A review of the ecological health and water quality in four Southland estuaries

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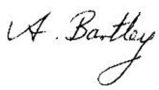	Formatting checked by:	Alison Bartley
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Executive summary

The National Institute of Water and Atmospheric Research was approached to review existing ecological monitoring information available for Southland estuaries in order to assess monitoring methodologies and how monitoring results are being used to assess the health status of the region's estuaries. The report includes:

- A relatively detailed review of the methods, analyses and interpretations of data that have been collected in Southland estuaries over a number of years.
- An evaluation of the relative status of the four most comprehensively monitored estuaries (Waikawa, Fortrose, Jacobs River, and New River).

Monitoring conducted in the estuaries focusses on fine and broad-scale data collection. Fine-scale sampling includes sediment properties such as sediment accumulation rate, redox discontinuity potential depth (RDP), grain size and organic matter content, nutrient and heavy metal contaminant concentrations, and macro-invertebrate abundance and richness. Broad-scale sampling includes the areal coverage of habitat-defining features such as seagrass, nuisance macroalgae, fringing terrestrial vegetation, fine mud, very fine mud, and 'grossly eutrophic' areas. From these data, a number of indices and 'health condition bands' have been developed.

The fine-scale variables that have been assessed in Southland estuaries are appropriate, and are comparable to those that are routinely measured in estuaries throughout New Zealand for state-of-the-environment reporting. However, interpreting temporal trends and evaluating the significance of differences between sites was often difficult due to a poor characterisation of data variability. This compromises the appropriate allocation of sites into particular environmental bands (poor, moderate, healthy). Other general methodological problems that we encountered included:

- Variation in the total area used in broad scale habitat mapping, which compromised spatial summary statistics.
- Inconsistent and problematic methodologies used for assessing sedimentation accumulation rates.
- Lack of validation of several indices used (many simplistically adapted from a single overseas AMBI formulation) to indicate estuarine health.
- Lack of validation of the values used to define the 'bands' of estuarine health conditions.

Although we have concerns with various methods, analyses, and interpretations of the data—and the lack of underpinning information to support indices and bandings—the authors of the Southland Estuaries monitoring reports have collected a variety of useful information over a number of years from the four estuaries we examined. Importantly, by and large, the authors have correctly assessed the key issues affecting the four estuaries, and have been able to identify the stark contrast between healthy and unhealthy sections of the estuaries. Furthermore, although strong statistical trends may be absent for individual variables, the weight of evidence from multiple measured parameters indicates that the individual and combined effects of nutrients and sediments are negatively impacting the four estuaries to varying degrees.

The combined approach of fine scale monitoring coupled with broader-scale tracking of habitat change (e.g., increases in mud and macroalgal coverage) is certainly appropriate, and we are not recommending the suspension of sampling or the relocation of sites in any of the estuaries. Instead

we offer ways in which the methods, analysis and interpretation could be improved. We want to see more effort put into standardising, ground-truthing, and verifying the accuracy and replicability of the data being collected. The indices and bandings also appear to require more scientific reviewing. At present, we are concerned that some of the data and reports on estuarine status and trends are not rigorous enough to withstand the scrutiny of external scientific experts, which could affect environmental management discussions and policy outcomes.

1 Introduction

Environment Southland and DairyNZ have a common interest in maintaining the ecological health of their freshwater and estuarine systems. In October 2014, the National Institute of Water and Atmospheric Research was approached to review existing ecological monitoring information available for Southland Estuaries in order to assess the suitability of the monitoring methodologies and how they are being used to assess the health status of the region's estuaries. The review focused on New River Estuary, Jacobs River Estuary, Fortrose Estuary and Waikawa Estuary (Figure 1). These four harbours were selected because of the comprehensive monitoring information available at these sites and because they represented a range of conditions in terms of size (relatively small to relatively large) and environmental condition (poor to relatively healthy). The main objectives of this study were to report on:

1. The main drivers of change in Southland estuaries and the risks and uncertainties regarding current status and temporal trends.
2. The current status and trends of water quality and ecological health in the four focal Southland Estuaries.

By addressing these two objectives, we aimed to highlight:

3. The critical science gaps that need to be addressed to better identify thresholds relevant to the management of Southland estuaries.

The first objective, assessing the main drivers of change in Southland estuaries, is addressed in a companion report by Green (2015).

To achieve objective 2.), our work is divided into a series of steps: Firstly we outline and review the rigour of methods and data analysis used to date (Sections 2.1 to 2.4). We focus on the 'bandings' used to designate ecological condition and their justifications (Section 2.2). We also examine a number of indices that have been used to summarise environmental variables (Section 2.3). Section 2.4 specifically addresses the measurement of water quality.

In Section 3, we discuss our observations from a trip to Southland in February 2015. We spent a day at each of the four focal estuaries making observations at variety of sites in each, including at all of the long-tern ecological monitoring sites. Section 3 includes comments on the suitability, representativeness, and number of existing monitoring sites in each estuary.

In Sections 4.1-4.4, we provide our assessment of the current status of water quality and ecological health in each estuary based on all available information (including our own observations) by examining physical, chemical and biological variables and the reported changes in habitat coverage over time. We compare and contrast our impressions with previous ecological assessments of Southland Estuaries in Sections 4.5-4.6.

Section 5 provides concluding remarks, including comments on uncertainties and science gaps, and recommendations for improved data analysis, collection and management.

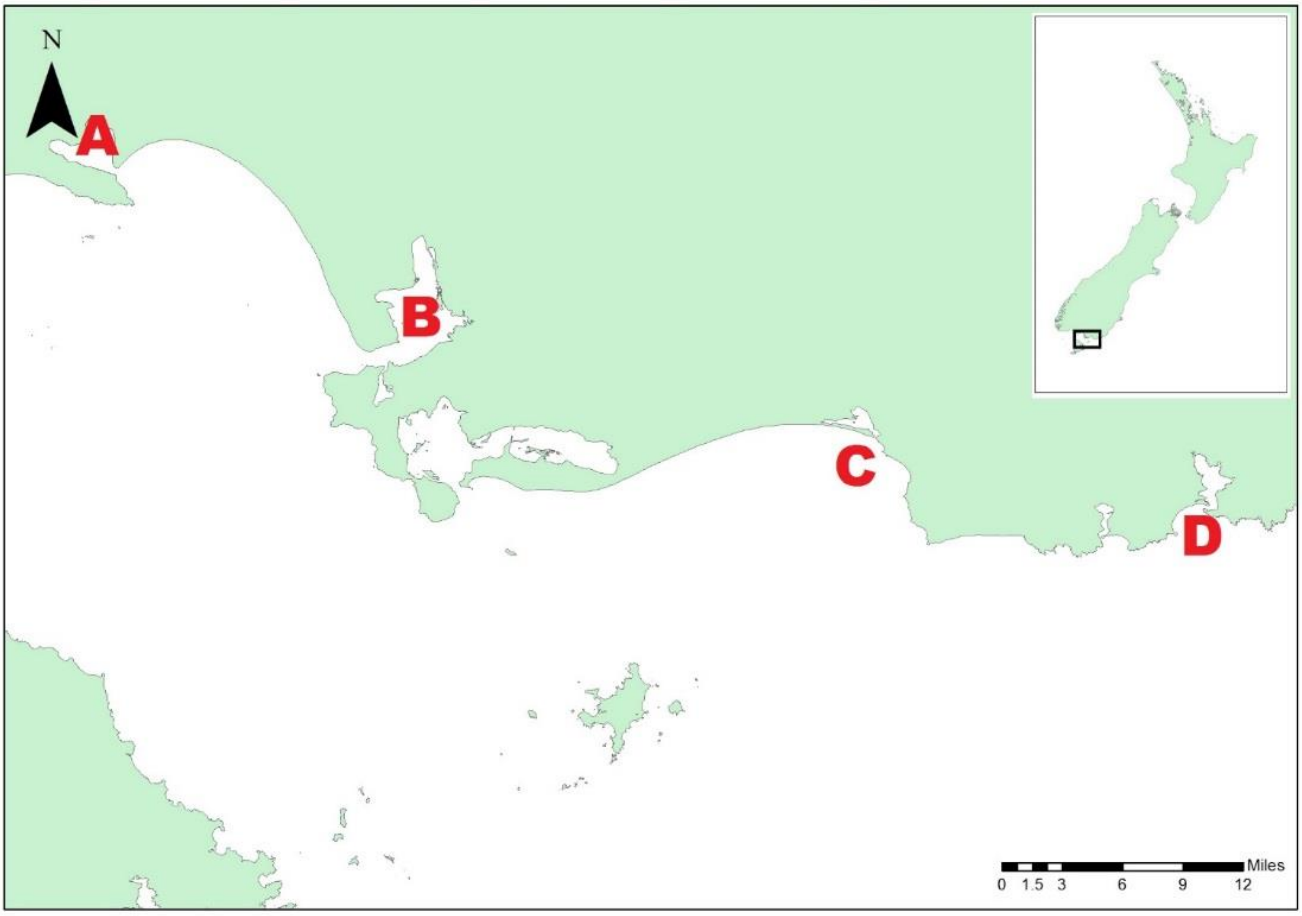

Figure 1: Location of the 4 Southland estuaries. A.) Jacobs River Estuary, B.) New River Estuary, C., Fortrose Estuary, D.) Waikawa Estuary.

2 Assessment of the Methods

2.1 Methods

Estuaries in the Southland region have been assessed since 2001 using broad and fine scale monitoring. This largely follows the Estuarine Environmental Assessment and Monitoring Protocol developed by Cawthron (NEMP, Robertson et al. 2002). There are a number of other smaller initiatives including historic sedimentation and sedimentation rates, macroalgal and contaminant assessment.

Fine Scale Monitoring

Fine scale monitoring focuses on a number of 60m x 30m lower intertidal sites within each estuary. Sites are monitored for a range of physical, chemical and biological indicators based on the methods described in the NEMP (Robertson et al. 2002). Samples are collected and analysed for the following variables:

- Water column salinity.
- Redox Discontinuity Potential (RDP) depth, a metric of the depth of oxygen penetration into the sediment.
- Sediment grain size (% mud, sand, gravel).
- Sediment organic matter content: Total organic carbon (TOC).
- Sediment nutrient concentrations: Total nitrogen (TN), Total phosphorus (TP).
- Sediment heavy metal concentrations: Cadmium, Chromium, Copper, Lead, Nickel and Zinc.
- Sediment macro-invertebrate abundance and diversity (infauna and epifauna).

Broad Scale Monitoring

Broad scale monitoring follows the NEMP approach (Robertson et al. 2002) and focuses at the habitat scale. Broad scale monitoring assesses the habitats present in each estuary, the percent area that each habitat class covers, and changes in the coverages over time. Assessments are made using aerial photography and ground-truthing by site visits. These are converted into health scores based on categories or 'bands' based on aerial extent and percentage change in extent, of variables including:

- Soft mud.
- Macroalgae.
- Seagrass.
- Saltmarsh.
- 'Grossly eutrophic' habitat.
- Fringing terrestrial vegetation (buffer zone).

Sediment monitoring

The broad-scale sediment monitoring that is undertaken in Southland area estuaries examines sedimentation in two ways:

- Historic rates of sedimentation have been assessed using radio-isotope analyses of sediment collected in long cores.
- Current sediment accumulation rates (SAR) are assessed by measuring the depth of sediment on top of buried concrete plates at regular intervals over time. There are four replicate SAR plates at up to four sites per estuary (some of which are in close proximity to fine-scale monitoring sites).

Macroalgal monitoring

Macroalgae coverage in estuaries can indicate the presence of elevated inorganic nutrient concentrations. This variable is occasionally assessed as a subset of the broad scale monitoring, with a specific focus on the density and coverage of macroalgae and the changes in these over time.

2.1.1 Critique of current methods

Most variables included in ES monitoring are commonly measured for the benthic marine environment and have well tested methodologies. For example, the methods used for the collection of macro-invertebrates (core size and level of replication) have been used in New Zealand and internationally for a considerable time (Thrush 1988). The 'Methods' sections in the reports prepared for Environment Southland (e.g. Robertson and Stevens 2007, Robertson and Stevens 2010), typically focus on the banding rather than on the full collection and analytical techniques *per se*, which are instead referenced to the NEMP (Robertson et al. 2002). There are still technical details that are absent that would be useful for method evaluation, for example, details of the processing/drying of heavy metal samples, and how salinity and grain-size are measured. A previous review of the methods used by Robertson and colleagues (Jenkins 2013) identified some of the following problems:

- Variation in the total area used in broad scale habitat mapping and the consequence this has for the interpretation of spatial summary statistics.
- Inconsistent and flawed methodology used for sedimentation measurements.
- Use of the term Redox Discontinuity Potential (RDP), when in fact aRDP was measured. RDP depth is based on an electrical potential difference and thus requires an electronic sediment probe. The method of visual assessment that was used should be referred to as the Apparent Redox Discontinuity Potential depth, or aRDP (see Gerwing et al. 2013 and references therein).

We broadly agree with the assessments made by Jenkins (2013), and we will comment further on the first two points above. The advantage that we had over Jenkins (2013) in conducting our review was that we were able to visit all of the Environment Southland sampling sites in four estuaries in order to see how/where the NEMP methods were being applied.

<u>Variation in total area:</u> Jenkins (2013) identified change in total area surveyed as a point of concern and the need for standardisation. We repeat this concern, but also demonstrate why this is an issue for the interpretation of data. In 2001, the total benthic habitat area of New River Estuary was listed

as 2,617 ha, in March 2007 as 2695.7 ha, and in January 2012 as 2952 ha (Robertson and Stevens 2007, Stevens and Robertson 2012). There was a 13%, or 335 ha, difference in the area surveyed between 2001 and 2012, and there was a 10%, 256 ha, difference in the amount of area surveyed between 2007 and 2012. The authors incorrectly used the 2001 baseline data as the comparison point for their interpretation (Stevens and Robertson 2012). They concluded that the area of soft mud increased by 4% from 2001 to 2007 and by 18% from 2001 to 2012[1], and used this statistic to support their assertion of increased estuarine muddiness. However, because the amount of area sampled increased between survey periods (rather than being standardised), the area of almost all habitat types increased concomitantly. For example, using the same Stevens and Robertson (2012) formulae, it can also be calculated that, from 2001, there had been an 8.1% increase in sand and a 147% increase in 'other'. With standardisation, there is a more modest increase in mud of 1.8% (moving from 20.9% to 22.7% between 2001 and 2012), but this still has limitations. Difference in the total area of the harbour between years is due to variance in the degree of tidal exposure captured in aerial photos. Therefore the apparent 'increase' in total area in more recent assessments compared with 2001, is dominated by habitats located lower on the shore, i.e., towards the MLWS level. Habitats located on the lower shore are more likely to be dominated by muds or sands, and are less likely to contain features such as saltmarshes, rock fields, and artificial structures (with respect to their percentage coverage as a proportion of the total area). The New River Estuary example is the most extreme of the four estuaries we examined (Waikawa Estuary had a 5% variation in area between 2004 and 2008/09, Jacobs Estuary 1% between 2003 and 2013 and Fortrose Estuary <1% between 2003 and 2013), however, there is still a need to make changes in the future and retrospectively re-evaluate standardised data/statistics:

- Consider the implications for other broad scale surveys in other Southland estuaries.
- Re-evaluate data using summary statistics that are less sensitive to total area. Include caveats relating to the implication of variance in total area in the evaluation of change in habitats.
- Re-evaluate changes in habitat coverage using a standardised area. GIS techniques are available for cropping, to ensure a consistent 'frame-size' between years. The downside of this is that it will restrict comparison to the year with the smallest area covered. Another possibility is that each estuary could be separated into two areas (an upper intertidal zone and a lower intertidal zone) that are both standardised and analysed separately.
- OR alternatively, if consistent larger areas can be recorded moving forward then establish a new baseline and ignore earlier, smaller area data. The disadvantage of this is that reduces the time series.

<u>Issues with sedimentation measurement:</u> Sedimentation is measured from the annual accumulation of sediment on top of buried concrete plates. Plates are buried to a depth where they are stable in the sediment strata and covered (usually around 30cm). Sedimentation (or erosion) over time is measured as the difference in sediment height above the plate from a baseline measurement. Two arrangements of plates were observed during the sites visits in February 2015. One setup used 4 plates located in the corners of a square approximately 30-50m apart. Plates were located at the centre point between two stakes approximately 2 meters apart e.g. Bushy Point in New River

[1] Note that the authors also made errors in these calculation as this should be a 3.8% and 22%. In 2012, they looked at the change in soft mud from 2001 (548 ha) and 2012 (669 ha) which was 121 ha, but this was calculated with as a percentage of 669 as the total rather than the 548. Percentage <u>change</u> should use the earliest measurement as the total.

Estuary, Upper North in Waikawa Estuary. In this setup each plate was close to (~1m) the marker stakes. A second setup used 4 plates located in a straight 20m transect marked out with 5 stakes e.g. Fortrose Estuary. Plates were located at the centre point between adjacent stakes which were located approximately 5 meters apart. In this setup, each plate was relatively far away from its nearest stake (~2.5m). While both setups measure the same variable (plate depth) having methodical variation is not ideal and should be standardised unless there is specific justifications for variation.

During the site visit to Bushy Point in New River Estuary, a severe flaw was noted. At this site the two stakes marking the plate showed high degrees of scour (Figure 2). This was caused by aggregations of macroalgae wrapping around each stake and abrading the sediment surface under inflow and outflow of tides. At the mid-point between the stakes, where the plate was located, there was a notable depression. This poses a significant issue for the reliability of sedimentation data. It is likely that this is of greatest concern at sites that have higher current/wave conditions (that may stimulate erosion/scour processes) and where stakes are close together. For example, this may be less problematic in upper reaches of Waikawa Estuary where conditions are calm (although the Lower South site had macroalgae around the stakes, see photos in Stevens and Robertson 2009). There is a need for further considerations:

- Data from Bushy Point in New River Estuary is unreliable and should be removed from consideration.
- As multiple sedimentation methods are used and issues of reliability may vary depending on technique, site, and from year to year, it is difficult to discern which historic sedimentation measures are valid and which are flawed. A fuller evaluation of historic data should be undertaken. Reports do not indicate which version of the technique is used at which sites.
- Greater distances between marker stakes (i.e. 5m), will help to minimise the effects of external disruptions. Using a single reliable method at all sediment sites for comparisons between sites/estuaries is a desired outcome.
- A photo record of each site/plate at the time of sampling would be a useful in the evaluation of reliability with respect to external influences.

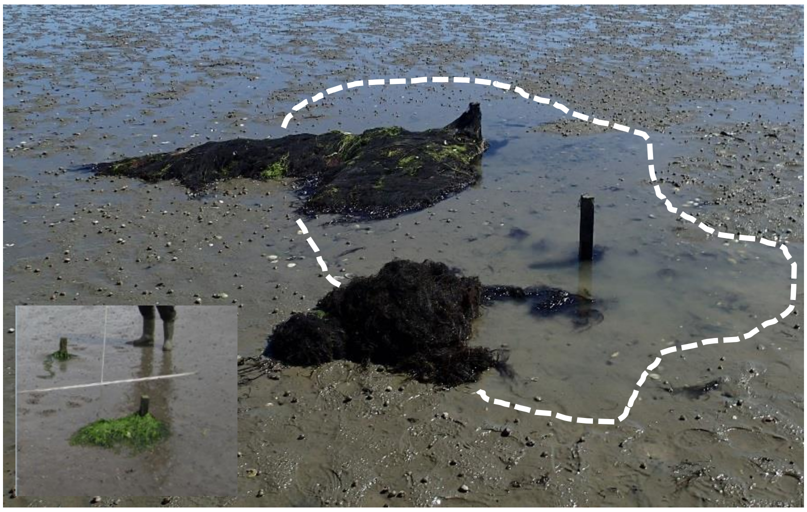

Figure 2: Sedimentation Plate at Bushy Point, New River Estuary. White dotted line shows the areas in which scour and depression was visible. Inset photo from Waikawa Estuary with macroalgal accrual on sediment plate stakes.

2.2 Variance, bandings, indices and their interaction

2.2.1 Variance, precision and accuracy

Variance

A primary reason for making repeated 'replicate' observations of environmental characteristics or populations is to understand variability. Although averages are often reported to represent environmental or population characteristics in particular locations, times, or categories, it is the variance that allows us to assess the significance of differences between them. There are many components to variance, and means and variance should always be interpreted within context (for example, a mean value for 'estuary A' assumes that the samples collected were broadly representative of that estuary as a whole), though we will not conduct a review of statistical theory here. Nevertheless, one of the major weaknesses of the current estuarine assessments occurring in Southland area estuaries is a poor characterisation of variance, which means that fit of data into particular environmental bands (poor, moderate, healthy, very healthy) is subjective and difficult to evaluate and interpret. For example, does a result fall clearly within a 'good' band, or does it overlap with one or more neighbouring bands (e.g., ranging from 'fair' to 'very good'). In the reports that we reviewed, mean and ranges are provided for some variables including total organic carbon, total nitrogen, total phosphorous and heavy metals[2]. In other cases, variance was not able to be measured, for example, for the biotic indices. However, variance should have been reported for aRDPL, sedimentation depth and grain-size. For aRDPL the method states that 'within each of 10 plots one random core is collected and the average RPD depth recorded'. From this there should be at least 10 measurements, yet the appendices report the raw data as the average of cores 1-4, 5-8 and 9-10 (e.g. Robertson and Stevens 2013A) and the results present the average in a bar-graph without variance. Similarly, multiple measurements are made for sedimentation depths yet only averages are reported (in the form of net accumulation). Without gauging the level of variation it is difficult to determine whether differences between mean values and the trends over time are significant.

[2] Jenkins (2013) commented that there was a "*lack of error estimates (variance and standard error) associated with the mean values for many of the indicators presented*". This statement a little harsh, given that ranges are provided for most chemical and physical variables.

Precision

Issues associated with the precision of macrofaunal indices are dealt with in section 2.2.3. Issues associated with the imprecision of broad scale habitat mapping have been dealt with in section 2.1.1.

Accuracy

Accuracy of broad-scale mapping techniques is not fully addressed in either monitoring reports or the EMP (Robertson et al. 2002; Part A page 36). Specifically, details on the use, repeatability and accuracy of ground-truthing information is absent; this was noted by Jenkins (2013).

The methods in the Roberson et al. reports focus exclusively on the digital mapping processes of vegetation and substrate features[3]. While this is appropriate for certain habitat types (e.g. seagrass, saltmarshes), for others, it is impossible to accurately map solely from digital images (e.g. soft mud, very soft mud) and there must be a reliance on field collected information. There is no explanation as to how field information is used in the mapping process. For example, the coverage of field personnel within an estuary to show the extent of ground covered (e.g., see Figure 3, which shows our coverage in Jacobs River Estuary February 2015). Given the difficulty of walking in deep mud, the size of some of the intertidal areas, and the length of time that intertidal habitats are exposed, we have some doubt as to whether all of the soft-mud / very soft-mud habitat boundaries were walked and ground-truthed. This becomes important if changes in the areal extent of these habitat categories are being used to rank the health of Southland estuaries.

There is no assessment of how accurately observers can demarcate different habitat types or the variability between observers. This can be termed “repeatability”. That is, if one observer marks out a boundary of a specific habitat, what is the average deviation of another observer attempting to mark out the same patch? Do we interpret habitat boundaries in the same way, or does our judgment change from year to year? For example ‘soft mud’ and ‘very soft mud’ are distinguished by sinking depth (the former “When you’ll sink 2-5 cm” and the latter “when walking you’ll sink >5cm”; Steven and Robertson (2009)). The depth to which one sinks into the sediment is influenced by many variables, for example, a person’s weight and shoe-size, the water content of the sediment (which can vary depending on when after emersion the site was visited) and biota and shell hash present. We would expect some differences between observers and this affects our confidence in the accuracy of the position of a habitat boundary. Useful examples for appraising habitat boundaries are shown in Needham et al. (2013) (Figure 3). Also relating to accuracy, commentary on the spatial resolution of habitat maps is needed. We noted a high degree of variation in coverage of macroalage in the estuaries, even within generally dense areas (e.g. Figure 4).

[3] “Vegetation and substrate features were then digitally mapped on-screen from the rectified photos using the Arcview ‘image analysis’ extension. This procedure required using the mouse to draw as precisely as possible around the features identified from the field surveys on the computer screen and saving each drawing to a shape file or GIS layer associated with each specific vegetation or substrate feature”. Robertson et al. (2002).

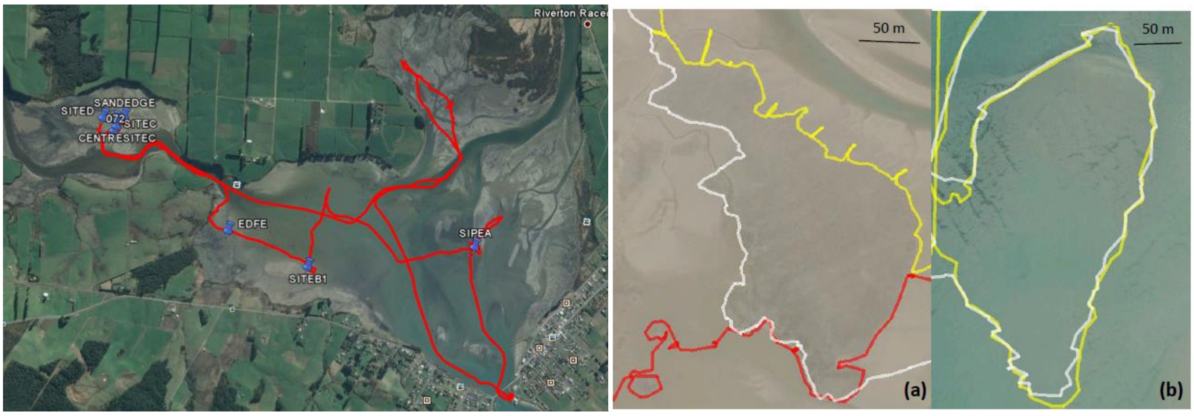

Figure 3: **GPS track marks.** The route taken through Jacobs River estuary (left) and an example of repeated tracks around habitat boundaries by different researchers from Needham et al. (2013).

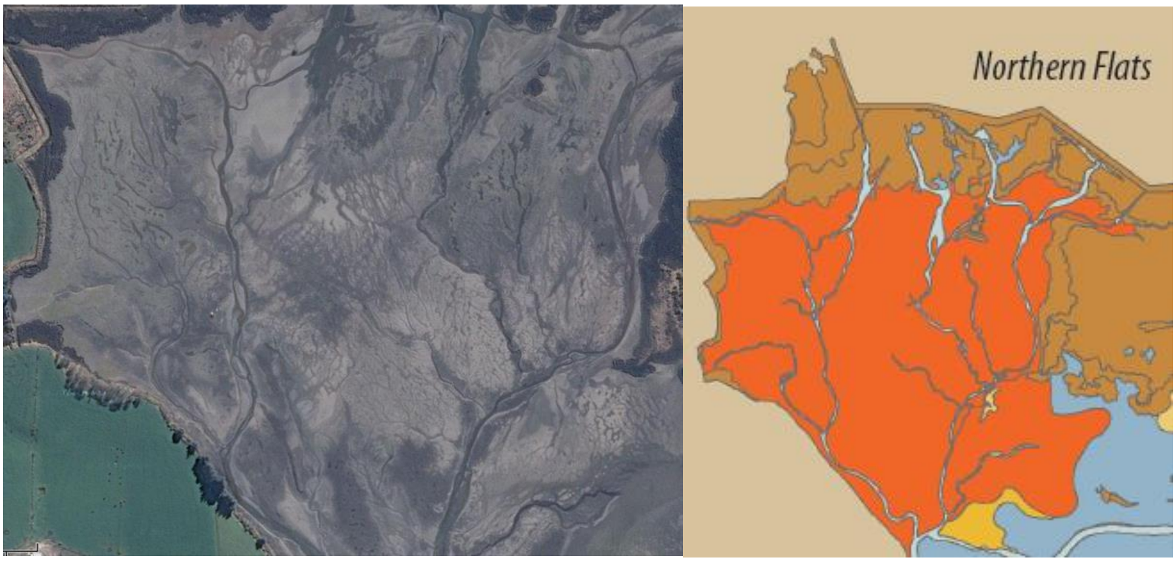

Figure 4: **Macroalgal coverage in the nothern flats of Jacobs River Estuary.** The photo (left) shows a high variation at a small spatial scale, compared with habitat classification (right).

2.2.2 Condition rating - bandings

In the Robertson et al. reports on Southland estuaries, a series of "condition ratings" are used to categorise various types of data into bands (e.g., 'Good', 'Fair', 'Poor'). Condition ratings are stated as being "*based on a review of estuary monitoring data, guideline criteria, expert opinion, and are regularly reviewed*" with the Robertson and Stevens (2006) report cited. However Robertson and Stevens (2006) also state that "*the guidelines proposed require further development and refinement but are intended as the first phase…at present many of the specific criteria for rating estuaries using the specific indicators (e.g., very good, good, fair and poor) are preliminary or as yet undeveloped*". Supporting information is not provided to indicate how each of the bandings has been further developed or tested. For example, Robertson and Stevens (2006) state in section 8.3 that for TOC or nutrients that "there is no New Zealand rating criteria in estuary sediments" and that "in the future a multi-indicator estuary condition index be developed for TOC, TP and TN, but in the interim use an index based on the US rating". This uncertainty is not mentioned in the reports or included in the interpretations. Furthermore, the values in the TOC bandings change in 2013 "based on newly available data" (Robertson and Stevens 2013A), but the specifics of the data or the mechanism for adjustment is not provided. This need and mechanism of adjustment may have been valid and correct, but the necessary information to make that judgement is not provided. The lack of justification leads to uncertainty over the validity of bandings:

- For each variable, is expert opinion used on its own or in combination with data to create the banding?
- Which experts have been consulted?
- Which monitoring data have been used to advance condition ratings and what is the process for their development.
- To what range of data are the bandings applicable (from what range of data have they been developed)?
- What is the time-scale and the process for the 'regular reviewing' of bandings?

It is likely that most or all marine ecologists would agree with the general directions of the bandings in relation to the measured variables i.e. very good>good>fair>poor. However, the categorical groupings of continuous environment variables and the specific cut-off values that separate bands (e.g., that separate 'fair' from 'poor' or 'very good' from 'good'), are not straightforward and require evidence of detailed consideration. Furthermore, these bandings do not just separate the range of a measured variables (i.e. groups 1-5 or high to low), they specifically adjudicate on condition, which necessitates higher confidence. In many instances the placement of cut-off values can be considered arbitrary. For the majority of variables listed in reports, the foundation of the condition ratings is not provided and so it is difficult to evaluate their rigour. Taking a cautious approach, it is necessary to assume that these condition ratings are fallible until information is provided proving their reliability. An exception to this is the condition ratings for heavy metals, which have cut-off values based on the interim sediment quality guideline trigger values (ISQG) from the Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZECC 2000). In this instance there has been a development process in cited literature that increases confidence (ANZECC 2000). However, it is important to note that changes in macrofaunal communities have been detected at values well below ANZECC guidelines in the Auckland Region (see Hewitt et al. 2009 for details). Moreover, Auckland Council have developed their own Environmental Response Criteria (ERC) for estuarine environments (ARC 2004) with substantially lower metal concentrations. Their ERC uses a traffic-light coding with 'green' threshold similar to the concentrations of the Threshold Effects Level (TEL) from MacDonald et al. (1996) and the 'red' threshold close to Effects Range Low (ERL) of Long and Morgan (1990). The further bandings of metals in the Wriggle reports seems unnecessary, when data can simply be reported in reference to the standard values mentioned above.

There are further examples where bandings conceal important details from which the status and change in the ecological system should be evaluated. For example, the broad scale 'seagrass area changing rate' has bandings based on changes from the baseline conditions, ranging from 'small decrease' of <5% to very large decrease at >50%. We suggest that the total magnitude of change (number of hectares lost) should be presented and analysed as well. To illustrate this we consider Awarua Bay, which has 600 ha of seagrass. Based on the index bandings, the loss of 90 hectares

(900,000 m^2) of seagrass would still be considered a moderate loss in Awarua Bay. Alternatively, an estuary with only 4 hectares could lose half a hectare and this be considered large and require annual monitoring. The loss of half a hectare could be important or trivial depending on other factors. Analysing data in purely a categorical way misses many of the important complexities and decrease clarity. In summary:

- The type of information, or lack thereof, used to develop condition ratings should be reported for each individual variable in the methods of the report.
- Confidence in the condition rating bandings is related to a demonstration of their validity, i.e., studies that specifically test them. Primary literature demonstrating statistical testing or evidence of validity should be cited.
- The reliability and veracity of the bandings is a key consideration when evaluating the data and the conclusions made from the results (i.e., how confident are we that 'poor' is 'poor'?).

2.2.3 Indices

When used correctly, marine indices can be useful for condensing multiple strands of data and producing univariate summaries that can be interpreted meaningfully and with high confidence. Confidence in an index stems from a demonstration of its validity; it should have a sound and transparent conceptual basis, a demonstrated ability to change over a gradient of interest (responding in a suitable manner), and data on precision or repeatability. If used erroneously, indices can cloud judgement and impede interpretation (Diaz et al. 2004).

For Southland estuaries, many of the measured variables have been used in adaptations of the AZTI Marine Biotic Index (AMBI) (Borja 2000). AZTI's Marine Biotic Index (AMBI) developed by Borja et al. (2000) is based upon the proportion of species assigned to one of five levels of sensitivity to increasing levels of disturbance, from very sensitive to opportunistic species. This index has been used in Europe primarily, but has also been applied in Asia, northern Africa and South America (Borja et al. 2008), although its suitability in New Zealand estuaries has not been fully tested. AMBI was designed to assess effects of organic over-enrichment, but it has subsequently been used to account for the effects of different types of stressors (e.g., Borja et al. 2003; Muxika et al. 2005). For Southland estuaries, Wriggle Coastal Management Ltd usages include:

- Benthic Community Organic Enrichment Rating - based on the AMBI formula adapted for New Zealand taxa.
- Benthic Community Mud Tolerance Rating - based on the AMBI formula using ecological groups developed for New Zealand taxa responses to mud.
- Wriggle Estuary Benthic Index (mud and organic enrichment) - based on the AMBI formula using ecological groups developed for New Zealand taxa responses to both mud AND organic enrichment.
- Macroalgae Co-efficient Condition rating - based on the AMBI formula, using macroalgal percentage coverage instead of ecological groups.
- Seagrass condition rating - based on the AMBI formula using seagrass percentage coverage instead of ecological groups.

Evidence has not been provided or cited that explores the validity of the indices used on Environment Southland estuarine data. Van Houte Howes and Lohrer (2010) and Rodil et al. (2013) found that the AMBI and other overseas indices did not track estuarine health conditions particularly well around Auckland, using macro-invertebrate data from 84 intertidal sites in the region. However, it should be noted that organic enrichment is not a predominant stressor in this part of New Zealand. It has been used somewhat more successfully in New Zealand aquaculture impact assessments (e.g., Keeley et al. 2015) when testing for intensive localised organic enrichment effects.

An example of the fallibility of indices is demonstrated with the macro-invertebrate organic enrichment index for the Jacobs River Estuary eutrophic sites (Figure 13 in Robertson and Stevens 2012, shown in Figure 5). The index failed to demonstrate the known eutrophic status of Sites D and E, showing them as similar to the fine scale monitoring sites that have a low AMBI coefficient scores. This was explained by the sediment mostly being devoid of life and the species used in the index being associated with overlying water and macroalgae (mostly mobile crustacea). Being able to predict the conditions under which an index does not work well is key; if failures are unpredictable, the index will have limited utility.

A second example of index fallibility is the macroalgae co-efficient (MC) for New River Estuary, which suggested “Good” conditions, despite New River Estuary having the most extensive eutrophic areas and some of the largest patches of *Gracilaria* and *Ulva* of all the estuaries. The reasoning for the favourable rating was due to the large unvegetated areas in New River Estuary, which down-weighted the co-efficient score. This again illustrates that both magnitude and proportion are important when assessing coverage, and that single index scores can be misleading or unhelpful. Subsequent adjustments to the MC index calculation have been made, but the details have not been provided[4]. The review by Jenkins (2013) wondered why indices for seagrass and macroalgae were necessary at all, when macroalgae coverage data (without arbitrary weightings) would suffice perfectly well.

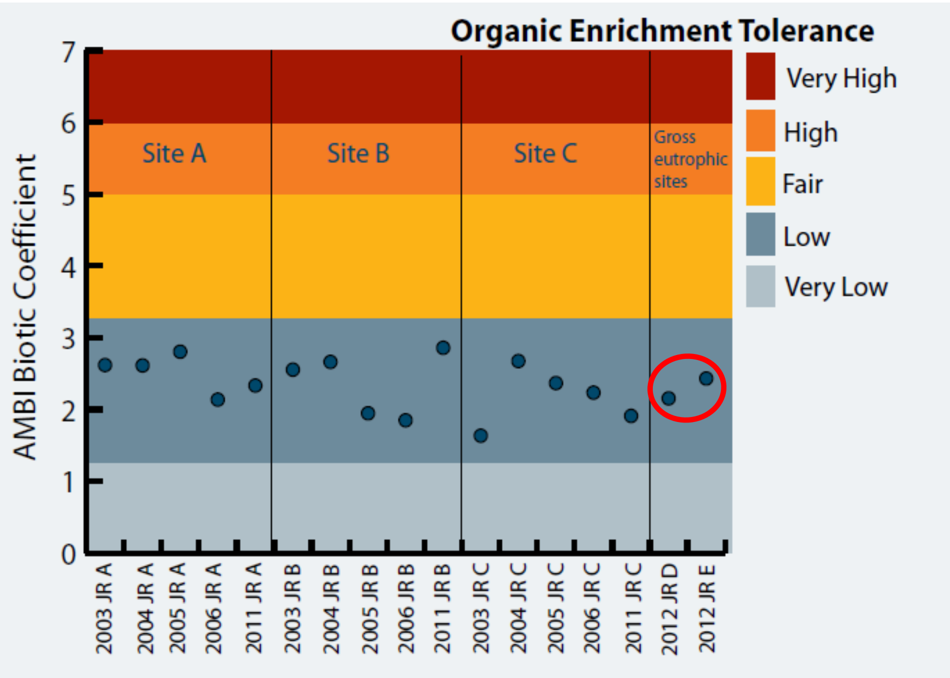

Figure 5: Organic enrichment macroinvertebrate rating for Jacobs River Estuary, 2003-2012. Red circle marks the eutrophic sites for which the index failed to show their enrichment status.

[4] Reports state that the condition rating was “revised in 2011 following a review of data compiled for Southland since 2007”, no further information is provided, including on whether past data were recalculated with the new methodologies.

The process used to allocate species to sensitivity groups (1-5) for increasing levels of disturbance for the AMBI index is unclear; as are the reasons or justification for changes to the allocations that have occurred over time. For example in New River Estuary in 2010, *Potamopyrgus estuarinus* was listed by Robertson and Stevenson (2010) as being NA for AMBI grouping in Appendix 3, but listed as being group III in Appendix 2. Thus the role of this species in the index is unclear. Furthermore, this AMBI score has changed over time, as *Potamopyrgus estuarinus* was listed as group II in 2009 (e.g. Fortrose Estuary, Robertson and Stevens 2009). No justification or explanation is provided for this adjustment. Also the implications of excluding taxa during calculations are also not considered. For example, in the mud tolerance biotic coefficient in 2013 one fifth of the species found at Waikawa Site A (4/21 species) and Site B (5/23 species) were not assigned to a mud group (1-5)[5]. As AMBI-based indices look at the proportion of taxa present in certain groups, the absence of ~20% of the species in the calculation is an important omission. For example although the MTBC score is 1.8 for Waikawa A in 2013, this score could vary between 1.43 and 2.63 if the unallocated species were assigned to either the SS or MM groups respectively (indicating the best and worst case scenarios for the unknown species).

A new development has been the implementation of the Wriggle Estuary Benthic Index (WEBI), which is the AMBI index with a new procedure for grouping of species based on taxa sensitivities to both mud and organic enrichment[6]. Demonstration of the performance of this index, and how it complements (or simply duplicates) the AMBI, has not been presented for the science and management community to evaluate as of yet[7]. Information on the interactions between the two stressors is of particular importance here.

The AMBI was not developed for plant taxa and these were not included in the original index (Borja 2000). However this has been adapted for Environment Southland Estuaries to explore macroalgae and seagrass coverage. There is no exploration or justification for the application of this index on macrophytes. We are in agreement with Jenkins (2013) who stated that it was "not clear that this (AMBI) index can be arbitrarily transferred to other indicators without some form of validation". There is no justification provided for the percentage covers used in the index (% cover of <1%, % cover of 1-5%, % cover of 50-80% etc). There is no justification for the multipliers used in the index (0, 0.5, 1, 3, 4.5, 6, 7.5). The same score can be achieved in multiple ways (for example, an index score of 3.2 could be achieved by an estuary having 14.3% in all coverage groups, or it could be achieved by having 30% cover of 1-5% and 5-20% and 20% cover of 50-80% and >80%). There is no consideration of what this means for the health or the functioning of the system. We see these metrics as being less suitable and more confusing than simply reporting on the raw data; particularly when their validity is unproven.

The use and validity of the seagrass condition rating is uncertain. This index again uses an AMBI formulation but with higher scores indicating a better rating (the reverse of the macroalgae condition rating). This is because high seagrass coverage is seen to be beneficial. However, the major problem is that there is no definitive coverage that seagrass should have in an estuary and, under low stress conditions, its distribution will be influenced by sediments, hydrodynamics and a raft of other environmental variables. This means that seagrass can naturally be a small or large component of an estuary. To quote Robertson and Stevenson (2006): "*Currently there are no published numerical criteria for assessing the condition of seagrass habitat based on its percentage cover. The main reason is that different types of estuaries have grossly different capacities to support natural seagrass cover".* Therefore an index based on percentage coverage and a band rating based on "bigger =

[5] *Magelona dakini* is not included here as it is found over the wide variety of sedimentary condition. The unassigned taxa are Amphipod Sp.1 and Phoxocephalid Sp.1 which are listed as "uncertain" and Amphipod Sp.2 and *Halicarcinus whitei* which are listed as NA.
[6] We presume this replaces the benthic community mud tolerance rating and the benthic community organic enrichment rating.
[7] We understand that this is in progress

better" misses the mark for evaluating seagrass habitats. Also for a healthy, normal estuary we would expect a unimodal relationship for seagrass coverage with 0% and 100% both being undesirable. This index may function when seagrass is virtually absent, i.e., when stressors are so pervasive that seagrass coverage is minimal, but the use of an index in this context may again have limited value relative to reporting the raw data.

In summary, the information on the indices does not demonstrate that they are well explored for their validity and appropriateness:

- It is good scientific practice, and the responsibility of an investigator to demonstrate or cite the validity of metrics and banding before they are used. A suitable pathway may include an initial study that tests an index, followed by refinement over time, followed by publication of the information in peer-reviewed scientific papers, before eventual implementation in SOE reporting. This has been the case for the health indicators currently being used in Auckland and Waikato Regional Councils (Benthic Health Models for heavy metal contaminants and sediment mud content, Anderson et al. 2006, Hewitt and Ellis 2010, Hewitt et al. 2012, http://stateofauckland.aucklandcouncil.govt.nz/marine-report-card/upper-waitemata-harbour-reporting-area-2014/; Traits Based Index (TBI) that is sensitive to mud and metals, van Houte-Howes and Lohrer 2010, Lohrer and Rodil 2011, Rodil et al. 2013).
- Where categorisation of a metric into bands is being trialled, an investigator has the responsibility to clearly state where bandings are associated with expert judgment, or where ongoing testing is necessary.
- In agreement with Jenkins (2013), it is recommended that less reliance is placed on AMBI type indices that are not validated (particularly non macro-invertebrate usage).

2.3 Analysis

2.3.1 Trends

A key component of any environmental monitoring are the trends in data that indicate how variables, sites or estuaries are changing over time. Due to the low frequency of sampling in Southland Estuaries, there is insufficient data to be able to detect trends confidently in most instances without some background information on natural cycles. Trends detected over a period of 5 years (the minimum data set length used in Auckland Council time-series trend analyses) may actually be a part of longer term cycles associated with ENSO or other greater-than-annual phenomena. Moreover, intra-annual sampling (e.g., quarterly or bimonthly) can be invaluable for identifying trends, as seasonal variation can mask or illuminate temporal patterns.

With the data available, trends in Southland estuaries have not been quantitatively measured or formally tested and are judged from visual observations, that is, comparing a recent measurement with historic values and deciding whether they are commonly higher or lower (e.g., Robertson and Stevens 2013B). This visual qualitative assessment fails to consider natural variance and is subjective: for example, at Waikawa Site A, mud content is suggested to be decreasing, but the low value in 2006 casts doubt, and it is difficult to be confident with so few data points, other than to state that mud content has been low and variable (R^2 of 0.21, P=0.43 for Site A mud content). We disagree that there is strong evidence in this case. If this type of assessment is to be used then it can be made more objective by using quality control chart methods, e.g., Anderson and Thompson (2004).

A second example is the interpretation of trends in the number of species per core for Waikawa Sites A and B. In the Robertson and Stevens (2013B) report it was stated that "*the 2013 results indicate that the macro-invertebrate abundance and species richness had declined considerably since the baseline period 2005-2008*". Low numbers were recorded in 2013, but strong evidence of a trend is lacking given the low values in 2005 (Figure 6). To illustrate the difficulty, we show the effects of removing a single data point and how this could modify our interpretations (Figure 7 and Figure 8), changing the 'trend' between positive to negative. Southland Estuarine data may be particularly susceptible to this given the long gaps between data collection (i.e., it is the presence/absence of the influential 2013 data point that determines the trends). This demonstrates that more data is needed before trend status can be reliably interpreted. Moreover, as demonstrated, trends can be statistically detected with only 4 points. However, assessment of trends by regression analysis also involves examination of the residuals. In this case the residuals would demonstrate a strong temporal pattern, and the regression analysis would stop at that point (Fox 1991). Overall, qualitative assessment of trends is detrimental to the strength of the results and the following interpretation.

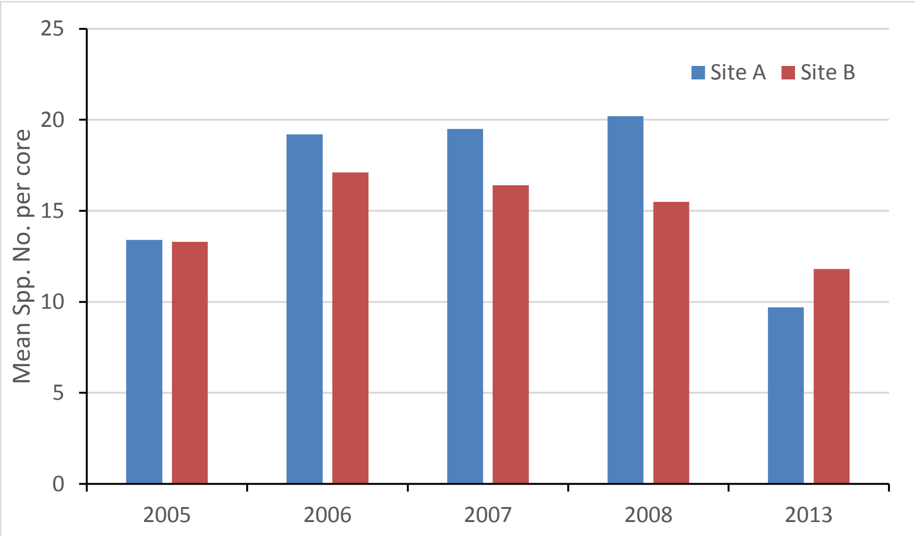

Figure 6: **The mean number of species per core over time at Waikawa, 2005-2013.**

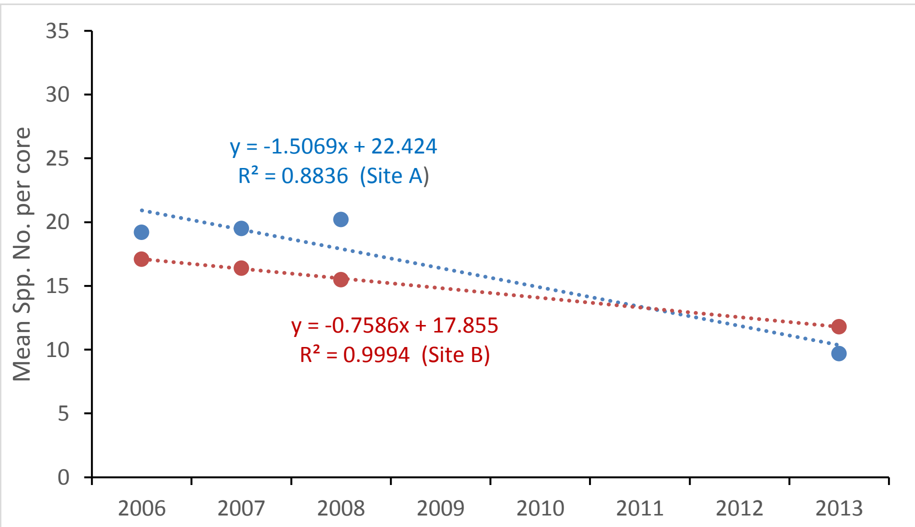

Figure 7: **The trend in species number removing the 2005 data point for Waikawa.**

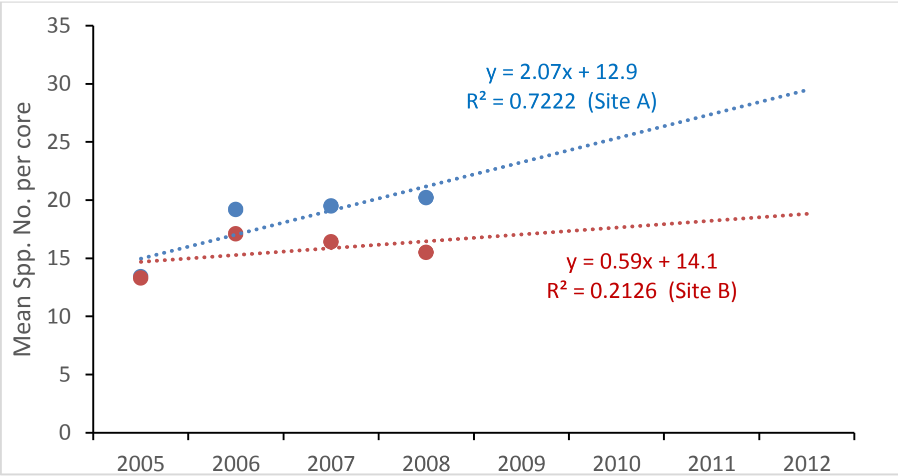

Figure 8: **The trend in species number removing the 2013 data point for Waikawa.**

2.3.2 Misinterpretation of, or lack of evidence for, trends

In reference to Waikawa in 2013, mention was made of *"declining populations of mud tolerant organisms (i.e., more sand tolerant organisms) over the five years of monitoring"*. This statement is not possible to verify from the information shown in the reports. In the assessment of invertebrate data, there is little awareness of cyclical behaviour which has been demonstrated in other locations for many of the taxa found in Southland estuaries (Townsend et al. 2012, Hailes et al. 2012, Greenfield et al. 2013, Halliday 2013, Parkes et al. 2014). Although it may not be possible to show with limited temporal data and the absence of intra-annual sampling, it is still important to consider that changes in species abundance do occur for multiple reasons, not solely due to mud-content. Too narrow a consideration limits the interpretation. For example, in 2013, numerous species abundances are cited as the demonstration of the macro invertebrate community showing strong sand preferences and being adapted to a sand dominated habitat[8]. *Macroclymenella stewartensis* is highlighted as the most abundant species at Waikawa Site A with its preference for sandy conditions, as are *Austrovenus stutchburyi* and *Boccardia syrtis*. However the analysis fails to consider that each of these species has shown large drops in abundance from 2008 to 2013 (*M. stewartensis* dropped from 281 to 133, *A. stutchburyi* dropped from 84 to 16 and *B. syrtis* dropped from 37 to 18). This does not infer that these species are now declining *per se*, but simply there is more going on than the interpretation has suggested. Patterns over time for individual species are not considered as part of the benthic invertebrate analysis, and several conclusions are not supported by the data presented.

2.3.3 Relationships

The relationship between physical/chemical variables and their effects on macroinvertebrates is not well established. 'Causality' can be difficult to verify, but showing close correlation between environmental factor/stressors and changes in the community is a useful starting point. Again using Waikawa as an example, the three step process (page 9, Robertson and Stevens 2013B) attempting to link sediment mud content to the invertebrate community does not do so. In the 1st step the biotic summaries are noted, the second step discusses mud tolerances but this is not relating the biota to physical measurements of the site, and the 3rd step presents a MDS plot with sediment data

[8] "The ratings also indicate that the macroinvertebrate community was dominated by organisms that are intolerant of enrichment rather than highly tolerant species e.g. *Macroclymenella stewartensis*"

superimposed, but there is no further linkage to demonstrate the influence of sediment on the grouping of sites (i.e. PRIMER BIO-ENV, BVSTEP or BEST).

In Summary:

- The trend analyses in the Southland estuary monitoring reports is generally qualitative and weak. The extent of data analysis is also limited, which leads to over interpretation or conclusions aren't necessarily supported by the data presented. Concluding remarks are often broad and speculative.
- Regression analysis (e.g., ordinary least squares regression) could be used to test for trends in sediment properties, macrofauna and contaminants. For most estuaries there are low number of data points, but statistical power will increase over time. Techniques are available to examine cyclical behaviour of macrofauna, or spatial auto-correlation if sampling frequency is increased to multiple times per year (Townsend et al. 2012, Greenfield et al. 2013, Parkes et al 2014), though there are obviously fiscal costs associated with additional sampling and analysis.
- As per Jenkins (2013), future reporting should include statistical analysis (both univariate and multivariate), including analyses of the relationships between abiotic and biotic variables.
- Improvements to analyses could include; DISTLM or Canonical analysis of principle co-ordinates as well as other PRIMER functions to see if the percentage mud or other environmental variable explain the variability seen in the community data.
- SIMPER analysis (PRIMER) could be used to look at species which contribute to the differences in in community structure between sites/times.
- We recommend removing fish taxa and nematodes (meiofauna) from the analysis.

2.4 Water quality

It is relatively simple to measure the concentrations of solutes and solids that are suspended in estuarine waters (there are standard sampling protocols and analytical methods). However, using this information to assess 'water quality' in estuaries is notoriously difficult, because of the high variability in water quality parameters associated with tidal and freshwater flow variation. Moreover, uptake of nutrients by plants can confound the interpretation of nutrient concentration data; an estuary with high rates of nitrate loading may have relatively low concentrations of nitrate in the main water body if the loaded nitrate has been taken up and incorporated into plant biomass. Thus, in theory, an estuary with many of the symptoms of eutrophication may have lower than expected dissolved inorganic nutrient concentrations.

There is little direct information/data on estuarine water quality in the Southland Region. Instead monitoring focuses on freshwater quality and marine quality in relation to human health. Freshwater quality is measured at monthly intervals at 71 sites in 41 rivers/streams for physical and chemical variables (pH, temperature, turbidity, nutrients, DO) and further 'bio-monitoring' is conducted in summer months (measures of phytoplankton concentrations, invertebrates, etc.) (http://www.es.govt.nz/environment/water/surface-water/quality/). Marine monitoring focuses primarily on outer coastal locations in relation to the suitability/risk for bathing (concentration of bacteria). Estuarine water quality is not measured but is inferred from i.) macroalgal coverage (%

cover & density), ii.) sediment nutrients (TOC, TN & TP), and iii.) nutrient loading models, specifically, CLUES model predictions.

2.4.1 The suitability and precision of proxy measures

i) macroalgae as an indicator of nutrients

Macroalgae can be valuable indicators of changes in nutrient availability (Barr et al. 2013). They can take up and assimilate nutrients directly from the water column, or from sediments. As they are sessile, they generally represent an integrated measure of nutrient availability at a fixed location. Although macroalgae are likely to have been present in Southland area estuaries for millennia, the presence of large quantities of specific types of macroalgae (e.g., 'luxury consumers' of nutrients such as *Ulva* and *Gracilaria*) can be viewed as symptomatic of elevated nutrients. However, the sensitivity of macroalgae coverage to indicate the current magnitude of nutrient enrichment is not well explored or established in Southland. There are added complications of legacy effects, temporal changes and feedbacks, for example, macroalgae facilitating sedimentation and sedimentation affecting macroalgae, etc. Macroalgal distribution is highly variable between and within estuaries, and nutrient concentrations are just one component affecting macroalgal growth. Therefore the macroalgal percentage cover or the total area coverage (ha) of an estuary will not necessarily be tightly related to the degree of nutrient loading. Other factors will be highly influential: temperature, insolation, water turbidity, the estuary's water residence/flushing time, the degree of wind exposure/fetch, substrate types, and biotic influences such as the densities and types of grazers (Steffensen 1976, Ren et al. 2014). Due to these complex dynamics, the absence of macroalgae does not indicate the absence of a nutrient problem *per se*. However, in the estuaries where macroalgal coverage is increasing in extent and density, we know that nutrients are in surplus and management is a high priority. Rather than coverage *per se*, other components of macroalgae can be employed as indicators. For example, Savage (2009) and Barr et al. (2013) demonstrate how the tissue-δ^{15}N and tissue-N values of *Ulva* can be used to indicate anthropogenic nutrient loading in New Zealand estuaries after factoring out natural cues.

ii) Sediment nutrients

The nutrients that are held in estuarine sediments are an integration of multiple processes over time. The Southland Estuaries reports contain information on Total Nitrogen (TN) and Total Phosphorus (TP). These are relatively broad measures that do not discriminate between organic forms (fresh tissue, decaying material, meiofauna, phyophytins etc) and inorganic forms (e.g. ammonium, nitrate, nitrite, phosphate etc) and thus whether or not they are available for uptake by plant species. High concentrations of nutrients in sediment pore waters typically indicate historically high levels of organic loading and an exceedance of the sediment's capacity to process and recycle available organic/inorganic matter. This capacity is influenced by multiple factors, including the types and densities of benthic biota present (Lohrer et al. 2015). Sediment nutrient measures are indicative of processes happening at a site scale, but do not necessarily reflect current water quality conditions and do not necessarily drive macroalgal growth (which is influenced by both water-column nutrients and nutrients that are released from sediments).

iii) Model predictions or nutrient concentrations

CLUES (Catchment Land Use for Environmental Sustainability) is a GIS based model for predicting water quality and socio-economic indicators as a function of land use (Elliot et al. 2011). This approach can be used to calculate nutrients entering estuaries. CLUES models estimate loads of nitrogen entering Fortrose Estuary based on 2002 land cover (2,450 tonnes N year^{-1}) and are predicted to be higher based on current land use (>4,000 tonnes N year^{-1}, Stevens and Robertson 2013A). CLUES nutrient outputs are not provided for the other three estuaries (only sediment loading) by Wriggle Coastal Management.

2.4.2 Water quality of Southland estuaries

The high abundance of the macroalgal species *Enteromorpha*, *Cladophora*, *Ulva* and *Gracilaria* in Southland estuaries is generally suggestive of nutrient over-enrichment, with the problem appearing to be particularly acute in the New River and Jacobs River estuaries (each with extensive areas of 100% macroalgal cover). Current Southland Estuary monitoring indicates nutrient status at a broad, qualitative level (presence/absence of macroalgal mats that suggest eutrophication). It does not measure precise or nuanced changes in nutrient status over time. Achieving the latter point is difficult even with comprehensive and frequent sampling. Estuarine water quality monitoring is conducted elsewhere in New Zealand (e.g., Walker and Vaughan 2013), as a component of an integrated monitoring framework. However, it is difficult to comment on the effectiveness and costs: benefits of implementing such a programme for Environment Southland, particularly given the complexities mentioned above. If Environment Southland is to move towards limit setting of nutrient inputs and the identification and management of nutrient sources, it likely that a comprehensive monitoring programme will be required to a.) monitor the nutrient status moving forward b.) to refine and validate model predictions, c.) evaluate and report compliance and identify non-compliance.

2.4.3 Water Quality monitoring elsewhere in New Zealand

Auckland Council has a long-term marine water quality programme operating at 35 estuarine and harbour sites across the region. Water samples are collected on a monthly basis and analysed for dissolved oxygen, DO saturation, temperature, conductivity, salinity, pH, suspended sediments, turbidity, chlorophyll *a*, nitrate, nitrite, ammoniacal nitrogen, nitrate/nitrite sum, total kjeldahl nitrogen, total nitrogen, soluble reactive phosphorus, total phosphorus and enterococci concentrations. The purpose of the monitoring programme is to i) satisfy Resource Management Act obligations and the need to maintain and enhance the quality of environment monitoring (Local Government Act 2002), ii) provide baseline, regionally representative data to support the resource consent process and compliance monitoring and iii) to assist with the identification of large scale and/or cumulative impacts of contaminants associated with varying land uses and disturbance regimes and link these to particular activities (Walker and Vaughan 2013). Data from AC's marine water quality monitoring programme is joined with the sediment contaminant programme and the benthic ecology monitoring programmes[9] to provide an integrated view of the physical, chemical, and biological conditions of the marine environment (Walker and Vaughan 2013).

[9] It was linked to the shellfish contaminant monitoring programme in the past, but is no longer linked.

3 Site selection and representativeness of monitoring sites

3.1 Site selection

Whilst most of the estuaries in the Southland region have been sampled, the four where the most comprehensive monitoring information was available were selected for investigation by NIWA. Additional reasons for selecting these estuaries included their spatial spread along the Southland coast, the fact that they represented a gradient in size from relatively small to large, and that they spanned a perceived gradient in ecological condition from poor to good.

3.2 Appropriateness and critique of existing monitoring sites

Fine scale monitoring and eutrophic sites were visited by Drs Lohrer and Townsend from the 23rd-26th February 2015 (Figure 9).

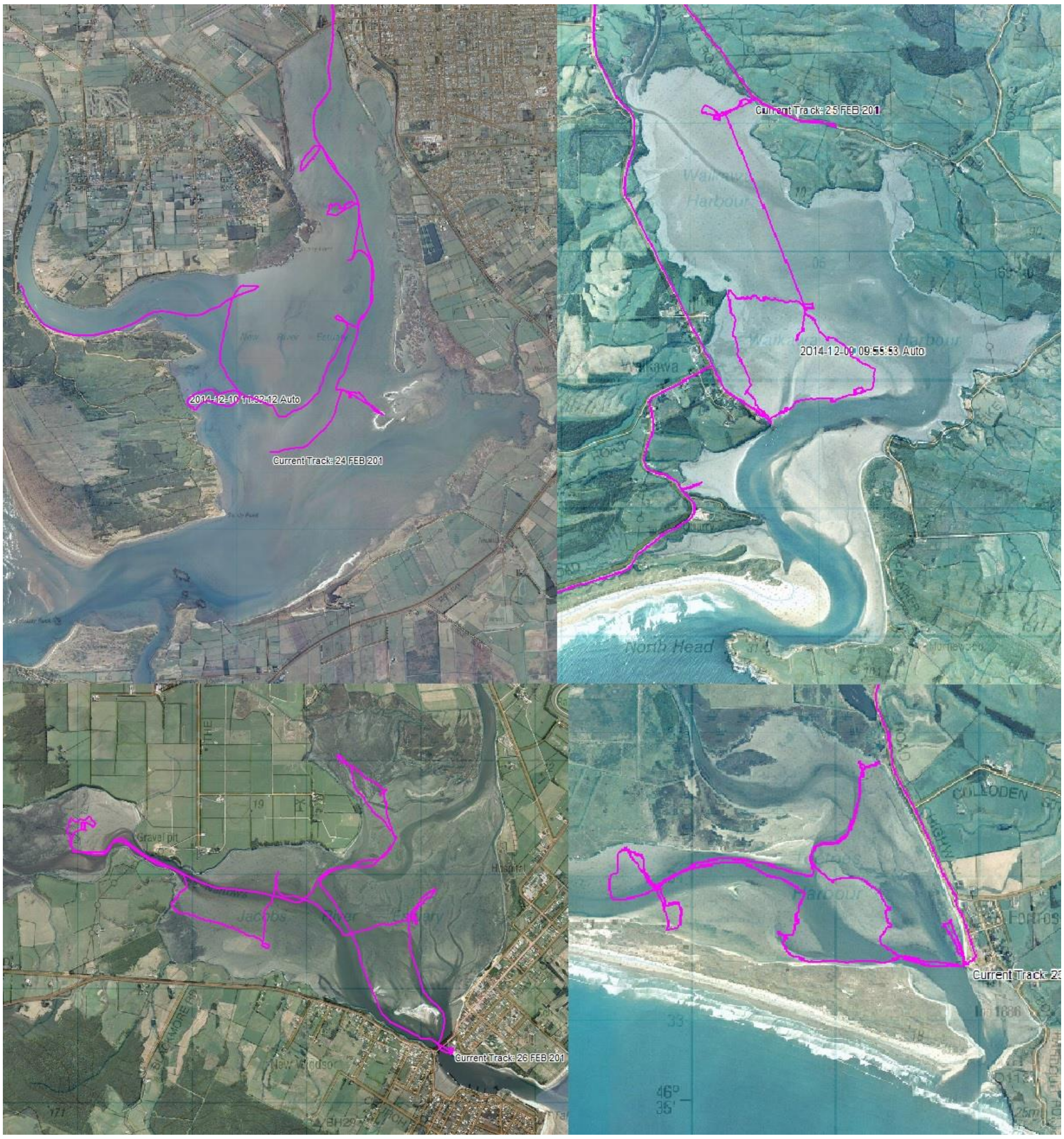

Figure 9: Routes (pink line) taken around the Southland Estuaries. New River (top left), Waikawa (top right), Jacobs River (bottom left) and Fortrose Estuary (bottom right); estuary maps not to scale.

3.2.1 New River Estuary

The New River Estuary covers a range of conditions; with sizeable areas that are reported to be muddying, enriched and in poor ecological condition, and other "healthier" sections. Existing fine-scale monitoring sites B, C & D do a suitable job of representing the prevalent conditions in the central section of the estuary; reflecting the sandier sediment characteristics and fauna. Likewise the eutrophic sites (E, F), located in two of the most degraded parts of the estuary, adequately reflect depauperate and impacted conditions. Collectively these two types reflect the contrasting conditions of the New River Estuary (Appendix A).

The Eutrophic sites were sampled in 2011 (Site W, a preliminary assessment), 2012 (Site E & F) and 2013 (Site E & F) with recommendations for sampling in February 2014 and 2015 (Robertson and Stevens 2013A). Annual sampling of impacted sites may pose lower value to Environment Southland given that i) sites are highly impacted already and it may be difficult to show further community changes (barring the complete loss of all invertebrates), ii) sites are unlikely to substantially improve over short time-scales. Further deterioration may be assessed visually at first or indicated from a wider presence of *Beggiatoa* bacterial mats.

Site C is a useful monitoring site due to its proximity to eutrophic conditions. Should the large macro-algae area in Daffodil Bay expand further down the shore, it is likely that this will be detected and demonstrated in the macrofaunal data at Site C. Thus Site C serves as a potential early indicator of change. Site B appears relatively wave-swept compared with higher up the shore. While the monitoring site would be better located ~150 higher further up the shore, for consistency in the data it is not recommended that the site is moved.

3.2.2 Jacobs River Estuary

The fine-scale monitoring sites in Jacobs River Estuary are collectively a good representation of the Estuary. Sites A and B reflect the Central Basin and Southern Sandflats respectively, in terms of sediment composition and macro-algal coverage. Site C in the Pourakino Arm, is less reflective of the wider estuary and has issues with homogeneity across the monitored area (see Appendix A for full site description). However, the lack of uniformity at site C is likely linked to the hydrodynamics and mobile sediments. For consistency in the data, it is not recommended that site is moved. However in the future when sampling is conducted, it is suggested that the degree of intrusion of the sandbar is noted. Records could be kept of the position of sampled cores in relation to this intrusion (i.e. which samples come from the sandbar, next to it, away from it, etc.) as this could allow a more structured data analysis. If the intrusion increases, the suitability of the site would need to be re-evaluated. The strength of Site C is its proximity (50 m) to the eutrophic side of the Pourakino Arm facilitating detection of any spread of eutrophication in this location. The current distribution of macroalgae in the Pourakino Arm reveals a relatively sudden transition between the vegetated and unvegetated sides. As with other Estuaries in the Southland Region, the value of intensively sampling eutrophic sites may be relatively low; resources could perhaps be reallocated to improve the temporal frequency of existing fine-scale monitoring.

3.2.3 Fortrose Estuary

Full descriptions of the fine-scale monitoring sites are provided in Appendix A. *Ulva* was widespread in Fortrose estuary and absent only in limited places towards the Mataura River. Its distribution appeared to be structured by hydrodynamic forces and attachment substrate (pebbles, cobbles, shellhash, etc). Monitoring Sites A and B had *Ulva* patches (including thick decaying piles of *Ulva* with anoxic sediment underneath) throughout their 60 x 30 m areas. Due to this, and the fact that waves and currents may move the *Ulva* wrack from time to time, macro-invertebrate composition may be highly variable within and among sampling periods, and among years. Locations D and E (Appendix A) illustrate the potential for legacy effects of *Ulva*, and how the length of time the *Ulva* has been present at a site can alter the magnitude of its influence. For example, at Site D, there was an absence of *Ulva,* but an indication that *Ulva* wracks had been there previously and had disturbed the benthic environment. At Site E, *Ulva* was present at the time of our visit, but there were no visible indication of gross changes to the benthos (mature benthic invertebrate community, well oxygenated sediments), suggesting that no *Ulva* die-offs had occurred.

Given the patchy nature of *Ulva* throughout the majority of the estuary, Fine-scale monitoring Sites A and B are broadly representative and in suitable locations. Sandflat "C" (Appendix A) is another interesting location where a monitoring site could be usefully established. Sandflat "C" had been noted as 'soft mud' and "getting muddier" by Stevens and Robertson (2013) but this was not evident on the lower two-thirds of the sandflat (Figure 10). Consideration should be given to the establishment of sediment accumulation plates and a quantitative measurement of sediment properties at this location. This will reduce the arbitrative/anecdotal assessment of increasing muddiness.

Figure 10: Designated soft mud locations at Fortrose Estuary. Photo on the left, (position 46 34 10.8240, 168 45 52.146); photo on the right (position 46 34 7.999, 168 45, 52.999).

3.2.4 Waikawa Estuary

Fine-scale monitoring Sites A and B are both representative of the central section of Waikawa Estuary. They complement each other: Site A playing a sentry role for incursion of mud from the upper estuary and Site B containing a diverse, sand preferring community. Existing monitoring sites do not reflect the conditions and the macrofaunal communities of the Upper Estuary, however, establishing a site there should be of lower priority compared with continued monitoring of the existing sites.

From a visual assessment, Site A was heterogeneous, with two distinct 'halves' to the 30 x 60 m area. This is not ideal for monitoring, as relative homogeneity is a common design criterion, though the stratified-random sampling design used by Environment Southland will help limit the impact of this on the macrofaunal data. We recommend that notes or photos documenting degree of difference across the site are collected on each visit to help in the interpretation of the data. The sediment monitoring site at the north end of the estuary was quite soft, and visitation of the site resulted in deep footprint marks throughout the sampling area, which could influence the accuracy of sedimentation measurements (particularly if sampling is frequent, e.g., monthly).

4 Our Assessment of Ecological Health

In the following section, we report on the observations we made during single-day site visits to the four Southland estuaries in question. Obviously, a visit to a site at a single point in time does not allow for the assessment of temporal trends, only the general status of the estuary at the time of the visit. Moreover, although we observed a number of distinct areas within each estuary (and documented a reasonable proportion of each using geo-positioned photos and video), our observations were entirely field based and qualitative (in that no samples were collected and analysed at a laboratory). Thus, in the sections below, we rely to a degree on information presented in the Wriggle reports, particularly when discussing the likelihood of trends over time.

4.1 Waikawa Estuary

In Waikawa Estuary, there appeared to be a strong contrast between the central and lower sections, which are in a good state of ecological health, and the upper estuary which is moderately healthy.

The upper section is muddy underfoot and appears to be a highly depositional area. Despite this, and in comparison with other muddy areas in Southland, it does not appear to be highly degraded: the sediment is relatively well oxygenated, lacks a sulphide smell, and the cover of macroalgae (in this case *Gracilaria*) is currently low (~1-5 %). This area was observed to support a muddy community of species (*Amphibola crenata,* mud crabs and polychaete worms). The monitoring programme undertaken in this estuary does not include invertebrate sampling in the upper estuary, so an assessment of macrobenthic communities based on the written reports was not possible. According to the reports, mud covers approximately ~40% of the estuary, which is less ideal with respect to an ecological functioning perspective or that of amenity values. However, observations by Stevens and Robertson (2009) have indicated that the boundary of soft mud has retreated 80m towards the upper estuary over the last 10 years, suggesting that conditions may gradually be improving. Sediment plate measurements are interpreted with caution, but in this less exposed harbour, results are probably reliable. They corroborate a reduction in fine sediments over time (loss of sediments, rather than accumulation) at the 3 sediment monitoring sites, which may be indicative of recovery.

The central and lower sections of the estuary are in good ecological condition. Sediment characteristics are predominantly sandy, around 90% and 96% sand at Sites A and B, respectively, and have shown minimal change over time. The sediment is well oxygenated with deep aRDP layers of around 3 cm. Heavy metal contaminants have been consistently and substantially below the ANZECC ISQG Low trigger values over time and also below the Threshold Effects Level (TEL)[10] values Auckland use and TOC% has been variable but low (<1%). The benthic communities appeared healthy at both Sites A and B at the time of our visit, and appear to have remained this way over the majority of the monitoring period with functionally important species present (*Austrovenus stutchburyi, Macomona liliana*) and moderate diversity. However, there was a notable reduction in abundance and diversity at both sites during 2013. The cause of this is unknown at present, however, data collected by NIWA in 2014 and visual observations in February 2015, indicate conditions more similar to the pre 2013 communities (i.e., a high density of bivalves within monitoring sites and indications of moderate/high biodiversity: 35 species recorded, abundance of 44 *Austrovenus stutchburyi* in 10 cores, abundance of 587 *Macroclymenella* stewartensis in 10 cores). Over time, species that are common in muddy or degraded areas have not been abundant at Sites A or B, such as Nereids, Oligochaetes, Corophids, and mud crabs. The benthic communities of Sites A and B are distinct and

[10] TEL values for copper, lead and zinc are 19, 30 and 124 mg/kg respectively. In comparison, concentration at Waikawa Site A in 2013 were on average 2.4, 1.6 and 13.0 mg/kg for copper, lead and zinc respectively.

both have shown similar change over time (Sites A and B tracking in the same direction on nMDS plots; Robertson and Stevens 2013B). Changes in community structure were relatively small between 2005 and 2008. The drop in 2013 does not seem to have been caused by solely low mud content (as suggested by Robertson and Stevens 2013B), given that diversity and abundances were much higher in 2006 when mud content was at its lowest (5.3 and 1.1% for Sites A and B respectively).

Macroalgae monitoring was conducted in Waikawa estuary in 2007 and 2008 but has not been conducted since. Overall macroalgae coverage was low but there was variation between these years: In 2007, 75% of the harbour had 5-10% macroalgae coverage, whereas in 2008 98% of the harbour was <5%. In February 2015, during our site visit, we observed both *Ulva* and *Gracilaria* to be present in low density in the central and lower sections of the harbour. Macroalgae was commonly attached to cockle shells or shellhash.

In summary, Waikawa Estuary appears to be mostly in a good health, based upon the majority of variables examined to date. The muddy sediment in the upper harbour is a negative, although the areal extent of this appears to be contracting. Nevertheless, the sheltered conditions of the upper estuary and small changes in mud since 2004 suggest that residence time for the deposited mud is high, and that Waikawa may remain muddy for an extended period even if current sediment loads in the catchment are well managed. Importantly, though, suitable management would prevent the upper estuary from further deterioration and would facilitate recovery over the longer-term. The possible expansion of macroalgae in Waikawa should be a concern, given that this estuary has sheltered sections and a prevalence of attachment substrate (shell material) that could facilitate expansion. Management of nutrient input is necessary to prevent a threshold change. The sandflats here could begin to resemble Fortrose Estuary in its current state if not managed appropriately.

4.2 Fortrose (Toetoes) Estuary

At the time of our visit, Fortrose Estuary appeared to be in a moderate state of ecological health, with the degradation seeming to be driven by the effects of nutrient enrichment. According to the Wriggle reports, the physical and chemical sediment properties at the two monitored sites were in good condition with low mud content (<9%), well oxygenated sediment, low total organic carbon (≤0.5%), low nitrogenous and phosphorus nutrients, and heavy metal contaminants well below the ANZECC ISQG-Low threshold[11]. These physical and chemical properties suggest that sedimentation is not a major problem at these sites and is secondary to the main stress of nutrient enrichment. There have possibly been increases in fine sediments on the flats towards the Mataura River and other smaller sections in the central body of the harbour, but underpinning information in support of this assertion is not strong.

Macrofaunal community data is available for this site from 2004, 2005, 2006 and 2009 (Robertson and Stevens (2009). Community abundance was dominated by few species, with *Paracorophium* sp. accounting 56% of the total abundance and two further species, *Potamopyrgus antipodarum* and *Potamopyrgus estuarinus*, accounting for another 25%. These species are detritivores and grazers, and their high abundance is likely responding to algal food sources. Functionally important species, notably large infaunal bivalves such as cockles and wedge shells, were less common.

The ecological condition in Fortrose Estuary is most notably indicated by the amount of macroalgal coverage (predominantly *Ulva*, but with some *Gracilaria* present also) in the intertidal areas. We observed thick piles of tangled drift *Ulva* (still green, up to 20 cm thickness, patches as big as 2-3 m^2)

[11] Last monitoring was 2009

on the sandflats, and these piles appeared to have smothered the cockles and pipis present underneath (dead bivalve shells were abundant under the mats). Live cockles and pipis were also observed attached to long blades of *Ulva* (via the holdfast); when the blades become long enough, the shellfish are susceptible to being pulled from the sediments by hydrodynamic forces dragging on the blades. Macroalgal coverage also appeared to be driving low oxygen and sulphide rich conditions in parts of the estuary (Figure 11). Small-scale patchiness is apparent and widespread, and the broad percentage coverage categories of Stevens and Robertson (2013A) should be used as a guide only. There are large areas of high density macroalgae, particularly in the vicinity of the Tokanui Gorge Road. Since 2003, the density and the scale of the problem of macroalgae has increased. In 2013, 22% of the estuary had macroalgae cover >50%. Our observations in 2015 found the sandflat on the southern side of the Harbour to be covered in 80-100% macroalgae (Figure 12), a location that had just 5-10% coverage in 2013. This suggests that macroalgal coverage has increased since the last broad-scale survey in 2013. The presence of macroalgae is triggered by nutrient enrichment, but its distribution is in part structured by hydrodynamic forces and attachment substrate (pebbles, cobbles, shellhash etc). Seagrass is not prevalent in the harbour according to recent reports. Saltmarsh coverage was recorded as 84 hectares in 2013. Historical baseline information on seagrass and saltmarsh cover is unavailable for this estuary.

In summary, Fortrose Estuary is dominated by firm sandy and gravelly sediments, with the accumulation of fine sediments appearing to be less of a problem than the expansion of *Ulva* coverage since 2004 (which is now extensive). Limiting nutrient loads from the catchment to the estuary may address the estuary's primary issue, resulting in a relatively rapid reduction in *Ulva* coverage in the estuary's intertidal areas, which would benefit resident macrofauna and shellfish populations and amenity values to humans.

Figure 11: Anoxic sediment in Fortrose Estuary.

Figure 12: **Macroalgal coverage in Fortrose Estuary.**

4.3 Jacobs River Estuary

The ecological health of Jacobs River Estuary contrasts between the central and lower sections of the harbour, which are in a good or moderate state, and large sections of the upper estuary, which are in a poor state of health. The areas in poor health exhibit characteristics of nutrient over-enrichment and sedimentation.

Sediment grain-size at all 3 of the environmental monitoring sites is dominated by sand (>90% with the exception of Site C in 2006) and has been variable over time with no strong trends (range in mud content: Site A 0.78-6.9%; Site B 2.83-7.7%; Site C 3.95-13.6%). The chemical characteristics were in a good to moderate state in 2011: heavy metal contaminants (Cu, Zn, Pb, Ca, Cr, Ni) were well below the ANZECC ISQG Low level for all metals with the exception of Ni at Site A (2011 Concentration of 13.2 mg/kg at Site A, ANZECC ISQG Low level is 21 mg/kg). The Site A value is close to the more conservative nickel concentration of McDonald et al. (1996) and their Threshold Effects Level (TEL) of 15.9 mg/kg. The total organic carbon, phosphorus and nitrogen have been low at all three monitored sites with only small changes over time. The biotic characteristics were in a good to moderate state: both Sites A and B are in good health with high species diversity (25 and 29 spp. present in 2011). Pipi (*Paphies australis*) are an abundant species at Site A and comprise 43% of the total community abundance. Across Sites A and B *Boccardia syrtis* and *Aonides* sp.1 are the two most abundant polychaetes and cockles (*Austrovenus stutchburyi*) are present but in low density. Site A is exemplifies the wider sandflat in the exposed section of the Central Basin, characterised by wave rippled firm sands and low macroalgae coverage. Site C also has a good level of species richness (23 species in 2011), however, evenness is low with the snail *Potamopyrgus antipodarum* comprising 64% of the total community abundance. This snail grazes and feeds on decomposing material, and the high abundance is likely symptomatic of nearby disturbance and enrichment, indicating poorer ecological health. Communities have shown some change over time (Figure 8 in Robertson and Stevens 2012), with the greatest variability occurring at Site C.

Since 2003, there has been little notable change in the areas of saltmarsh and Seagrass (>50% cover) across the estuary. Across the estuary, a significant proportion of the intertidal flats are soft mud (~30%) and this has remained largely unchanged since 2003. Soft mud is associated with the areas of high macroalgae density (typically 80 to 100%). In 2013 there were large areas of 80 to 100% macroalgae coverage in the Pourakino River arm, the upper flat by the Aparima arm and on the western site of the southern flats (immediately prior to the Pourakino River arm). There were also small patches of 80 to 100% macroalgae coverage in the central basin of the estuary. In 2013, close to 60% of the estuary had a macroalgae cover of greater than 50%. Changes in macroalgae coverage between 2008 and 2011 were relatively small; most notably a reduction in the area of 80-100% coverage along the fringe of the channel edge on the Southern Flats. However, much larger changes were evident from 2011 to 2013, with a large increase in coverage in the Pourakino River arm, expansion of the large 80-100% coverage area in the Northern Flats, and an increase in coverage in the central basin (approximately half of this section was 20-50% coverage in 2013 compared with 1-5% in 2011; Figure 13). Expansion of macroalgal coverage in the Pourakino River arm has resulted in the loss of seagrass.

Associated with the macroalgae, there is compelling evidence of large eutrophic areas in the Pourakino River arm and the upper flat by the Aparima arm. Sites D and E are in poor ecological health due to nutrient and sedimentation issues. For example, the high coverage of macroalgae (typically associated with nutrient over-enrichment) appears to have resulted in the trapping of fine sediment, with both sites having mud content >50%. Heavy metal contaminant concentrations were well below the ANZECC ISQG Low level for most metal species (Cu, Zn, Pb, Ca, Cr), reflecting the largely rural catchment. The exception was nickel, which was in concentrations of 15-16 mg/kg compared with the ANZECC ISQG Low level is 21 mg/kg. Sediment nutrients are elevated at Site E and considerably higher at Site D compared with the monitoring Sites A-C for total organic carbon, phosphorus and nitrogen. Invertebrate communities at both sites had low species richness, evenness and diversity. The community at Sites D and E were dominated by surface dwelling epifauna rather than infauna; amphipod and corophid species and *Potamopyrgus* species. Two infaunal species with relatively high abundances were the mud crab *Hemiplax hirtipes* and *Arthritica* sp., both of which are tolerant of muddy conditions. We observed oxygen poor and sulphide rich sediments at these sites during our site visit.

In summary, the ecology of Jacobs River Estuary appears to be suffering from the impacts of nutrient enrichment, with the broad and increasing coverage of macroalgae an indication of its declining health. Algal coverage will vary from year to year for numerous reasons and is not solely a reflection of the magnitude of nutrient inputs. However, the changes from 2008 to 2013, and observations in 2015, suggest expanding eutrophication effects in this estuary. This is compounded by the ‘dead spots’ (few infaunal species, low oxygen and poor sediment environment), which are left in the wake of these expanding of macroalgal mats. Jacobs River Estuary is severely compromised in certain parts, with no indications of improvements, and some evidence of further deterioration.

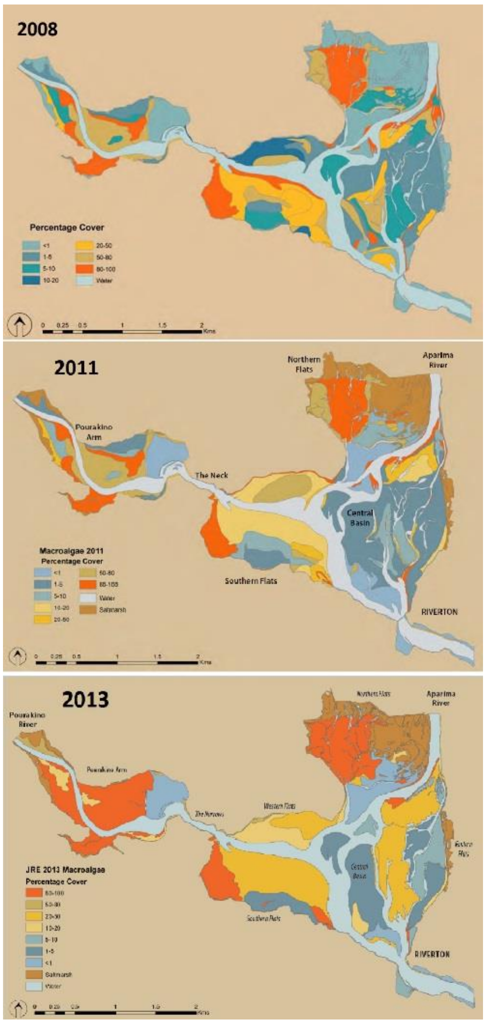

Figure 13: Changes in macroalgal coverage in Jacobs River Estuary over time. Macroalgal coverage from Stevens and Robertson (2008), (2011) and (2013B).

4.4 New River Estuary

New River is the largest estuary that we visited, and there are large sections in the central and outer portions of the estuary that appear to be moderately healthy. Yet, other sections of the estuary contained some of the most degraded areas we have ever seen. According to the Wriggle reports, the degraded eutrophic zones cover a greater total area than those present in Jacobs River Estuary, though the these zones comprise a smaller proportion of total estuarine area in New River Estuary. Our observations, in agreement with data from the Wriggle reports, suggests that the degraded areas in New River Estuary are in far poorer condition than those in Jacobs River Estuary.

Eutrophication is most apparent in the Waihopai Arm of New River Estuary, which is also the single largest area of soft mud. A high proportion (20%) of the sediment in New River is soft mud and there seems to have been a long legacy of sedimentation (Robertson and Stevens 2007). However, data supporting increases in soft mud coverage since 2001 are not strong (Section 2.1.1). The areas of soft mud in this section of the estuary are commonly associated with dense macroalgae. Comparison of percentage cover of macroalgae from 2007 to 2012 (Figure 14) showed a marked increase in the area of dense (>50%) cover. Our observations in 2015 did not indicate obvious changes in the area of dense cover relative to 2012. Wriggle have reported increases in macroalgal coverage in the western flank of the Waihopai Arm, on the eastern side by Kew Road, South of Bushy Point, and in Daffodil Bay on the western side of the harbour (Figure 14). Increases in the Waihopai Arm have purportedly driven a loss in the coverage of seagrass. Both eutrophic sites are muddy (>50%), have an aRDP layers at the sediment surface, and have high levels of sediment pore water nutrients; particularly at Site F in the Waihopai Arm. Decomposing algae drives the high sediment oxygen demand and the resultant hypoxic/anoxic pore water. This also results in the release of odorous gases from the sediment, particularly hydrogen sulphide[12]. Judging from the Wriggle reports, invertebrate communities at Sites E and F are highly disturbed: low species richness (16 and 14 species respectively), low species evenness (at Site E two amphipod species comprised 90% of community abundance, at Site F an amphipod, a paracorophid and the snail Potamopyrgus comprised 75% of community abundance) and the abundant species were epifaunal (not living within the sediment itself).

These grossly eutrophic areas contrasted sharply with the other fine scale monitoring sites (sites B, C and D). Sites in the central harbour had oxygenated sediment with aRDPs of 2-3cm, low mud content (<6%) and low organic content. Data from 2001, 2003-2005 and 2010 indicates possible[13] increasing trends in sediment mud content at Sites C and D over time (R^2 of 0.71 and 0.83 respectively), although this is partially driven by high values in 2005/2010. Mud content at Sites C and B showed similar variation over time. Organic carbon, nitrogen and phosphorus levels are generally low, although total nitrogen was elevated in 2010 compared with previously measured quantities at all sites.

Invertebrate communities at the fine scale monitoring sites appear to be moderately healthy and distinct from each other (Figure 8, Robertson and Steven 2010). Sites B and D were similar in 2004, but have gradually become dissimilar over time. In 2010, Site B had moderate species richness (18 spp. across ten replicates), low abundance and high evenness. Site C had higher species richness (24 spp. across ten replicates) and moderate evenness, with the most abundant species being the mud tolerant polychaete *Microspio maori*. Site D had the same species richness as Site C, but low evenness, with 3 species comprising 84% of the community abundance. These were the snails *Potamopyrgus antipodarum* and *Potamopyrgus estuarinus* and the mud tolerant bivalve *Arthritica bifurca*. The common species at these sites are all tolerant of some mud content.

In summary, the ecology of New River Estuary is suffering from the impacts of nutrient enrichment, and the broad and increased coverage of macroalgae is an indication of declining condition. The eutrophic areas are in particularly poor ecological condition, few species are able to persist in the sediment and conditions so poor that macroalgae are unable to grow during summer months (Figure 15). The estuary is displaying signs of severe hypoxia with sulphide complexes and Beggiatoa sp. at the sediment surface (Figure 15). Hydrogen sulphide gas emissions are readily smelt in the most impacted areas and are high enough to cause headaches and nausea.

[12] From time spent in the Waihopai arm, airborne concentrations of hydrogen sulphide were likely ~5-20 ppm based on negative health symptoms we experienced https://www.osha.gov/SLTC/hydrogensulfide/hazards.html
[13] Mud content has been higher in 2005 and 2010, although this is at an early a stage to confirm with certainty

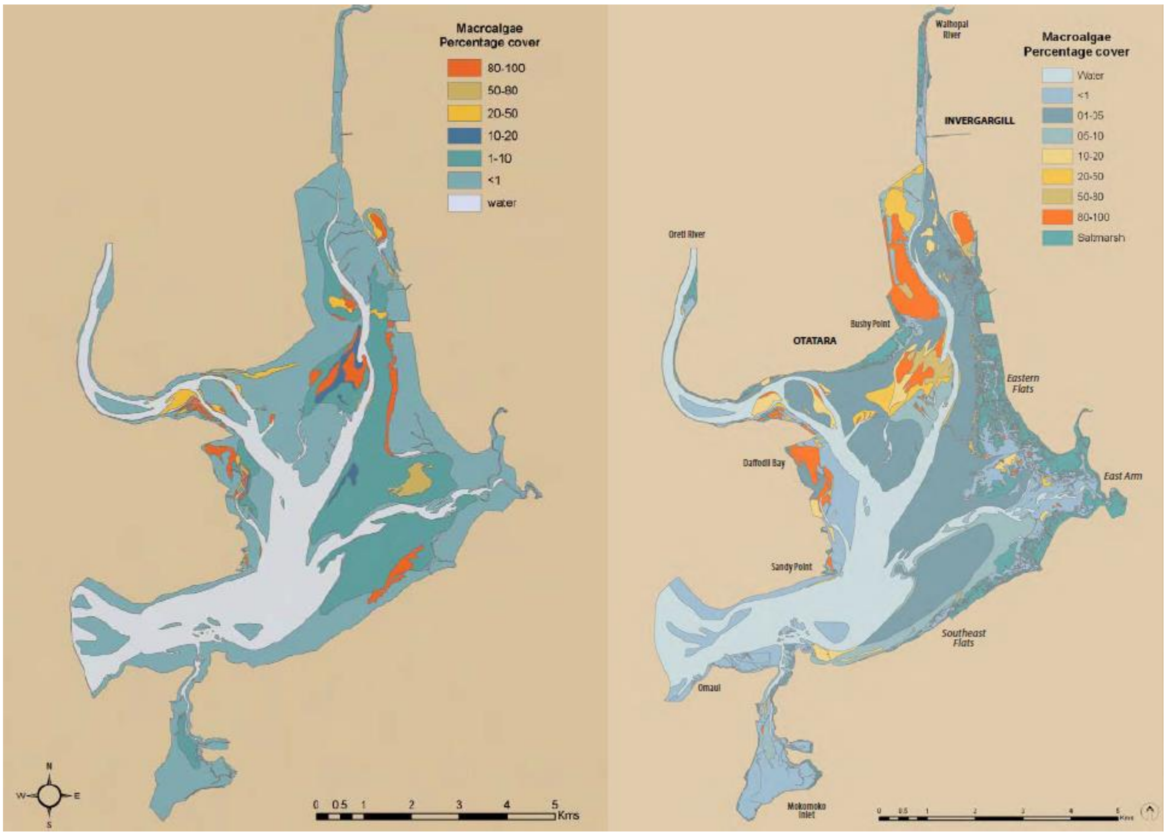

Figure 14: Changes in macroalgal coverage in New River Estuary. Macroalgal coverage from Robertson and Stevens (2007) and Stevens and Robertson (2012).

Figure 15: An indication of the sediment surface in the Waihopai Arm of New River Estuary. From the site visit in February 2015.

4.5 Deviations from previous assessment

In reviewing monitoring reports, we encountered a number of small inconsistencies and instances where statements were not backed up by the available evidence. Most of these problems are relatively minor, but together they begin to take a toll on the integrity of the assessments. Our concern is that this opens up the reports to criticism, and will cause the overall conclusions (which we generally agree with) to be discounted. Some examples of problems include:

- Community changes are not appropriately or accurately linked to environmental variables.

For example, in 2013, the low abundance and diversity of fauna at Waikawa Sites A and B is attributed to low mud content (2.7-6.3%). This is speculative and not supported by other data (e.g., mud content of 1.1-5.3% in 2006 at the same sites was associated with diverse and abundant communities). While other data from a range of Auckland areas sites suggests that mud content <2% can have low species richness, richness is generally at near its maximum in the 2.7-6.3% mud content range (Rodil et al. 2013).

- Missing information on how things have changed over time.

For example, in 2013, the fine sediment content in Fortrose Estuary was said to have increased since 2003. This appears to be an anecdotal assessment, as the maps from 2003 and 2013 did not cover the same area of intertidal habitat (Figure 16 below). The tidal state during the two samplings was probably different. Another possibility is extensive sand transport (i.e., the appearance of entirely new sandbars).

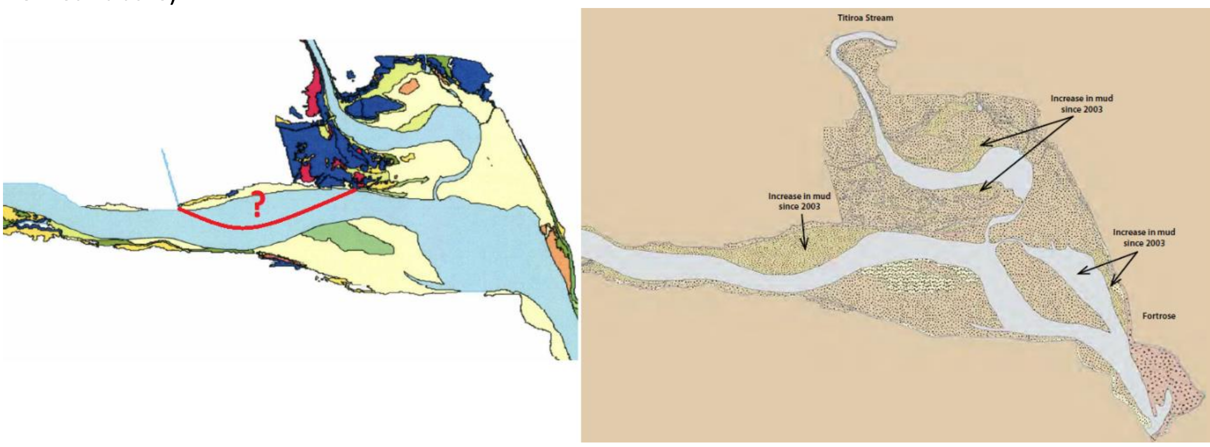

Figure 16: Broad scale coverage at Fortrose Estuary, 2003-2013.

- Prioritisation for recovery is not properly explored.

In Stevenson and Robertson (2012) it states for Fortrose Estuary that "because the estuary is currently in a low to moderate state of enrichment, the estuary does not have the same high urgency as New River or Jacobs River estuaries". This statement does not explore or justify recovery dynamics or prioritisation. It can be well argued that Fortrose Estuary should be the highest management priority, because the impacts are associated primarily with (a) water borne nutrients, rather than accumulated bed sediment mud content, and (b) the dominance by ephemeral/drift *Ulva*, rather than attached/meadow-forming *Gracilaria*. Fortrose does not yet contain large eutrophic/anoxic zones, and this estuary may be able to recover relatively quickly if water column nutrient loading from the catchment is sharply reduced. In contrast the vast muddy areas and extensive cover of thick *Gracilaria* mats in New River and Jacobs River may persist for extended periods even after appropriate management steps have been taken, a factor that should be considered in management prioritisations.

- Unsupported and untested statements and the interpretation of data.

Differences in the data are largely untested (Jenkins 2013). There are multiple instances in the reports where statements are not supported by the available data. For example, In Robertson and Steven (2013A) they discuss the greater number of species living atop the sediment at eutrophic sites, compared with those living and feeding within the sediment column. In their interpretation they state:

"…Figure 2 which shows a reduced presence of subsurface feeders at the gross eutrophic sites compared with the main estuary basin sites."

This statement is untested, and we are not swayed by the supporting evidence presented (Figure 17).

Figure 17: **Figure 2 in Robertson and Steven (2013A).** The figure shows the mean number of species and abundances per core plus feeding types for New River Estuary 2001-2013.

4.6 Agreement with previous assessment

Although we have concerns with various methods, analyses, and interpretations of the data—and the lack of underpinning information to support of indices and bandings—the authors of the Southland Estuaries monitoring reports have collected a variety of useful information over a number of years from the four estuaries we examined. Importantly, by and large, the authors have correctly assessed the key issues affecting the four estuaries. They have been able to identify significant degradative changes over time, and have generally ranked the estuaries accurately in terms of their degree of healthiness. Essentially, we agree with Jenkins (2013) that the comprehensive range of measured indicators of the physical, chemical and biological condition provides a solid platform of information that is able to demonstrate the stark contrast between healthy and unhealthy sections of estuaries. There is overwhelming evidence to support significant anthropogenically generated eutrophication and sedimentation in the upper regions of New River and Jacobs River Estuary. Furthermore, although strong statistical trends may be absent for individual variables, the weight of evidence from multiple measured parameters indicates continued undesirable changes for New River, Jacobs River and Fortrose Estuaries.

5 Conclusion

Although we have concerns about the consistency and accuracy of some of the methodologies, we are not recommending the suspension of sampling or the relocation of sites in any of the estuaries. Instead we have offered ways in which the methods, analysis and interpretation need to be improved. The combined approach of fine scale monitoring coupled with broader-scale tracking of habitat change (e.g., increases in mud and macroalgal coverage) is certainly appropriate. Importantly, however, if care is not taken to standardise, ground-truth, and verify the accuracy and repeatability of the data being collected, and if the indices and bandings are not correctly underpinned and justified, then the exercise of attributing undesirable changes in estuaries to particular practices in the catchment becomes difficult. This then presents problems for managers who must set limits and convince the citizenry to curtail particular catchment practices in order to achieve better environmental outcomes. We are concerned that the data being collected at present and the reports on estuarine status and trends are not rigorous enough to withstand the scrutiny of external scientific experts, which could affect environmental management discussions and policy outcomes.

6 Recommendations moving forward

We have raised a number of concerns regarding the data acquisition methodologies, the potential misinterpretation of trends, and the lack of index justification / verification in the Robertson and Stevens reports (Robertson and Stevens, 2007, 2010, 2012, 2013A, 2013B, Stevens and Robertson 2008, 2009, 2011, 2012, 2013A etc). However, we believe that some of the questions and concerns raised in our review can be allayed with additional analyses of existing raw data, and it may be useful to reallocate sampling effort in some instances to generate better data sets moving forward.

The aim of these recommendations is to provide a more concrete and evidence-based link between catchment activities (principally, nutrient and sediment loading to estuaries) and the ecological condition of estuaries in the Southland region. It is important to recognise that a 'smoking gun' (tight, empirical, cause-effect relationship) will always be elusive due to a variety of methodological complications and scale-dependent factors.

Recommendation 1. Mine existing data, focusing on what we believe are the two most important and indicative data sets, namely, macroinvertebrate community data (fine scale), and ‘nuisance’ macroalgae cover data (broad scale). This may involve checks on the taxonomic resolution and quality of the macroinvertebrate data, and/or recalculation of the macroalgal coverage statistics from raw images.

Recommendation 2. Explore alternate macroinvertebrate sampling strategies to maximise their fitness-for-purpose moving forward. For example, it may be beneficial to cease macroinvertebrate monitoring at ‘grossly eutrophic’ sites (where rapid changes for better or worse are unlikely, and where general site conditions can be tracked with photos), and instead increase temporal resolution at the other fine-scale macroinvertebrate monitoring sites.

Recommendation 3. Conduct a comprehensive analysis of existing macroinvertebrate datasets, focusing on (i) changes in macrofaunal community composition over time, (ii) temporal trends in selected individual taxa, (iii) changes in site-specific environmental data and modelled/measured catchment loading information. There are a variety of advanced univariate and multivariate statistical modelling techniques that can be utilized to test for the existence of temporal trends and to analyse stress-response relationships while accounting for natural temporal cycles.

Recommendation 4. Conduct a comprehensive analysis of patterns of nuisance macroalgae coverage in estuaries, with a focus on (i) trends over time and how they correlate with trends in modelled/measured contaminant loadings from the catchment, (ii) an examination of the role of internal estuarine hydrodynamics, and degree of match between water circulation patterns and the spatial distribution of macroalgae within estuaries, and (iii) other contributing factors such as benthic habitat/substrate types, interactions between turbidity and nutrients, etc.

Recommendation 5. Quality assurance / quality control procedures are needed throughout the programme, including better standardisation of field methods, macroinvertebrate identification checks (e.g., 10% of samples; Hewitt et al. 2015), and regular critical review of technical reports. Independent reviewing of reports could be done at minimal cost via a number of different avenues, for example, reciprocal relationships with Regional Councils or small contracts to experts identified by the Coastal-SIG.

If these recommendations are acted upon, and if our criticisms of existing monitoring methodologies and reporting are addressed, we believe that the Environmental Southland monitoring data can be used to generate scientifically defensible conclusions regarding the role of catchment activities on estuarine health and firmer links between dairy-associated nitrogen loading and the degradation of estuarine receiving bodies.

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Appendix A Detailed review of monitoring sites following visit

7.1 Fortrose Estuary

Fortrose estuary has two sites, A and B, which have been sampled with the fine scale monitoring techniques (Figure A1, Robertson and Stevens 2009). Both of these sites are on the East side of the estuary (Titiroa Stream side).

Site A: is in a relatively high current area near the mouth of the estuary (Figure A1). The sediment was coarse and firm underfoot with pea-gravel covering the surface and evidence of some silt-clay content. *Austrovenus stutchburyi* (cockles) were common and large *Macomona liliana* (wedge Shells) were observed. Juvenile *Paphies australis* (Pipi) were present and *Amphibola crenata* (mud-flat snail) were found on exposed surfaces of sand flats. A salient characteristic was the presence of scattered *Ulva* at this site at the time of inspection (February 2015). The *Ulva* was mobile and ephemeral; with evidence of growth and attachment, detachment and aggregations of tumbling piles that smother the sediment surface. Many of these drift algal mats caused complete anoxia on the sediment surface which was black and sulphurous.

Site B: is in the north eastern corner of the estuary close to the Tokanui-Gorge road (Figure A1). The sediment was muddier compared with Site A. *Amphibola crenata* were in high abundance, bivalves were rare and there was a notable presence of Spionid polychaete tubes at the sediment surface. The site had 10-20% coverage of *Ulva*, which were in long strands rather than in dense mats.

In addition to the two monitored site, other areas of the estuary were surveyed to consider the applicability and suitability of the existing monitoring sites (labelled C-E herein) relative to other locations. Location "C" (Figure A1) was on a large flat on the bank of the Mataura River. This has been recorded as 'soft mud' (Stevens and Robertson 2013), although much of the lower shore is a firm sandflat, with some silt-clay content (Appendix Photos). This areas was characterised by a low *Ulva* content, a low pea gravel content, a notably "wet" non-compacted and well oxygenated sediment surface (sediment still had surface pools <1hr from low tide), a high abundance of tubeworms, and an increasing abundance of crabs moving further towards the top of the shore. High numbers of *Amphibola crenata* were present but bivalves were not common in this area. Location "D" was opposite "C" on the seaward side of the estuary. There was a low and patchy covering of *Ulva* (~5-20%) however, across much of this areas there was only a thin veneer of oxic sediment (5-3mm thick) suggesting this had formerly been covered in *Ulva* or subject to hypoxic or anoxic conditions (Appendix Photos). Further eastwards, location "E" (Figure A1) was covered in dense *Ulva* mats (100% in places) that were pronounced on the lower to middle shore particularly in drainage areas, compared to the upper shore. The *Ulva* in this location appeared to be recently settled as the sediment was well oxygenated. Cockles, pipi and tubeworms were observed here.

Figure A1 **Fortrose estuary. Sites A and B are the existing fine-scale monitoring location of Environment Southland. Locations C-E are other areas that were considered.**

7.2 New River Estuary

New River Estuary has three sites (B, C and D) which have been sampled by fine scale monitoring techniques since ~2001 (Figure A2, Robertson and Stevens 2010). In addition, three eutrophic sites (E, F and W) have been monitored since 2012.

Site B: is in the middle and eastern side of the estuary (Figure A2). The sediment across this large flat is sandy with a low mud content and is covered by ripples (approx. wavelength 5 cm, Wave height 0.25 cm). The monitoring site is located lower down on the shore (~150m from the main harbour channel) and is more wind/wave swept compared with higher up the shore (Plate A1). The monitoring site, and much of the large intertidal flat here has a high density of *Amphibola crenata,* worm casts on the surface (in greater density lower on the shore) and a low density/absence of bivalves. The sediment shows concentrated patches of microphytobenthos (light brown in colour), typically in the troughs of ripples (Plate A1), and a general absence of *Ulva* (only sporadic drift occurrences). There are the remnants of seagrass patches higher up the shore.

Plate A1: Site B lower-shore monitored area (location 46 28 37.000, 168 20 8.999), mid-shore, Site B (location 46 28 36.299, 168 20.530).

<u>**Site D:**</u> is at the top of the central sandflat in New River Estuary, to the west of Bushy Point. The site was visited when partially submerged (3.5hrs after low water). The site appeared to be characteristic of the middle harbour; having firm sand, high-density *Amphibola crenata,* and low density macro-algae. This site was notable for large *Austrovenus stutchburyi*.

<u>**Sites W & F:**</u> are eutrophic sites located in the Waihopai Arm of New River Estuary. This area had thick coverage (near 100%) of macroalgae, predominantly *Gracilaria*. From the channel, moving toward the sites, there was a 200m strip where the *Gracilaria* was in good condition. However, moving shoreward beyond this, the macroalgae was in a state of decomposition, with anoxic conditions prevalent (black sediments below the surface few mm, sulphurous scents, presence of *Beggiatoa* etc). The sediment across this area and at the monitoring sites was deep mud (typically >30 cm deep). Few invertebrates were observed: a few *Amphibola crenata*, crabs and amphipods in the pools of surface water. However, birdlife was notable.

<u>**Sites C & E:**</u> are in Daffodil Bay on the western side of New River Estuary. The Fine-monitoring site (C) is characterised by large cockles, surface shellhash, high density *Amphibola crenata,* and a low coverage of *Ulva* (~5-10%). The eutrophic site (E) is 200m north-west of Site C, and is situated within a dense macroalgae area (100% coverage). Site E is on a raised area, probably due to the trapping of fine sediment by the macroalgae, and extremely muddy. Further up shore of the site, there are dead zones of algae.

<u>**Bushy Point:**</u> This point at the southern tip of the Waihopai Arm has been monitored for sedimentation rate. This area was low in *Gracilaria* and *Ulva* coverage (1-5%), but was located only ~200m away from the dense macroalgae beds to the north. The sediment had a high density of *Amphibola crenata* and other gastropods, with mud crabs and large bivalves present higher up the shore. At the Bushy Point Site, the two posts marking out the sedimentation plates showed high degrees of scour. There was a notable depression between these posts (at the mid-point where the plate is located) and large aggregations or macroalgae wrapped around each.

Figure A2 **New River Estuary. Sites B, C & D have been subjected to fine-scale monitoring. E, F & W are eutrophic sites and have also been subjected to fine-scale monitoring.**

7.3 Waikawa Estuary

Waikawa Estuary has two sites (A and B) where fine scale monitoring has occurred since 2005 (Figure 3, Robertson and Stevens 2013). These sites are in the central basin of the estuary (Figure A3). Site UN in the upper estuary is used for the measurement of sedimentation rates only.

<u>Site UN:</u> The Upper North Site is in a muddy area and has soft underfoot conditions (Figure A3). The sediment surface is characterised by a high density of *Amphibola crenata* (~20 m^{-2}) and a low density of mud crabs. The surface sediment had ripples features (Plate A2) without visible microphytobenthos and the aRDP was around 4 cm deep. The upper area had a low density of *Gracilaria* (~1-5 %) that was typically attached to live cockle shells.

<u>Site B:</u> is immediately adjacent to the estuarine channel; due east from the Waikawa township (Figure 3). From a visual assessment the diversity of Site B was high with large bivalves (*Macomona*, *Austrovenus* in high density) and organisms associated with their shells e.g. barnacles, anenomes (*Antheopluera auroradiata*), limpets (*Notoacmea*). Polychaete worms (maldanids, spionids) and gastropods (*Diloma*, *Cominella*) were also common. Surface sediment showed prominent ripple characteristics (5 cm wave length, 1 cm wave height) and a low density of *Gracilaria* (<1%). The sediment aRDP was thick (>2cm) and *Amphibola crenata* were rare along this section of shore. Nearby to the monitoring site there were areas of raised sediment where the density of organisms was lower; notably bivalves and whelks.

<u>Site A:</u> is in the central section of the harbour, close to the transition into muddier sediment (Figure A3) and was noticeably muddier than Site B. It is relatively low on the shore (near the MLWS channel edge) and lacked ripple features on its surface. The site was heterogeneous and could be separated into two sides across the 60 m x 30 m area: The north-western side of the site was muddier and had a lower density of macroalgae compared with the south-eastern side (Plate A3). Macroalgae were commonly attached to cockle shells.

<u>Plate A2: The Upper North site</u>

<u>Plate A3: The North-west side of Site A (left) and the South-East side (right)</u>

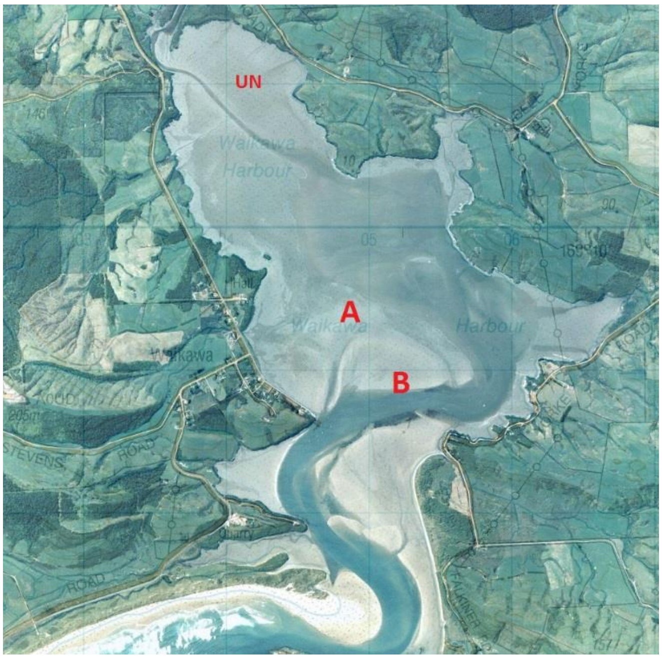

Figure A3 **Waikawa Estuary with fine scale monitoring Sites A and B, and the sedimentation plate site UN (upper North) in the upper estuary.**

7.4 Jacobs River Estuary

There are three sites in Jacobs River Estuary (A, B and C) that have had fine scale monitoring since 2003 (Figure A4, Robertson and Stevens 2013). Two of these sites are in the central basin of the estuary (Figure A4) and the third site is in the Pourakino Arm. Jacobs River Estuary also has two Eutrophic Sites (D and E), which are located in the Pourakino Arm and the Northern Flats of the Aparima River Arm, respectively.

Site A: is in the Central Basin of the estuary. The sediment at this site was a firm dark sand. The sediment surface was dominated by ripple features (~5 cm WL, 1cm WH) (Plate A4). The sediment surface also contained burrow depressions and mounds (likely belonging to *Biffarius filholi*) and there were large *Austrovenus stutchburyi* and *Macomona liliana* present (~50mm shell length). The Site had a low density of *Gracilaria* that was attached primarily to cockle shells.

Plate A4: Sediment at Site A

Site B: is located in the middle of the southern flats. The site is comprised of a uniform firm sand with ripple features (~6 cm WL, 0.5 cm WH) and has a lots of small patches of *Ulva* (5-10% coverage). The sediment is well oxygenated (aRDP ~1-2 cm) and there are indications of high biological activity from the sediment surface. Taxa here include polychaetes worms, a low density of cockles, a high density of *Amphibola crenata* and a moderate density of *Macomona liliana*.

Site C: is on a relatively firm, *Amphibola crenata* dominated sandflat in the Pourakino Arm. There is a gradient across the site from sand to mud: On the eastern side of the site there is a mobile sand bank that intrudes into the side by ~8 m (across the 30 m width). This sand bank is unvegetated and raised above the rest of the site (Plate A5). The sediment of this bank is an iron rich black sand, is well drained and lacks *Amphibola crenata*. On the western side of the site there is noticeably muddier section with a higher density of *Amphibola crenata* and mud crabs. In the middle of the sites the sediment is less muddy and has a moderate density of *Amphibola crenata*.

Plate A5: Sediment in the Pourakino Arm

Site D: is near Site C on the western side of a drainage channel that separates the eutrophic from the non-eutrophic sides of the Pourakino Arm. This eutrophic side has large swathes of >50% cover and 100% macroalgal cover. Macro-algae at the site is patchy in coverage. Site D appears to be lower than the surrounding flat and numerous drainage channels run through the site. The area around Site D is also covered with small drainage channels. Site D has a slightly lower density of *Gracilaria* compared with the surrounding area in 2015 (this likely varies year to year). The sediment surface is oxygenated although the aRDP is thin.

Site E: The lower section of the Northern flat is clear from dense macroalgae (~1-5% of filamentous *Ulva*). Near the main Aparima River channel the substrate is coarse with a gravel content. The *Amphibola crenata* were of mixed sizes (adults and juveniles, ~40 per m^2) and there were fewer egg cases than observed in the other estuaries. Moving towards Site E the sediment becomes increasingly muddy and the Northern flat becomes dominated by red filamentous *Gracilaria*. Here the macroalgae forms a dense turf that is re-engineering the landscape and results in elevated muddy hummocks holding surface water. The biota here is characterised by mud crabs and fewer, smaller, *Amphibola crenata*. This northern section is not homogenous, and there are still many patches of relatively firm sandy sediment (particularly in the channels). The monitoring site is characterised by high macroalgal coverage, shallow RDP (<0.5 cm) and high mud content.

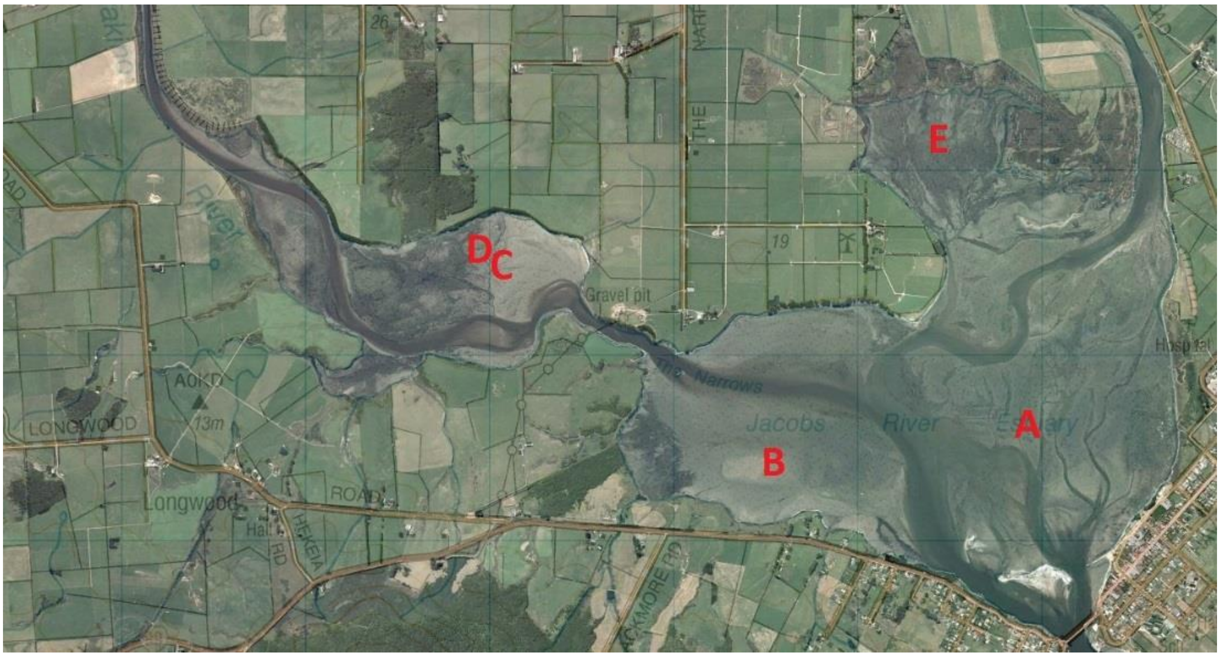

Figure A4: Jacobs River Estuary with fine scale monitoring Sites A, B and C, and the eutrophic sites D and E.