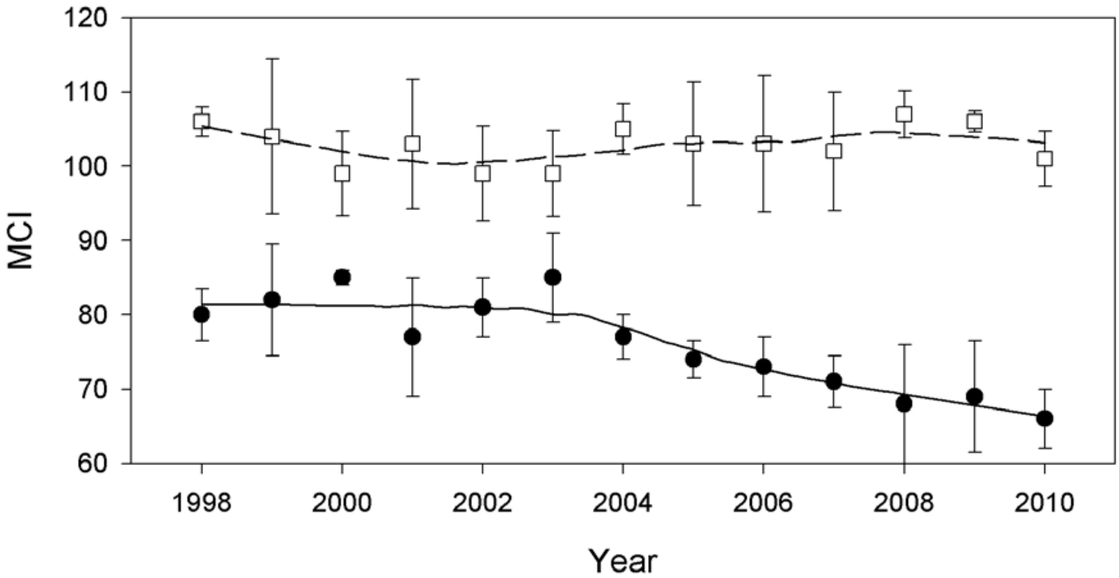

Figure 1. Comparison of MCI scores at an impacted site (closed circles, solid line) with those at an upstream reference site (open squares, dashed line). Lines are fitted LOWESS curves (tension=0.5).

There are several factors to consider when establishing an upstream reference site to ensure that the biomonitoring programme will effectively detect the impact (if any) of the discharge being monitored. These include:

1. Location – the control site should be <u>as close as practicable</u> to the discharge point to minimize any potential for extraneous factors (*i.e.* those not related to the discharge) causing a difference in MCI scores between the reference and impacted sites (*i.e.* confound the effect of the impact).
2. Habitat – the control and impact sites should be located in habitats that are as similar as practical to reduce the likelihood of any differences in MCI being attributable to differences in habitat.
3. Physicochemical conditions – any difference in physicochemistry (temperature, dissolved oxygen, salinity *etc.*) between control and impact sites should be a result of the discharge of interest only.

One of the strong points of the MCI and related indices is that taxon identity is not important in the final calculation of the index. All the taxa in the sample are identified and these data are used to determine the taxon scores used in the calculation of the index. However, two taxa may have the same taxon score – and in the calculation of the MCI, substitution of one for the other makes no difference to the final score. In a practical sense, what this means is that the MCI will not be sensitive to minor changes in macroinvertebrate community composition resulting from slight differences in conditions between sites and/or years. Difficulties arise when determining how great differences between sites can be before the interpretation of the index is compromised (an approach to determine this is discussed later – see section 4).

3.1.1. *Modelled reference condition*

One approach to determining reference condition in the absence of suitable reference sites is to use a predicted reference condition from a national-scale invertebrate model (John Leathwick, unpublished data, Clapcott & Goodwin 2010). This approach is taken in the British RIVPACS (Clarke *et al.* 2003) and the Australian AUSRIVAS (Davies 1997). However, at present, the models developed in New Zealand have not been adequately verified at a regional scale and it is not clear how well they will deal with salt water-affected reaches.

3.2. Time-series analysis

In cases where no reference site is available, time-series analysis can be used to detect the effect of a discharge on water quality, if it is possible to establish a robust base-line prior to the discharge. The impact of the discharge can then be interpreted relative to the pre-impact data set (Figure 2). The robustness of this approach depends on how reliable the pre-impact data set is.

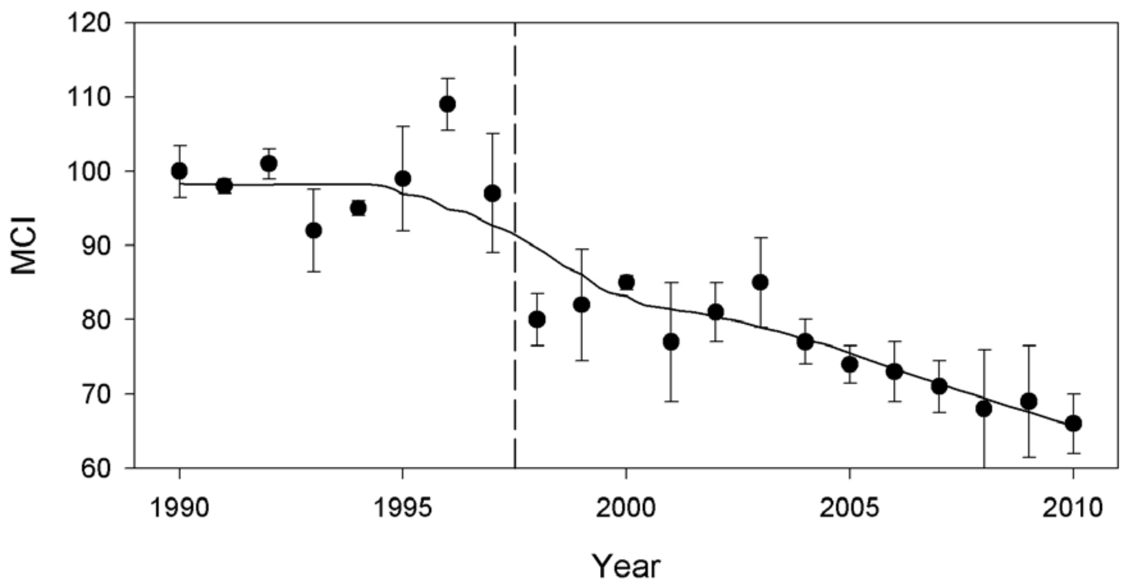

Figure 2. MCI scores at an impacted site through time prior-to and after the establishment of a discharge. The discharge started in 1998 (dashed line). Lines are fitted LOWESS curves (tension=0.5).

4. USING THE MCI IN THE LOWER REACHES OF RIVERS

As discussed in Section 3.1, one of the positive attributes of the MCI is its insensitivity to minor changes in community composition resulting from minor differences in habitat that are unrelated to the enrichment/sedimentation stressor gradient of interest. However, there comes a point when factors unrelated to the impact being monitored could confound the interpretation of the MCI. This section presents a case study where a discharge is located within a section of river that is influenced by saline water and explores options for assessing the effects of this discharge.

4.1. Case study – Porangahau River monitoring

4.1.1. Background

Monitoring is currently conducted on the Porangahau River as part of assessments of the effects of the Porangahau closed land fill and the discharge from the Porangahau waste water treatment plant (WWTP). In both cases, discharges are located within the lower river within the transition between freshwater and estuarine systems.

In the case of the Porangahau closed land fill monitoring, the upstream (reference – Site 1 in Figure 3) and downstream (impacted - Site 2, Figure 3) sites differ in their habitat and physicochemistry, with the downstream site being deeper, having finer substrate (mud/silt vs. gravels upstream), and a greater saline influence (higher electrical conductivity, dissolved potassium and dissolved chloride concentrations) (Hamill 2010). Given the tidal influence at the discharge point, the waste is likely to affect areas both upstream and downstream of the discharge point, with waste being transported upstream on the rising tide and flowing downstream during ebbing tides. This is likely to account for the distance between the upstream reference site and the site 100 m downstream of the discharge. However, the large inter-site differences make meaningful interpretation of biotic indices such as the MCI, highly unlikely in this case.

Assessments of the Porangahau WWTP discharge use the two sites used to assess the Porangahau closed land fill, which are 2 km and 1.3 km upstream (Sites 1 and 2 in Figure 3, respectively) of the discharge, in addition to another site 100 m downstream of the discharge (Site 3 in Figure 3). Given the distance between the two upstream sites and the downstream site (>1.3 km), it is doubtful that MCI can be meaningfully interpreted in this case as well.

Figure 3. Map of the lower Porangahau River showing the location of biomonitoring sites (red circles) and the locations of salinity loggers presented in Figure 4 (blue circles). Biomonitoring undertaken as part of the assessments of the Porangahau WWTP discharge is carried out at Sites 1-3. Sites 1 and 2 are also monitored as part of assessments of closed landfill sites.

4.1.2. Salinity in Porangahau River

Electrical conductance loggers were placed at four locations in the Porangahau River over the period 11-25 August 2010 to investigate the extent of the saltwater influence. These results indicate that at high tide, seawater reaches at least as far upstream as the bridge at Beach Road (Figure 4c). Site 2 shows a very small saltwater influence during the early part of the monitoring period indicating that this site was very close to the upstream limit of the saltwater influence (Figure 4b). The upstream limit of saltwater will be influenced by tide height and river flows. For example, electrical conductance was low at Sites 3 and 4 shortly after rainfall, indicating elevated flows in the Porangahau River (Figure 4c-d).

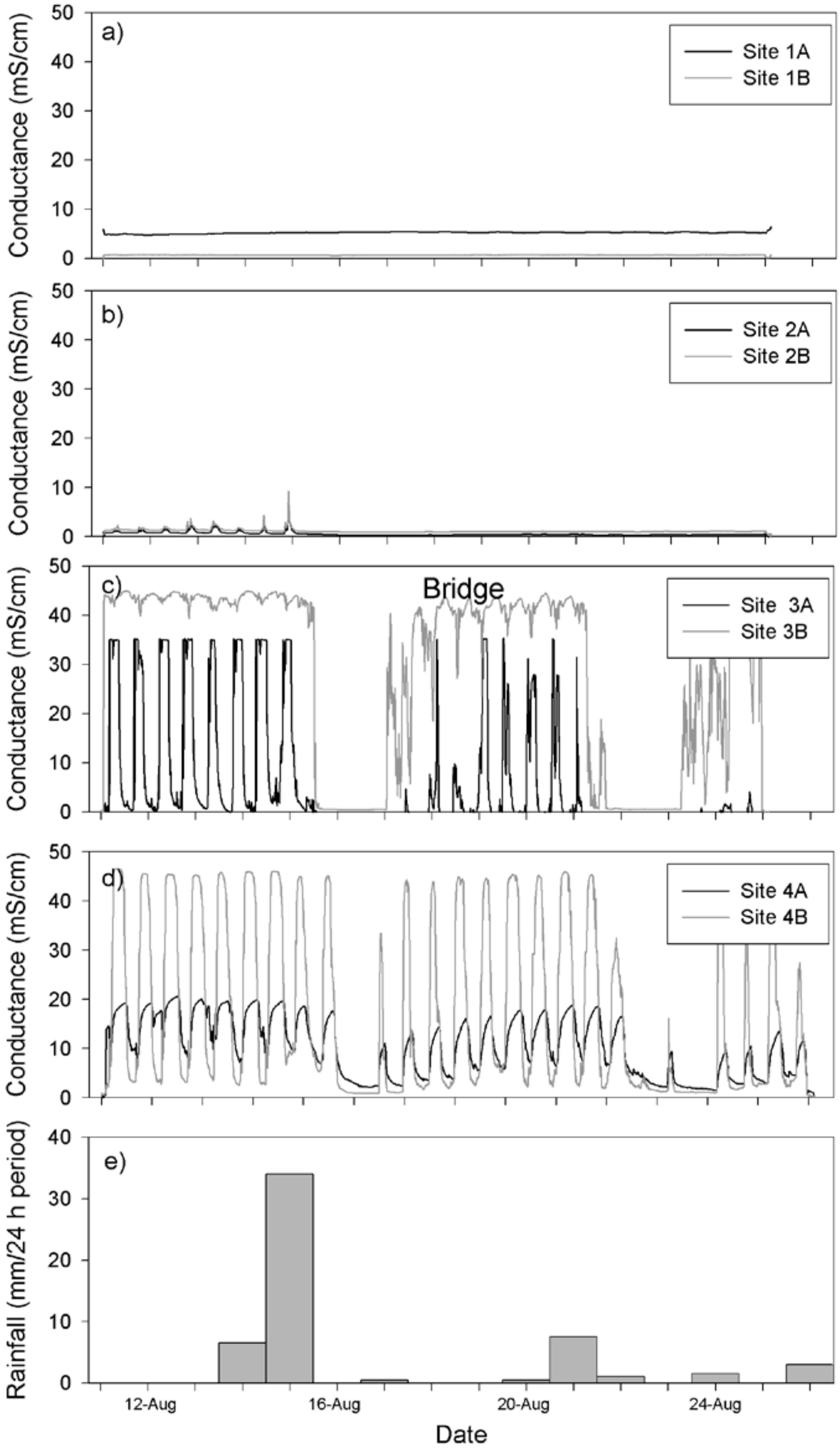

Figure 4. Electrical conductance at four locations in the Porangahau River over the period 11-25 August 2010 and daily rainfall at Ben Nevis.

4.1.3. Determining where the MCI can be used?

As discussed in Section 2, the MCI was developed for freshwater macroinvertebrates. Therefore, saltwater influences will complicate interpretation of biomonitoring in lowland streams where the discharge is located within the transition between freshwater and estuarine systems. Until such time as the performance of the MCI has been verified in such environments, its performance is unknown. This, in addition to the difficulties in establishing a comparable reference site (or reference time-series) leads to the recommendation that the MCI should not be used in stream reaches that are influenced by saltwater.

This recommendation relies on the identification of the point at which invertebrate community structure becomes modified by saltwater influences. Figure 5 outlines a conceptualisation of the changes within the freshwater-estuarine transition. Macroinvertebrate samples and physicochemical measurements across the full gradient of saltwater influence (from wholly freshwater sites to estuarine sites) could be used to delineate the transition in communities and physicochemistry within this part of the river. This information could be used to define the downstream limit for the use of the MCI in biomonitoring.

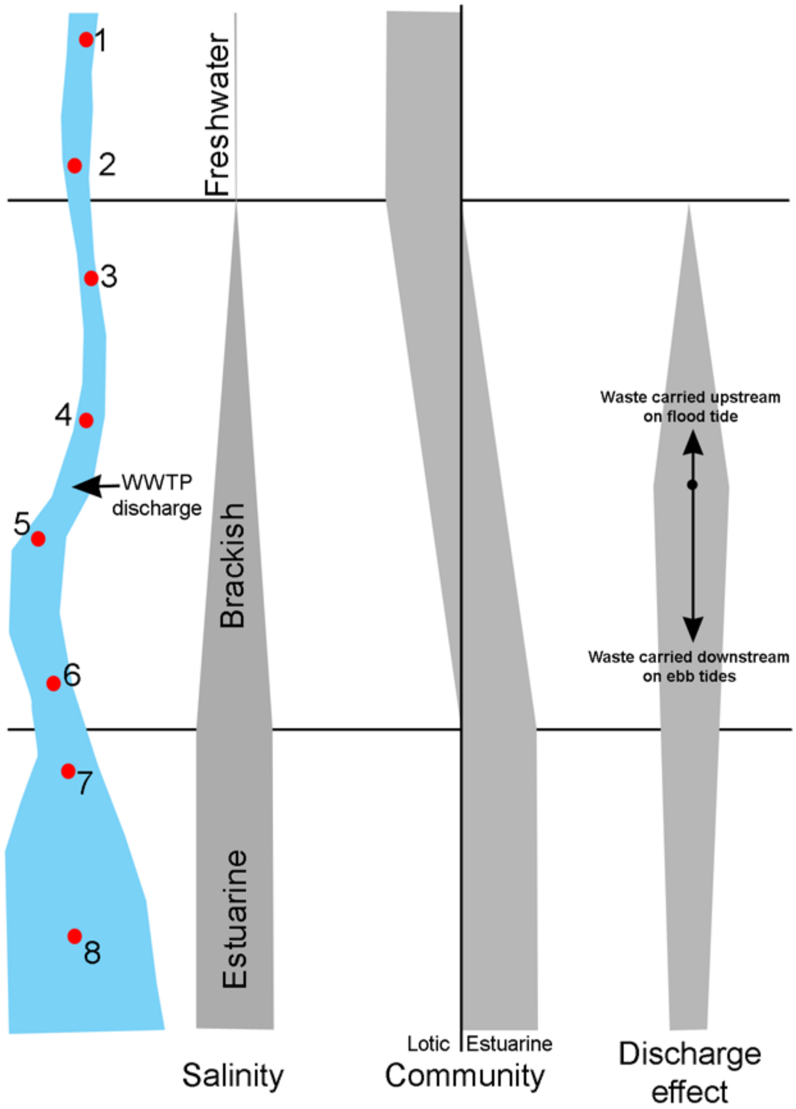

Figure 5. Conceptual diagram of transitions in the physicochemical conditions and community composition in the lower reaches of the Porangahau River and the effects of the WWTP discharge.

5. ALTERNATIVE OPTIONS FOR ECOLOGICAL MONITORING IN TRANSITIONAL ZONES

While MCI itself is not a stand alone solution for ecological monitoring in transitional zones, even if it were applicable, assessing adverse effects from wastewater discharges in estuarine environments often relies on a 'weight of evidence' approach. This approach generally uses a range of different indicators including physicochemical analysis of the sediments for indicators of enrichment (*i.e.* grain size, organic carbon, nutrients, *etc.*), and analysis of sediment dwelling biota (*sans* MCI). Along with these common indicators, other emerging tools like functional indicators and microbial source tracking (MST) are increasingly being used for routine assessments. There is considerable guidance on the common approaches contained in reference documents like the NZ Wastewater Monitoring Guidelines (NZWERF 2002) and the Estuarine Monitoring Protocols (Robertson *et al.* 2002). Some of the emerging tools are described briefly below.

5.1. Common approaches to wastewater monitoring

The range and potential magnitude of effects from wastewater and other organic-rich discharges are particularly well-documented and understood. Determination of enrichment effects is commonly assessed through physico-chemical analysis of the sediments for indicators of enrichment (*i.e.* grain size, organic carbon, nutrients, *etc.*), and by analysing benthic and epibenthic biota.

Sediment texture or grain size is an important variable when studying estuarine chemistry and ecology. It plays a significant role in determining the chemical make-up of sediment (*e.g.* muddy sediments tend to have relatively high organic and nutrient contents), and the range of organisms that may live in it (*e.g.* the types of species encountered in sandy sediments are generally different to those in muddy sediments). Sediment texture therefore provides a measure of the physical characteristics of an area that can be used to better interpret differences between sites for other parameters, such as organic content.

As with sediment texture, organic analyses (*i.e.* for nitrogen and carbon) are typically used to help assess the degree of enrichment around wastewater outfalls. Typically, this involves analysis of total carbon and nitrogen but can also include other forms such as dissolved inorganic nitrogen or the stable isotopes of carbon and nitrogen.

Along with these physicochemical analyses, the analysis of sediment-dwelling infauna, can be a sensitive indicator of wastewater effects with or without using MCI. For example, the composition of the macrofaunal community, the number of different species and the number of individuals of a given species, all provide a valuable indication of the quality of soft-sediment habitats.

While estuarine environments are subject to multiple different inputs and have variable flow patterns, analysis of both the physicochemical and biological indicators of enrichment can be used to also assess the relative scale of the effect using a 'weight of evidence' approach. For example, even in the absence of MCI, the relative richness and abundance, particularly of opportunistic taxa, can offer a good indication as to the scale of effect.

5.2. Functional indicators

Functional indicators of ecosystem function measure the rates of ecosystem processes as a way of assessing the ecosystem 'health' and are increasingly being used as part of regular monitoring programmes (Bunn 1995, Gessner & Chauvet 2002). Processes that have shown promise as functional indicators include the rate of wood decomposition and ecosystem metabolism (Young & Huryn 1999; Bunn & Davies 2000; Gessner & Chauvet 2002; Young *et al.* 2004; Young & Collier 2009). One advantage of functional indicators is that they allow flexibility in the types of habitats able to be assessed (Young *et al.* 2004) and it has been suggested that they may overcome the problems associated with biomonitoring in transition zones, such as in the lower reaches of tidal rivers (Young 2007), as is being considered here.

Young (2007) considered the effectiveness of using wood decomposition rates to consider the effects of a waste discharge to the lower Hokitika River where tidal influences on water flow (but not salinity) caused differences between reference and impacted sites. He found that wood decomposition rates were affected by site-specific factors and, therefore, didn't overcome the problems associated with biomonitoring using macroinvertebrates.

Despite the findings of Young (2007), other functional indicators may be suitable to apply within the freshwater-saltwater transition and it is worth considering which indicators may be appropriate to the anticipated effects of the discharge and testing the appropriateness of these indicators in this environment.

5.3. Microbial Source Tracking

In the case of a waste water discharge where the primary concern is microbial contamination, Microbial Source Tracking (MST) is a tool that can be used to distinguish different sources of contamination. MST uses molecular techniques to identify the origin(s) of microbial contamination and can distinguish between farm, domestic or feral animal, human or bird sources.

In the case of the Porangahau River, the WWTP discharge entering the lower river is likely to be the predominant source of human waste to the river, while microbial contamination from the upper and middle parts of the catchment are likely to be primarily from diffuse pollution (especially farming). Therefore, using MST, it may be possible to distinguish between the relative contribution of the WWTP discharge to microbial contamination of the estuary of the Porangahau River.

6. CONCLUSIONS

Biomonitoring within the transition from freshwater to estuarine environments is difficult using existing methods due to the impact of the discharge being confounded by the effects of changes in physicochemical or habitat variables. Ideally, comparison of biomonitoring results (*e.g.* MCI scores) at the impacted site with a reference site located a short distance upstream of the discharge would minimise the effect of environmental changes. However, tidal flows may result in the effects of the discharge extending in both an 'upstream' (during flood tides) and downstream from the discharge point, meaning that the reference site must be located further upstream than desirable. There is nothing that can be done after the fact to overcome a confounded study design resulting from a poorly selected reference site.

The MCI was developed for application in flowing freshwater environments and its performance has not been verified in environments influenced by saltwater inputs. Therefore, it is recommended that the MCI family of indices should not be applied to areas affected by salt water. Longitudinal physicochemical and faunal surveys should allow identification of the start of the freshwater-estuarine transition.

Functional indicators may provide an opportunity to assess ecosystem health within the challenging environment represented by such transition zones.

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