Estuaries and Coast: Classification and Attributes for Southland

Values and Objectives Technical Report

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Executive Summary

Environment Southland and Te Ao Marama (TAMI) have, through their People Water and Land programme, embarked on a community-involved process to further develop the approach to managing land and water in the region. This has included community engagement to support the development of community values and freshwater objectives, and the formation of Regional Forum to help develop limits and both regulatory and non-regulatory methods to achieve them.

The purpose of this report is to contribute to the process of developing draft freshwater objectives for consideration by Environment Southland's Council and the Te Ao Marama board. This report is one of a number of supplementary reports and memos that contribute to the report titled: *Developing Draft Freshwater Objectives for Southland* (Norton and Wilson, 2019) and: *Current Environmental State and the "Gap" to Draft Freshwater Objectives for Southland* (Norton et. al., 2019).

This report explains:

- The rationale for the **estuary classification** proposed to be used in developing freshwater objectives in Southland:
- A description of the **estuary and coast attributes** proposed to be used for numeric freshwater objectives, to support the values of 'Human Health for Recreation' and 'Ecosystem Health', along with the associated attribute state tables; and,
- A summary of the tabulation of data used to **assess attribute state** for 2010, 2016 and 2019 (current state).

Once the process for establishing freshwater objectives for Southland has been completed, it is recommended that a review of the Southland estuary monitoring programme be undertaken to ensure it aligns with any changes in approach to the management of land and water in the region.

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1 Introduction

Environment Southland and Te Ao Marama (TAMI) have, through their People, Water and Land – Te Mana o te Tangata, te Wai, te Whenua programme, embarked on a community-involved process to further develop the approach to managing land and water in the region. This has included community engagement to support the development of community values and freshwater objectives, and the formation of Regional Forum to help develop limits and both regulatory and non-regulatory methods to achieve them.

The People, Water and Land programme has three workstreams, one of which is 'Values and Objectives'. The objective of this workstream is to raise awareness of freshwater and to determine the community's values and freshwater objectives in accordance with the requirements of the National Policy Statement for Freshwater Management[1] (NSPFM). The outputs from this workstream are key components for the other two workstreams: the 'Regional Forum' workstream which is a community group providing Council and Te Ao Marama board members on methods and timeframes to achieve the communities' aspirations for freshwater, and the 'Action on the Ground' workstream whose goal is to enable and support change at the farm-to-catchment scale.

This report is part of a package of work being prepared through the Values and Objectives workstream. Specifically, this report is part of supplementary material that has been produced to contribute to the reports titled: *Developing Draft Freshwater Objectives for Southland* (Norton and Wilson, 2019) and: *Current Environmental State and the "Gap" to Draft Freshwater Objectives for Southland* (Norton et. al., 2019).

1.1 Report purpose

The purpose of this report is to describe the technical basis for the estuary and coast components used in Norton and Wilson (2019) and Norton *et. al,* (2019). This work covers three key areas:

1. **Estuary classification:** The rationale and description of the estuary classes proposed to be used in developing freshwater objectives in Southland. Currently, no classification has been adopted in either the proposed Southland Water and Land Plan (pSWLP) or the Regional Coastal Plan for Southland (RCPS):

2. **Attributes:** In addition to specifying national compulsory attributes, the NPSFM requires councils to develop attributes appropriate for their region to use when setting freshwater objectives. Currently, there are no national compulsory attributes for estuaries or coastal waters influenced by freshwater in the NPSFM. This report describes how estuary attributes have been selected to help describe the values of 'Human health for recreation' and 'Ecosystem health' (see Wilson et. al., 2019) [2], and

[1] The NPSFM was first released in 2011 and amended in 2014 and 2017. Unless otherwise stated, this report refers to the 2017 version of the NPSFM. The NPSFM was further amended in 2020, after this report was first prepared.

[2] The attributes used in this report are likely to support additional values, such as threatened species and mahinga kai (both introduced as national compulsory values in the NPSFM 2020), however attribute applicability to other values, and additional attributes that may be required to support other values, have not been considered in this report. This may be part of further work considering the implications of the NPSFM 2020 to the Values and Objectives package prepared under the NPSFM 2017.

how attribute state tables have been developed to support the setting of freshwater objectives for estuarine and coastal waters (see Norton and Wilson, 2019).

3. **Assessing attribute state**: The NPSFM requires regional councils to 'maintain or improve' water quality in their regions. Therefore, as part of the process for developing draft freshwater objectives, an assessment of attribute state was required. This report explains the data used and how it has been tabulated to inform attribute state for three time periods: 2010, 2016 and 2019 (current state) (see Norton et. al., 2019).

1.2 Where this report fits in the process

This report is part of supplementary material that has been prepared for two key reports in the 'Values and Objectives' workstream in the People, Water and Land programme:

- Estuary classification, estuary and coastal attributes and attribute state tables were used in the *Developing Draft Freshwater Objectives for Southland* report (Norton and Wilson, 2019) and;
- Assessment of attribute state for three time periods were used in the *Current Environmental State and the "Gap" to Draft Freshwater Objectives for Southland* report (Norton et. al., 2019).

It is noted that these reports in turn were used in the third iteration of the draft freshwater objectives, where the above community workstream was woven together with the iwi values and objectives work, to derive a combined set of draft freshwater objectives that provide for hauora, the health and well-being of waterbodies in Southland Murihiku (hauora being a requirement of Te Mana o te Wai in the NPSFM) (Bartlett et. al., 2020).

2 Estuary classification

Currently, there are no classification of estuaries in either the pSWLP (Decisions version, April 2018) or the Regional Coastal Plan for Southland (RCPS). Therefore, two national frameworks were considered for grouping estuaries into classes (typologies). These are:

- NZ Coastal Hydrosystems Typology (NZCHT) (Hume et al., 2016), and
- Estuary trophic index (ETI) typologies (Robertson et al., 2015)

The NZCHT has characterised estuaries within a relatively complex classification of twelve main types (and multiple (>12) subtypes) based on broad physical (geomorphic) features (NZ Coastal Hydrosystems Typology (NZCHT) - Hume et al., 2016). Susceptibility to eutrophication spans multiple geomorphic categories so applying a geomorphic class becomes unnecessarily complex. Susceptibility of estuaries to eutrophication is more directly influenced by specific physical modifying characteristics including dilution, flushing, residence time, depth and intertidal extent. As such it is recommended the classification presented in the Estuary Trophic Index is more suitable for the development of objectives to maintain or improve estuary health.

2.1 Estuary Trophic Index classification

The approach adopted in the Estuary Trophic Index (ETI) has used overseas approaches where they meet NZ conditions e.g. the US ASSETS framework (Bricker et al., 1999, 2003, 2007), the NSW ICOLLs approach (Haines et al., 2006) and ASSETS/DIPSIR approach used on Basque Estuaries (Borja et al., 2006). Background information on these approaches is presented in Tool 1 Appendix 1 of Robertson et al., 2015. However, because the majority of NZ estuaries fall outside of the types used to develop the overseas assessment procedures, the overseas approaches have, in many cases, been modified to better suit the physical characteristics of NZ estuaries.

This class system is designed to provide a relatively robust and cost effective approach to enable the prioritisation of estuaries for more rigorous monitoring and management. It applies a desktop susceptibility approach that is based on estuary physical characteristics for key NZ estuary classes. The ETI tools enable a score to be derived for the estuary for likely or known condition once catchment pressures are considered. For estuaries in Southland only the classification is currently being adopted as this is based on natural characteristics of the estuary that categorise according to the physical susceptibility (i.e. very high, high, moderate, low susceptibility to eutrophication).
The resultant ETI class system is a simple 4 category classification system specifically suited to NZ estuaries. The classes are as follows:

- Intermittently closed/open lakes and lagoons estuaries (ICOLLs)
- Shallow intertidal dominated estuaries (SIDEs)
- Shallow, short residence time tidal river and tidal river with adjoining lagoon estuaries (SSRTREs)
- Deeper subtidal dominated, longer residence time estuaries (DSDEs)

Descriptions of the different estuary classes have been reproduced from Robertson et al., 2016a. They are described below in sections 2.1.1 to 2.1.5.

2.1.1 Intermittently Closed/Open Lake and Lagoon Estuaries (ICOLLs)

Shallow tidal lagoon and tidal river type estuaries (<3m deep) that experience periodical mouth closure or constriction (called ICOLLs) have the highest susceptibility to nutrient retention and eutrophication, with the most susceptible being those with closure periods of months (e.g. Waituna Lagoon) rather than days (e.g. Lake Onoke).

ICOLLs experience "Very High" susceptibility to nutrient loads during periods of closure. Under these conditions these systems behave similar to lakes. As such for the purpose of the objective setting process ICOLLS have been included in the lakes work stream and are classified under Brackish Lakes and Lagoons (Robertson and Ward, 2020). In Southland, Robertson et al., 2015 included Lake Brunton and Waituna Lagoon under the ICOLL classification. For the purpose of the objective setting process, it is proposed these two systems are classified as Brackish Lakes and Lagoons and included with the development of attributes and objectives for lakes.

Susceptibility to Nutrient Loads:	Very High
Major Primary Producers:	Macroalgae and phytoplankton

2.1.2 Shallow, Intertidal Dominated Estuaries (SIDEs)

For NZ's dominant estuary classes (i.e. shallow, short residence time (<3 days), and predominantly intertidal, tidal lagoon estuaries and parts of other estuary classes where extensive tidal flats exist e.g. Firth of Thames, Kaipara Harbour, Freshwater Estuary), flushing is too strong for significant retention of dissolved nutrients. Nevertheless, retention can still be sufficient to allow for retention of fine sediment and nutrients (particularly if these are excessive), deleterious for healthy growths of seagrass and saltmarsh, and nuisance growths of macroalgae in at-risk habitat. In these latter estuary classes, assessment of the susceptibility to eutrophication must focus on the quantification of at-risk habitat (generally upper estuary tidal flats), based on the assumption that the risk of eutrophication symptoms increases as the habitat that is vulnerable to eutrophication symptoms expands. Nitrogen has been identified as the element most limiting to algal production in most estuaries in the temperate zone and is therefore the preferred target for eutrophication management in these estuaries (Howarth and Marino 2006).

Susceptibility to Nutrient Loads:	Moderate to High
Major Primary Producers:	Macroalgae

2.1.3 Shallow, Short Residence Time Tidal River, and Tidal River with Adjoining Lagoon Estuaries (SSRTREs)

NZ also has a number of shallow, short residence time (<3 days) tidal river estuaries (including those that exit via a very well-flushed small lagoon) that have such a large flushing potential (freshwater inflow/estuary volume ratio >0.16) that the majority of fine sediment and nutrients are exported to the sea. In general, these estuary classes have extremely low susceptibilities and can often tolerate nutrient loads an order of magnitude greater than shallow, intertidal dominated estuaries. These shallow estuary classes are generally nitrogen limited.

Susceptibility to Nutrient Loads:	Low to Very Low
Major Primary Producers:	Macroalgae, but low production, especially if freshwater inflow high.

2.1.4 Deeper, Subtidal Dominated Estuaries (DSDEs)

Mainly subtidal, moderately deep (>3m to 15m mean depth) coastal embayments (e.g. Firth of Thames) and tidal lagoon estuaries (e.g. Otago Harbour), with moderate residence times >7 to 60 days) can exhibit both sustained phytoplankton blooms, and nuisance growths of opportunistic macroalgae (especially *Ulva* sp. and *Gracilaria* sp.) if nutrient loads are excessive. The latter are usually evident particularly on muddy intertidal flats near river mouths and in the water column where water clarity allows. Deeper, long residence time embayments and fiords are primarily phytoplankton dominated if nutrient loads are excessive. Outer reaches of such systems which sustain vertical density stratification can be susceptible to oxygen depletion and low pH effects (Sunda and Cai 2012, Zeldis et al., 2015). In both cases, it is expected that the US ASSETS approach will adequately predict their trophic state susceptibility. These deeper estuary classes are generally nitrogen limited.

Susceptibility to Nutrient Loads:	Moderate to Low
Major Primary Producers:	Macroalgae (moderately deep) and phytoplankton (deeper sections).

Table 1: Summary of estuary class key features from Robertson et. al., 2015.

Broad Estuary Class		
SIDE	**SSRTRE**	**DSDE**
Lagoon shape	Tidal River shape	Varied shape
Shallow (<3m deep)	Shallow (<3m deep)	Deep (>3m)
>40% of estuary is intertidal	High Flushing Potential[2]	-
Generally short residence time[1] (<3 days)	Generally short residence time[1] (<3 days)	Generally long residence time[1] (<7 days)

[1]Residence time is the time a water parcel spends in the system, this is important for nutrients as the greater time spent in the system the greater the chance of interaction and ecological response.

[2]Flushing potential is the degree to which nutrients are flushed out of the estuary by freshwater inflow. For a given tidal range, a low freshwater inflow/estuary volume ratio indicates high retention of catchment nutrient inputs and a high susceptibility to eutrophication.

2.1.5 Natural state

In addition to the ETI classification proposed, a 'natural state' class has been included. The natural state class is defined in the current pSWLP as, Waters within:

- areas defined as National Park managed under the National Parks Act 1980 (including land for the time being administered as if it was a national park pursuant to any statute or written agreement with the owners); and
- public conservation land managed under the Conservation Act 1987 and the Reserves Act 1977 as detailed in Table 1 "Natural State Waters outside National Parks" in Appendix I "Natural State Waters outside National Parks" of this Plan.

2.2 Uncertainty for estuary classes

This classification system and associated ETI guidance (Robertson et al., 2015, 2016) broadly categorises systems. However, there will be differences between types and within types, some of which may be relevant to key ecological processes and their functions. The intention of the

ETI package is to provide a robust assessment of eutrophication for most NZ estuary classes, and to provide preliminary, screening-level, nutrient threshold guidance.

For establishing nutrient thresholds, the ETI recommends the use of more robust approaches; preferably relevant measured nutrient load/ecological response gradients, but if unavailable, using the modelling approaches it describes. This may include further spatial delineation within larger and/or more complex estuaries (Cornelisen et al., 2017). The work programme needed in order to improve these methods is likely to be costly, time consuming and needs to be fit for purpose (Ward and Roberts, 2018).

2.3 Methodology for mapping of estuaries and classes

The mapping exercise has produced a geographical information system (GIS) spatial layer that identifies the location, type and source of information to determine estuary class. The following steps were undertaken to map Southland estuaries:

1. Extraction from online Estuary Trophic Index tool. This identifies some estuaries but is not a comprehensive list. For those identified an estuary class is allocated. The ETI tool only supplies location coordinates for a given estuary so polygons were drawn around systems using aerial images or using existing GIS spatial layers, where possible. The ETI is based on NIWA's coastal explorer database. Note that this database excludes hundreds of small tidal river estuaries, many of which have intermittently closed/open mouths and have dilution potentials >10-9. For the purposes of mapping, it was assumed that all estuaries were not significantly salinity stratified, which is probably true for all Southland estuaries with the exception of the fiords, some embayments and small areas in the upper reaches of other estuaries. Note mapping of the ICOLL class was considered in the Lakes and Lagoons work stream.
2. Additional estuaries and their classes were identified in the Robertson and Stevens (2008) study. This list was used to compare against the ETI, estuaries missing were mapped in the GIS spatial layer and classified.
3. For estuaries not captured in steps 1 and 2, aerial photography was used to draw polygons in GIS in conjunction with LINZ nautical charts. The charts helped to delineate the extent of an estuary using the plotted sediment extent. The coastal extent was somewhat arbitrary by drawing a line between headlands. Names were derived from LINZ topography maps.
4. For some estuaries the class could not be determined. Most of these are in national park areas and therefore will be classed under the 'natural state' class.
5. An exception in the classification process and mapping was the Waiau River estuary which is discussed in further detail in section "Waiau Lagoon and Waiau estuary"

The mapping methodology is described further in a report titled: *Mapping of water body management classes for Values and Objectives* (Wilson and Darragh, 2020). A rendition of the mapped estuary classes is found in Figure 1.

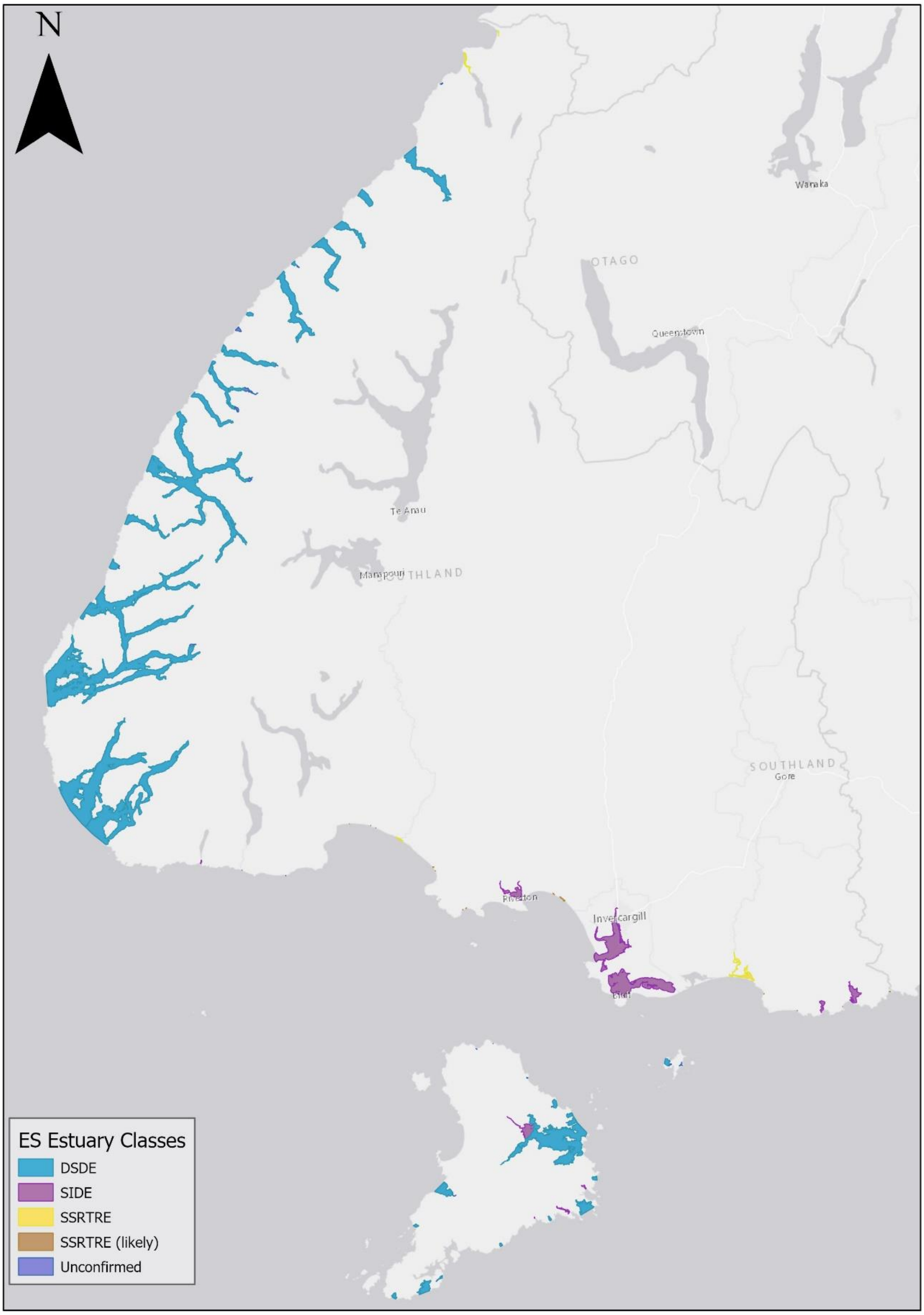

Figure 1: Map of estuary classes for Southland as of March 2020.

2.4 Te Waewae Lagoon and Waiau Estuary

The ETI classification system and associated ETI guidance (Robertson et al., 2016a and b) classifies 'Waiau River estuary' as an SSRTRE. This is assumingly based on a reference from Kirk and Lauder (2000; Figure 2), which describes the system as a River Mouth lagoon (Hapua) morphology, with a residence time is likely to be very short i.e. <1 day. Indeed, work conducted on classifying coastal hydro systems report (Hume et al. 2016) perpetuates the Kirk and Lauder reference which actually relies on an even earlier reference in 1991.

A key characteristic of Hapua type systems is that they change morphology through time. The gravel bar since 1991 has since changed and restricted river flow directly through the system, the river now flows out to the West of main river channel. A lagoon has developed at the eastern end of the estuary where flow exchange is restricted through a small narrow channel at the river mouth. The river flow and tidal exchange to the lagoon remains uncertain. However, the residence time of the lagoon has increased since the classification of the estuary in Kirk and Lauder (2000).

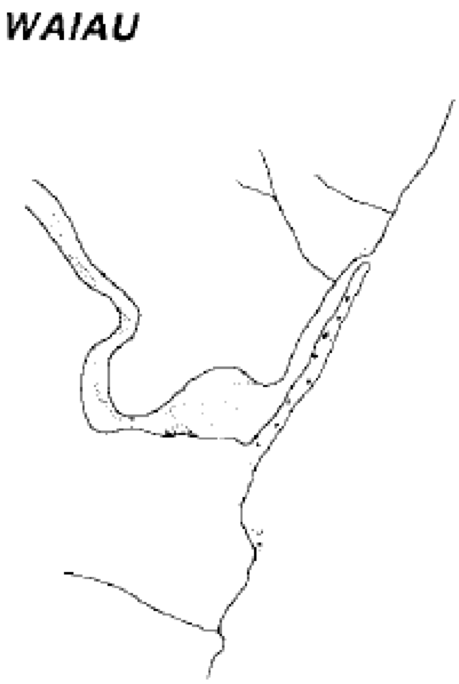

Figure 2: Waiau River Estuary morphology according to Kirk and Lauder (2000). Waiau River was flowing out the east end so residence time was less.

As a result, there has been a large shift in the morphology of the Waiau River estuary since its original classification. Taking this into consideration for the objective setting process the Waiau River estuary has been separated into two parts (Figure 3) because it broadly behaves as two unique systems:

- the eastern lagoon is classified as a brackish lake or lagoon and is discussed further in the Southland Lakes: classification, attributes and state assessment (2020) and;
- the western end is classified as a tidal river estuary (most likely SSRTRE) which is more heavily influenced by salinity and tides near the opening to the coast. The river force at that end results in low water residence time.

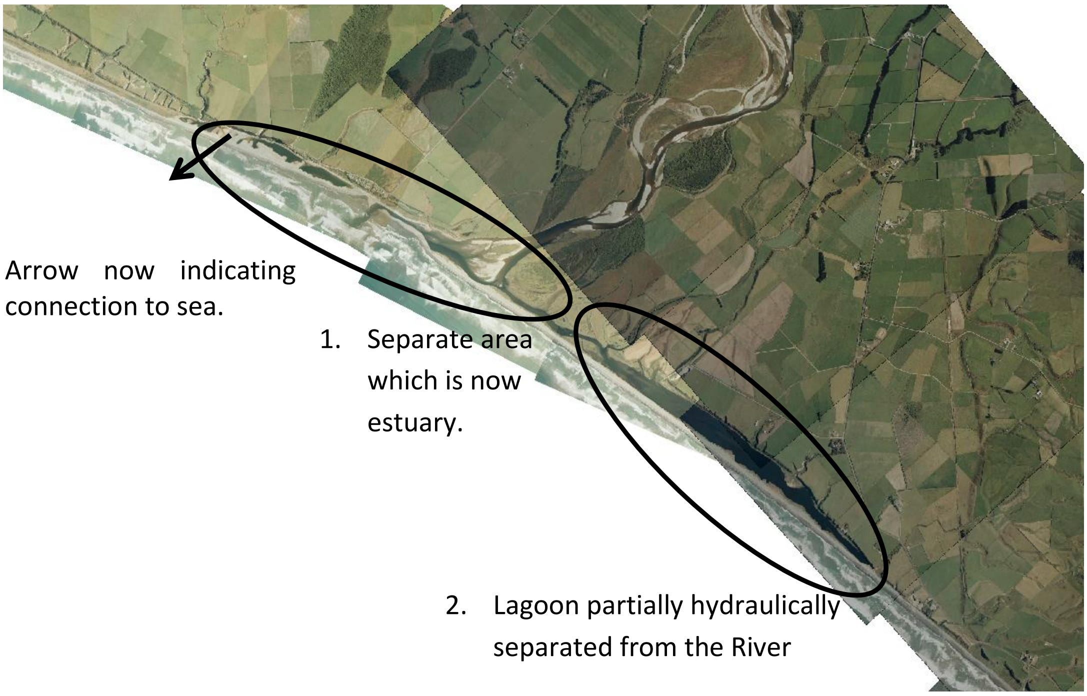

Figure 3: Waiau River now (2018) flows out to the west not the east end as described in Kirk and Lauder (2000). North is up in the photo.

3 Estuary attribute selection process

In 2017, a useful piece of work called '*Managing Upstream: Estuaries State and Values'* (Cornelisen et al., 2017) was commissioned by the Ministry of Environment (MfE) to consider the wider perspectives of NPSFM (2014 version) for the estuarine environment and their potential attributes. The Cornelisen et al., (2017) working group focused on three values of national relevance identified by MfE that apply across all estuaries, namely: ecosystem health, human health for recreation, and mahinga kai. Conceptually this is shown in Figure 4**.**

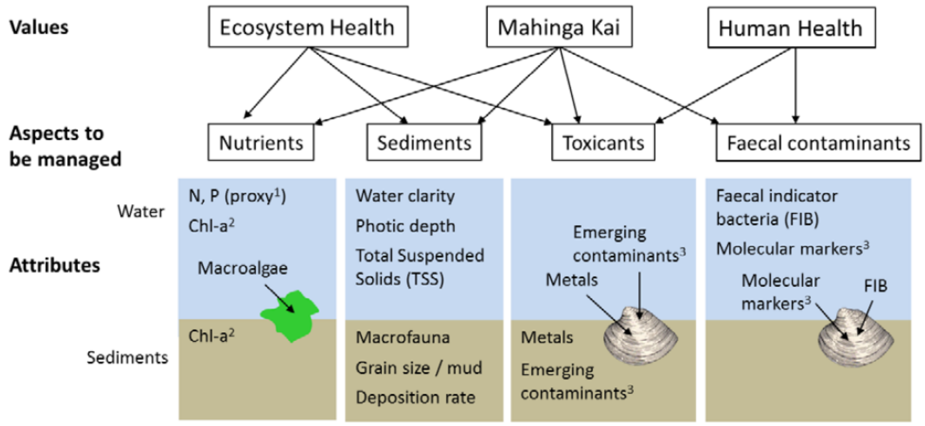

Figure 4: Conceptual diagram of recommended attributes from Cornelisen et al., (2017).

[1]For nutrients such as nitrogen (N) and phosphorus (P), a proxy, such as modelled potential nutrient concentrations may be used. [2]Chlorophyll-a is a proxy for phytoplankton in the water and microphytobenthos (small algae) in the sediments. [3]The inclusion of emerging contaminants and molecular markers for faecal bacteria and pathogens is intended to mark their potential future role in managing and monitoring estuaries following further research and development.

The Cornelisen et al., (2017) work involved a range of estuarine specialists from across the country to identify potential attributes for estuaries. According to Cornelisen et al., (2017) variables with the greatest potential for development into attributes can: link to the values; be manageable through freshwater inputs; be measurable and predictable; and be used to set management objectives. The rationale for identifying aspects to be managed is that estuaries can also be affected by other pressures that are not upstream (e.g. invasive species, fishing, dredging, aquaculture, shoreline armouring, flap gates, and wildlife disturbance), as well as the surrounding ocean (e.g., fishing and climate related changes including sea level rise, temperature, and ocean acidification).

Cornelisen et al., (2017) differentiates between the 'attributes' and 'state variables'; one being directly linked to aspects to be managed upstream and the other provide information about condition of a value. However, the NPSFM (2017 version) does not make this distinction so the term 'attributes' has been used subsequently for use in setting freshwater objectives and/or limits in the regional forum process.

The variables identified by Cornelisen et al., (2017) for further consideration have been collated into Appendix A.

Note that consideration of mahinga kai was outside field of expertise of the contributors to Cornelisen et al., (2017), as it is the authors of this report.

3.1 Attribute selection sequence

The sequence for attribute development has been developed as follows:

i. A list of possible attributes for estuaries was identified using the managing upstream project (Cornelisen et al., 2017) and additional estuary monitoring information (Robertson et al., 2016 and Robertson et al., 2017b). This is presented in Appendix A.
ii. The list of possible attributes was then short-listed to those which had data available for Southland estuaries (Appendix A).
iii. The short-list was then further refined, as shown in Appendix B. Recommendations are argued, and where an attribute ends up being "proposed", a corresponding attribute table is presented. The specific data requirements and data used to calculate state against these attributes has been collated into Appendix C. The attributes have also been considered for their applicability to the open coast i.e. saline areas outside of estuaries.
iv. The final short list of proposed attributes is summarised in Table 2. These attributes are described and considered in more detail in Appendix B and the specific data requirements and data used to calculate current state using these attributes has been collated into Appendix C.

3.2 Attribute selection criteria

An attribute has been recommended for adoption if it:

a) describes the values of ecosystem health or human health for recreation;
b) is supported by local monitoring that allow bands or thresholds to be identified;
c) there are not insurmountable measurement problems and;
d) the attribute is expected to respond to management.

Note that mahinga kai was not considered as the authors consider this field outside their expertise area. Further development of mahinga kai specific attributes and the consideration of proposed ones for mahinga kai suitability will therefore be needed.

Table 2: Summary of short-listed attributes with associated recommendation for use as numeric freshwater objectives in Southland

Aspect affected	Number	Attribute	Class applicability	Aspect to be managed	Recommendation
Sediment quality	1	Sediment organic content	SIDE, SSRTRE	Nutrient enrichment	Needs development
	2	Sediment grain size (includes mud content)/ Composition and areal extent of dominant substrate types	SIDE, SSRTRE	Sediment loading,	Grain size proposed; mud extent possibly as narrative
	3	Sediment nutrient concentrations (N, P, C)	SIDE, SSRTRE	Nutrient enrichment	Needs development
	4	Depth of RPD (REDOX Potential Discontinuity) in sediments / Areal extent of hypoxic/anoxic bottoms	SIDE, SSRTRE	Nutrient enrichment, Sediment loading	Proposed
	5	Inorganic compounds in sediment (metals/metalloids)	SIDE, SSRTRE, DSDE, Open coast	Toxicants (metals/metalloids)	Proposed
Sedimentation	6	Measured sediment deposition	SIDE, SSRTRE	Sediment loading	Proposed
Water Quality	7	Water nutrient concentrations (N, P, C)	SIDE, SSRTRE, DSDE	Nutrient enrichment	Needs development
	8	Water Chlorophyll-*a* (Chlorophyll-*a*)	SIDE, SSRTRE, DSDE, Open coast	Nutrient enrichment	Proposed
Habitat	9	Areal extent of seagrass / Percent cover of seagrass.	SIDE, SSRTRE	Nutrient enrichment,	Proposed
	10	Composition and areal extent of dominant saltmarsh types.	SIDE, SSRTRE	Nutrient enrichment, sediment loading	Narrative
	11	Areal extent of opportunistic macroalgae (measures include EQR calculated from the Opportunistic Macroalgal Blooming Tool)	SIDE, SSRTRE	Nutrient enrichment	Proposed
Macrofauna	12	Biodiversity of macrofauna (measures incl. biodiversity, multivariate indices, trait based index)/ Traits based macrofauna index/ Evenness of macrofauna / Multivariate macrofauna indices	SIDE, SSRTRE	Sediment loading, toxicants (metals/metalloids), nutrient enrichment	Needs development
Combined Index	13	Estuary Trophic Index Score (ETI)	SIDE, SSRTRE, DSDE	Nutrient enrichment, sediment loading	Needs development
	14	Gross Eutrophic Zone [i.e. >50% macroalgal cover and gross eutrophic sediment conditions (mud>25%, aRPD <1cm)]; (ha, % of estuary area)	SIDE, SSRTRE	Nutrient enrichment, sediment loading	Proposed
Faecal Indicator Organisms	15	*E. coli*	SIDE, SSRTRE, DSDE, Open coast	Pathogens loading and enrichment	Proposed
	16	Enterococci	SIDE, SSRTRE, DSDE, Open coast	Pathogens loading and enrichment	Proposed

Summary and recommendations

It is proposed that the following classes are used for Southland estuaries:

- *Tidal lagoon estuaries (SIDEs)*
  SIDE estuaries are characterised by short residence times (less than three days), are shallow and have large areas of the estuary that are intertidal. These estuaries have a moderate to high susceptibility to eutrophication. Examples of SIDEs in Southland are Jacobs River Estuary, New River Estuary, Haldane Estuary and Waikawa Harbour.

- *Tidal river estuaries (SSRTREs)*
  SSRTRE estuaries are shallow, have short residence times (less than three days) and have a high river flushing potential due to river dominance. Because of this high flushing potential from rivers, these estuaries have a low susceptibility to eutrophication. Examples of SSRTREs in Southland are Waimatuku Estuary and Toetoes (Fortrose) Estuary.

- *Fiords and Bays (DSDE)*
  DSDE estuaries are deep, subtidal and have high residence times (greater than seven days). These estuaries generally have moderate to low susceptibility to eutrophication. In Southland, most DSDE estuaries are also in the natural state class below, with examples being Milford Sound/Piopiotahi and Doubtful Sound/Patea.

- *Natural state*
  This class is defined in the pSWLP, and generally refers to estuaries with catchments in national parks and/or conservation land.

It is proposed that ICOLLs are included in the lake classes (see Robertson and Ward, 2020).

It is proposed that the following attributes are used for draft numeric freshwater objectives for estuaries:

- Phytoplankton;
- Sediment oxygen levels;
- Gross eutrophic zone;
- Mud content;
- Sedimentation rate;
- Macroalgae;
- Toxicants in sediment (arsenic, cadmium, chromium, copper, lead, mercury, nickel, zinc);

And the following attributes are used for both estuaries and open coastal waters which are influenced by freshwater:

- *E. coli;*
- *E. coli* at popular bathing sites;
- Enterococci; and,
- Enterococci at popular bathing sites.

Attribute state option tables have been developed for the above short-listed attributes (see Appendix B).

It is recommended further information is required on a number of possible attributes (such as sediment organic content and nutrients in the water column), before they can be considered for use as numeric freshwater objectives.

It is further recommended that that a review of the Southland estuary monitoring programme be undertaken upon completion of the setting of freshwater objectives (and limits) to ensure the monitoring programme aligns with any changes in approach to the management of land and water in the region.

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Appendix A – Data assessment of the potential attributes to determine short list

Table 3: Potential options for attributes assessed against Southland data availability. Note information is summarised from Cornelisen et al., 2017, Robertson et al., 2016 and Robertson et al., 2017b.

#	Variable	Relevance to upstream management	Dependent on estuary class?	Strengths and weaknesses, considerations	Supporting Data for Southland Estuaries
1.	Sediment organic content	Organic content increases with loading of terrestrial organic matter, and in response to nutrient driven algal growth within estuaries, and decomposition leads to more nutrient enriched sediments.	Organic enrichment will vary across different estuary classes (poorly vs highly flushed, shallow vs deep).	Monitor Total Organic Carbon concentration in upper 2cm of sediment. Annually Oct-March. AFDW (ash free dry weight) - a surrogate measure can be converted to TOC, but conversions give highly variable results. Can be cheaply and directly measured as TOC. Spatially variable within estuaries Likely to show non-linear response to nutrient loading. Links to ecosystem health status have been demonstrated.	Y
2.	Sediment grain size (incudes mud content)	Increased loading of fine sediments from catchments can result in an increased proportion of mud in estuary sediments.	Will likely vary among estuaries with differing water depths, flushing, and hydrodynamic characteristics.	Monitor grain size (especially mud) concentration in upper 2cm of sediment. Annually Oct-March. Two different methods typically used (sieving and laser). Previous work has indicated sieving is a more reproducible approach. Grain size has been well documented to correlate with macrofaunal measures/indicators. Spatially variable as a function of hydrodynamics and resuspension. Links to ecosystem health status have been demonstrated.	Y
3.	Composition and areal extent of dominant substrate types	As above for grain size. Terrigenous sediment input into estuaries from catchment activities can alter areal extent of substrate types (e.g., increase the area of the estuary with mud content >20%).	Yes. Will vary between different classes of estuaries, and within a given estuary due to influence of hydrodynamics, estuary morphology, resuspension, transport, etc.	Can be measured but is labour intensive and requires some level of expertise and calibration. Measured in broad scale habitat mapping by an experienced field ecologist, calibrated with grain size samples. Support from literature for relationships with values. Data on mud content routinely collected by Councils and significant databases exist. Some links to ecosystem biodiversity status have been demonstrated. Relates to sediment grain size.	Y
4.	Sediment Chlorophyll-*a*	Increase due to increased benthic productivity in response to nutrient loading from catchments, Decrease due to reduced light availability / sediment resuspension with sediment loading.	Not dependent on estuary class	Easy to measure. Measured as Chlorophyll-*a* extracted from the sediment. Seasonally and spatially variable. Multiple factors can influence; difficult to distinguish between upstream pressures. Links to ecosystem health status have not been demonstrated.	N

#	Variable	Relevance to upstream management	Dependent on estuary class?	Strengths and weaknesses, considerations	Supporting Data for Southland Estuaries
5.	Sediment nutrient concentrations (N, P, C)	Nutrients, and in particular N, P and C, can increase in response to upstream inputs of nutrients, although importance will depend on benthic fauna and flora cycling of nutrients and inputs from other sources (e.g., ocean).	May be more closely linked to upstream pressures in shallow, poorly flushed estuaries than highly flushed ones, or those that have a strong oceanic input of nutrients.	Monitor total nitrogen and total phosphorus concentration in upper 2cm of sediment. Annually Oct-March. Also see TOC. Easy to measure through standard laboratory accredited methods. Measured as total nutrient content (N,P,C) extracted from the sediment. Spatially variable within estuary. Temporally variable with changes in freshwater inflows, nutrient cycling processes, resuspension, etc. Generally considered that historic levels may take a while to change. Links to ecosystem health status have been demonstrated for N.	Y
6.	Depth of RPD (REDOX Potential Discontinuity) in sediments	Linked to nutrient loading and high magnitude sedimentation events. Corresponds with grain size and organic accumulation in sediments, and with sediment hypoxia/anoxia (see table with habitat attributes).	May depend on estuary class in a similar manner as those parameters that relate to oxygenation of the sediments (e.g., grain size, organic content).	Monitor RPD with ORP meter and visually. Use ORP meter with high resolution microelectrodes to calibrate visual method. Annually Oct-March. Currently ORP data of this kind is not available. Can be cost effective if visual method used, although this method does not always correspond with laboratory measures for sediments with high Fe (e.g. many west coast estuaries or black sands in Jacobs River estuary). The visual method provides an integrated measurement whereas the ORP gives a spot measurement. Slightly greater costs the ORP probe and meter in situ can be used. ORP is site specific and highly variable dependent on the sediment condition and time of measurement (e.g. tide time). Can be difficult to separate out effects of nutrients vs sedimentation event. Links to ecosystem health status demonstrated.	Y – almost exclusively aRPD data
7.	Sediment sulphide concentration	Closely correlated with depth of RPD (see above).	Not dependent on estuary class.	Sediment TS and SCr in upper 2 cm of sediment. Annually Oct-March. Usually measured using calibrated probe, which can be difficult to use in the field. Must be analysed within hours of arrival at lab.	N - Partial information, only done in 2015 for Total S.
8.	Micro-phytobenthos (MPB) biomass	See sediment Chlorophyll-*a*, which is a proxy for MPB biomass.	Not dependent on estuary class. Though extent of intertidal area and depth (limiting light penetration) determine the relevance and use. DSDE class estuaries are deeper and have less intertidal area.	MPB biomass requires conversion to Carbon units, which requires knowledge of C/Chlorophyll-*a* ratios that can vary depending on the classes of diatoms, other algae, cyanobacteria making up the MPB. Links to ecosystem health status have not been demonstrated.	N

#	Variable	Relevance to upstream management	Dependent on estuary class?	Strengths and weaknesses, considerations	Supporting Data for Southland Estuaries
9.	Water nutrient concentrations (N, P, C)	Can respond to upstream inputs of nutrients, although importance will depend on benthic fauna and flora cycling of nutrients and inputs from other sources (e.g., ocean).	Will be more closely linked to upstream pressures in shallow, poorly flushed estuaries than highly flushed ones, or those that have a strong oceanic or within estuary inputs.	Monitor total nitrogen and phosphorus in water column at representative sites (may need to include bottom water in stratified situations and dissolved inorganic and organic nutrients). Water column nutrients (TN and TP) - Requires development. Affected by nutrient cycling processes, and fluctuate hourly to seasonally in response to tides and primary production, respectively, therefore requires high-frequency sampling that can be expensive. High spatial and temporal variability, requiring high resolution sampling. In earlier Estuary National objectives Framework, nutrient loads and/or potential concentrations identified as alternative measures.	Y - only New River estuary, WQ report for New River (1991 -2015) - A329715.
10.	Water column dissolved oxygen (DO)	Indirectly related to increased nutrient loadings via rates of primary production and decomposition of organic matter. Under certain conditions bottom water anoxia can also be induced by prolonged salinity stratification e.g. in an estuary with a salt wedge.	Less likely to be an issue in well flushed estuaries. The same bands and bottom lines would likely apply to all estuaries (i.e., anoxia is not a natural feature of NZ estuaries)	DO concentration measured using a field meter. Daily for 7 days in worst case conditions (e.g. loggers), otherwise monthly (may need to include bottom water in stratified situations). Easy to measure with instrumentation, but requires ongoing maintenance High frequency required as it can vary considerably over hours, days, seasons DO is attribute under NPS-FM for use below point sources Measurements are affected by salinity and temperature Links to ecosystem health status have not been demonstrated.	N
11.	Turbidity	Increases with incoming sediments during flood events, and also resuspension of mud and sediments within the estuary, which can occur during wind/wave events.	Shallow, exposed estuaries and deeper estuaries with strong currents will be more susceptible to within estuary generated turbidity (i.e., resuspension of seabed sediments). Poorly flushed and smaller estuaries will have a reduced capacity to dilute and assimilate TSS inputs.	Turbidity is highly variable even on shortest of time scales. A range of measures are available, each with advantages and disadvantages. A key gap that needs to be addressed is how turbidity/light/SSC can be meaningfully monitored. Directly links to ecosystem health status but complicated non-linear responses likely. Relates to Water Clarity	N
12.	Visual clarity (Secchi disc, black disc, turbidity)	Decreases with incoming sediments during flood events, and also resuspension of mud and sediments within the estuary, which can occur during wind/wave events.	Shallow, exposed estuaries and deeper estuaries with strong currents will be more susceptible to within estuary generated suspension (i.e., resuspension of seabed sediments).	Need measures of sufficient frequency to assess variability and correlations with factors other than upstream pressures (e.g., resuspension, colour staining of water due to humic acids from highly forested catchments may effect measurement). There needs to be consideration as to how these measures directly link to ecosystem health status as there are often complicated non-linear responses. Visual clarity and TSS is being investigated as attribute under NPS-FM. It can be easy and cost effective to measure some of these.	N
13.	Suspended sediment (total suspended solids [TSS], suspended sediment concentration [SSC])	Increases with incoming sediments during flood events, and also resuspension of mud and sediments within the estuary, which can occur during wind/wave events.	As above Poorly flushed and smaller estuaries will have a reduced capacity to dilute and assimilate sediment inputs.	As above	N

#	Variable	Relevance to upstream management	Dependent on estuary class?	Strengths and weaknesses, considerations	Supporting Data for Southland Estuaries
14.	Light attenuation/penetration (Secchi disc, black disc, turbidity)	Attenuation increases with incoming sediments during flood events, and also resuspension of mud and sediments within the estuary, which can occur during wind/wave events.	As above	As above	N
15.	Water Chlorophyll-*a* (Chlorophyll-*a*)	Chlorophyll-*a* can increase with increased nutrient loading from upstream sources; it is one of the symptoms of eutrophication. Chlorophyll-*a* can respond negatively to increased sediment loading due to lower light levels.	Yes: estuaries with reduced tidal flushing will be more susceptible to measurable changes in Chlorophyll-*a*, whereas, more open, frequently flushed estuaries will not. However, primary production in the water column is important in deeper water estuaries.	Mean monthly from representative areas of estuary water column. 90th percentile based on monthly measures. Split into 2 types: [Oligohaline 0.5-5ppt salinity, Mesohaline >5-18ppt, Polyhaline >18-30ppt] and Euhaline>30ppt. Under the NPS-FM, Chlorophyll-*a* is proxy for periphyton (rivers) and phytoplankton (lakes) attributes. In oligotrophic systems, a non-linear response would be expected. Difficult to separate out response to different stressors. Spatially and temporally variable within estuary. Links to ecosystem health status have not been demonstrated.	Y - only New River estuary, WQ report for New River (1991 -2015) - A329715.
16.	Areal extent of seagrass	Seagrass (*Zostera muelleri*) is vulnerable to excessive nutrients and increased turbidity in the water column that can lead to light limitation and in some instances smothering by deposited sediment. Excessive macroalgal growths associated with nutrient loading can smother seagrass.	Yes. Generally, inhabits intertidal and shallow soft sediment areas, so is more likely to be dominant in shallow tidally flushed estuaries with tidal flats, and restricted to shallow margins.	Intertidal aquatic vegetation (including Seagrass) areal distribution, % cover, density, epiphyte load. Annual Oct-March, follow broad scale habitat mapping approach. SAV (Seagrass) Extent (% of ENSC Estimated natural seagrass cover). Easy to measure with aerial photography and ground truthing (broad scale mapping). Long term cycles of expansion and contraction of sea grass bed area have been recorded in northern estuaries. This is not well understood but could affect the interpretation of changes in cover. Likely to be non-linear response. Difficult to separate response to different stressors. Links to ecosystem health status have not been demonstrated.	Y
16.	Areal extent of hypoxic/anoxic bottoms	Excessive nutrient and sediment loading from upstream sources can both contribute to hypoxic/anoxic conditions within estuarine sediments. In some subtidal areas saline stratification can also induce hypoxia/ anoxia.	Yes. Hypoxia can be dependent on flushing potential of the estuary e.g. stratified or high nutrient water is removed or short lived in the estuary.	Direct link to ecosystem health status at high levels. Relates to several other variables, including depth of sediment RPD (Table 4-1), water column DO (Table 4-2), and also pore water DO. Relates to Depth of RPD.	Y – limited information on spatial extent of aRPD in Southland estuaries
17.	Areal extent of shellfish beds	Distribution of shellfish beds are influenced by sediment grain size characteristics, which is influenced by sediment loading and increased muddiness. Also affected by depositional events.	Yes. Different shellfish species will have different lifestyles. Some exclusive to intertidal areas and others to deeper waters.	Need to understand cultural and recreational pressures, conversely these aspects can be used to allow citizen monitoring. Direct link to ecosystem health.	N

#	Variable	Relevance to upstream management	Dependent on estuary class?	Strengths and weaknesses, considerations	Supporting Data for Southland Estuaries
18.	Composition and areal extent of dominant saltmarsh types.	Primary impact on saltmarshes is habitat alteration, engineering affecting flow rates and coastal development. Many estuaries have undergone extensive modification, particularly where land has been drained and developed, and/or hydrology compromised (e.g., by roads). Sea level rise and sedimentation rate also have an influence.	Yes	% cover, dominant species, presence of weeds. No significant decline in cover from established baseline. Easy to measure with aerial photography and ground truthing (broad scale mapping). Historical changes have been highly variable between estuaries; would likely need to set a rating based on change from present state. Likely to be affected by invasions.	Y
19.	Areal extent of opportunistic macroalgae (measures include EQR calculated from the Opportunistic Macroalgal Blooming Tool)	Can be directly a function of nutrient loading from catchments, particularly in cases where the downstream estuary is poorly flushed, and increased nutrients can result in blooms of nuisance macroalgae (e.g., *Ulva spp*., *Gracilaria spp*.). Alterations in estuary bathymetry, sediments (increased muddiness) and hydrology may influence macroalgae growth. Macroalgal growth can in turn influence bathymetry by stabilising and trapping fine sediments.	Yes – intertidal estuaries. SIDE and SSRTRE. Macroalgal blooms most likely to be associated with estuaries with high nutrient loading, non-limiting light conditions (shallow), low flushing. However, can be an issue in estuaries with short retention and high flushing.	% cover, biomass, entrainment (annual between Oct and early March). Follow broad scale mapping methods. Use EQR approach - % cover on Available Intertidal Habitat (AIH); Affected Area (AA) of >5% macroalgae (ha)*; AA/AIH (%)*; Average biomass ($g.m^2$ wet weight) of AIH; Average biomass ($g.m^2$ wet weight) of AA; % algae >3cm deep in sediment (entrained). *N.B. Only the lower EQR of the 2 metrics, AA or AA/AIH is used in the final EQR calculation. This variable is addressed in the ETI in a more comprehensive manner – needs to be further reviewed in context of implementing ETI approach (Opportunistic Macroalgal Blooming Tool). Easy to measure with aerial photography and ground truthing (broad scale mapping). Not all macroalgal growths are solely anthropogenically driven – can be facilitated by naturally high nutrient levels entering from the catchment or ocean. Present information suggests a strongly non-linear response once the system is degraded.	Y
20.	Shellfish biodiversity	Like other macrofauna, shellfish diversity will be strongly correlated to habitat conditions and affected by upstream pressures such as sedimentation.	Yes (as above for macroalgae)	Would be best measured at the estuary scale. Integrates over weeks to months. Not particularly temporally variable. Links to ecosystem health status have been demonstrated.	N

#	Variable	Relevance to upstream management	Dependent on estuary class?	Strengths and weaknesses, considerations	Supporting Data for Southland Estuaries
21.	Biodiversity of macrofauna (measures incl. biodiversity, multivariate indices, trait based index).	Measures of macrofauna community structure are highly sensitive to changes in pressures (good for early warning), and can integrate over time. Respond in different ways to contaminants, nutrients, organic enrichment, deposition rates, turbidity, and changes in muddiness.	Yes. Currently available indices and sampling are derived and targeted for intertidal areas.	Sensitivity of macrofaunal communities and power of multivariate community analyses make macrofauna-based indicators particularly good. Levels indicative of health are available using published indices. Spatial and temporal variation reasonably well understood. Will be important in monitoring and for use as state variable(s) for gauging Ecosystem Health. Macrobenthic taxonomic composition, abundance & biomass. Annual Oct-March at representative sites (both high and low susceptibility habitats). However, assessing estuarine condition by macroinvertebrates is complicated by the high variability of natural conditions in estuaries and their often modified nature. In particular, it is important to target sites that are representative of both highly susceptible habitats as well as less susceptible zones, and to ensure that sampling is undertaken at the same time each year. It is strongly recommended that NZ macroinvertebrate/ physico-chemical variable relationships be used to assess estuary condition in NZ. This is because the physical conditions of most NZ estuaries (dominated by largely intertidal, well-flushed, shallow, short residence time estuary classes and the absence of midwater saltmarsh), differ greatly from the majority of the overseas estuaries types and the associated data sets (dominated by marine/estuarine subtidal data) which have been used to derive international biotic indices.	Y
22.	Abundance / biomass of engineering species	Measures of macrofauna community structure are highly sensitive to changes in pressures (good for early warning), and can integrate over time. Respond in different ways to contaminants, nutrients, organic enrichment, deposition rates, turbidity, and changes in muddiness.		Dependent on the species selected, cockles, very limited info. Sensitivity of macrofaunal communities and power of multivariate community analyses make macrofauna-based indicators particularly good. Levels indicative of health are available using published indices. Spatial and temporal variation reasonably well understood. Will be important in monitoring and for use as state variable(s) for gauging ecosystem health.	N
23.	Frequency of major deposition events.	There are factors beyond human control that influence major deposition events, but land use practices in catchments can make them prone to erosion and slips during events.	Importance likely dependent on flushing and resuspension characteristics of estuaries. Location of the deposition within an estuary also matters – mud deposited on a coarse substrate will have greater effects than mud deposited on mud.	Spatially and temporally variable. Easy to measure in intertidal areas (if you can predict from weather models when they are likely to occur and can mobilise ground staff), but difficult in subtidal. Lag times between the rate of sediment loading to rivers, and delivery of sediments to estuaries. May be applicable for monitoring earthworks activities, rather than long-term monitoring tool. Issues in defining and measuring an "event" both at a spatial and temporal scale have been raised.	N
24.	Modelled sediment accumulation rate.	These both address the overall rate of sediment inputs from freshwater. This can be significantly increased by land use changes in the catchment and decreased by mitigation. They are also affected by within estuary activities such as building structures and dredging and alteration of hydrological regimes.	Will likely require estuary specific limits derived by using catchment loading information and multipliers.	Deposition rates can be predicted using existing tools (e.g., CLUES) and depositional modelling (e.g., S2S), including spatial variability. Ground truthing is required but this may need monitoring over a long duration before results are known (but see core profiling below). Links to ecosystem health status not demonstrated.	N

#	Variable	Relevance to upstream management	Dependent on estuary class?	Strengths and weaknesses, considerations	Supporting Data for Southland Estuaries
25.	Measured sediment deposition.	These both address the overall rate of sediment inputs from freshwater. This can be significantly increased by land use changes in the catchment and decreased by mitigation. They are also affected by within estuary activities such as building structures and dredging and alteration of hydrological regimes.	Yes, as above, and noting different estuary classes have different susceptibility.	Sedimentation Ratio (current annual mean relative to NSR) & % Estuary Area with Sedimentation Rate >5xNSR Rates are spatially variable within estuaries (depositional versus erosional zones, influenced by waves, currents and residual circulation). Sedimentation rate measurements can be made using sediment plates at locations within estuaries but so far results are highly temporally variable. Core profiling studies and bathymetric surveying are techniques often required for interpreting data. Effects of within-estuary vs freshwater inputs should be able to be separated. Some links to ecosystem health have been demonstrated.	Y
26.	*E. coli - Water*	Elevation of *E. coli* in estuary waters is primarily attributable to upstream sources in catchments. Exceptions include birds, marine mammals, discharge from vessels, and faecal shedding from bathers.	No, but patterns in survivorship and prevalence will vary temporally and spatially within estuaries as a function of rainfall, salinity levels, water temperature, and water clarity, which affects rates of die off due to solar radiation. Concentrations will be influenced by rates of mixing and dilution, which is affected by tidal flushing and volume of receiving waters.	Indicator of choice for health risk to recreational users of fresh waters (an attribute under NPSFM), and in some estuary waters. Variability in time and space (patchiness) can impede ability to identify trends in response to changes in pressures. Standard methods and easy to measure. Can persist and grow in the environment (e.g., in sediments). Often correlated with other water quality parameters (e.g., suspended solids).	Y - spatially limiting but information available
27.	Enterococci - Water	As for *E. coli.*	As for *E. coli.*	As for *E. coli*. Enterococci the indicator of choice for health risk to recreational users of coastal waters and some estuary waters. *E. coli* and enterococci considered transitional in brackish waters – both likely important for estuaries.	Y - spatially limited but information available
28.	Inorganic compounds in sediment (metals/metalloids)	With the exception of metals/metalloids from catchment sediments, the majority of inorganic compounds will enter estuaries from anthropogenic upstream sources, including sewerage discharges, landfill leachate, and storm water runoff. In some cases, aquaculture structures and ports can be sources of contaminants.	No, but likely to be concentrated in sediments closest to river mouths and discharges (outfalls).	Sediments are a good integrator over time compared to water samples. Can be expensive to analyse. ANZECC guidelines provide limits, but these are based largely on Australian conditions. Concentration in sediment, established protocols are already in use.	Y
29.	Inorganic compounds. - Water	As for sediment inorganic compounds.	As for sediment inorganic compounds.	High temporal and spatial variability in water. Expensive to analyse, and near detection limits.	N

#	Variable	Relevance to upstream management	Dependent on estuary class?	Strengths and weaknesses, considerations	Supporting Data for Southland Estuaries
30.	Faecal coliforms. - Water	As for *E. coli.*	As for *E. coli.*	As above for *E. coli* and *Enterococci.* Routinely used to assess the condition of shellfish growing waters.	Y - spatially limited but information available
31.	Anthropogenic organic compounds in harvestable species. - Biota	As for inorganic compounds in sediments and water.	As for inorganic compounds in sediments and water.	Shellfish are good integrators of what is happening in the water column, and the concentration of toxicants within their tissues can provide an indication of levels within the water body.	Y - one limited study done 2013
32.	Anthropogenic organic compounds. - Sediment	As for inorganic compounds in sediments and water.	As for inorganic compounds in sediments and water.	The seabed is often a good integrator similar to shellfish.	Y - one limited study done 2013
33.	Harvestable shellfish *E. coli. - Biota*	As for water *E. coli.* Contamination of shellfish primarily due to upstream sources.	As for water *E. coli*	Used by regulatory agencies as the faecal indicator for shellfish flesh.	N
34.	Inorganic compounds (metals/metalloids) in harvestable species. - Biota	As for inorganic compounds in sediments and water.	As for inorganic compounds in sediments and water.	Metals/metalloids concentrations in shellfish link to all three values. Shellfish good integrators over time for metals/metalloids, since they are too low/variable in water to reliably measure.	N - one limited study done 2013

#	Variable	Relevance to upstream management	Dependent on estuary class?	Strengths and weaknesses, considerations	Supporting Data for Southland Estuaries
35.	Harvestable shellfish *Cryptosporidium* oocysts. - Biota	The protozoan *Cryptosporidium* is associated with upstream sources of contamination. *Cryptosporidium* oocysts have been found to accumulate in shellfish (Graczyk et al., 2007).	No; however, characteristics of surrounding catchments (number of animals) combined with flushing characteristics of the estuary will likely influence concentrations.	Consider both parvum and hominis strains. The former is animal-related, the latter human-related and somewhat more infectious.	N
36.	*Cryptosporidium* oocysts. - Water	*Cryptosporidium* oocysts have been found to accumulate in shellfish (Graczyk et al., 2007).	As for *Cryptosporidium* in shellfish.	NZ Drinking-Water standards include requirement for its removal.	N
37.	*Campylobacter. - Sediment*	Animals in upstream catchments are the greatest reservoir of this potentially waterborne pathogen.	As for *Cryptosporidium* in shellfish.	Used as the basis of NPS-FM water quality standards. the greatest cause of bacterial dysentery worldwide; campylobacterisos is the most common reported notifiable disease in New Zealand. Prevalent in New Zealand freshwaters (McBride et al., 2002, Till et al., 2008). May be present in shellfish flesh. Sediment may be a good integrator .	N
38.	*Campylobacter. - Water*	As for sediment *Campylobacter.*	As for sediment *Campylobacter.*	As for sediment *Campylobacter* May be difficult to measure reliably in estuary waters if too diluted.	N - limited study in FW
39.	Universal PCR markers (e.g., Bacteroidales). - Water	As for other faecal indicators such as *E. coli.*	As for other faecal indicators such as *E. coli.*	As an anaerobe, *Bacteroidales* does not persist and grow in the marine environment, so may be a better indicator of ‘recent’ contamination.	N

#	Variable	Relevance to upstream management	Dependent on estuary class?	Strengths and weaknesses, considerations	Supporting Data for Southland Estuaries
40.	Enteroviruses. - Water	Directly a function of upstream human sources of contamination, although could enter estuaries through outfalls and activities on the water (vessels).	As for other faecal indicators such as *E. coli.*	Both a pathogen and an indicator (because they are a group with many members). Long-term health impairment may arise from exposure to these viruses whereas other viruses of concern for water contact tend to have shorter-term consequences. Human-specific strains found to be present on many occasions at freshwater recreation sites (McBride et al., 2002).	N
41.	*Finfish*	Complex interactions with habitat, flow regimes and food availability.	No	Needs more development.	N
42.	Cyanobacteria and Harmful Algal blooms.	Complex interactions with a multitude of drivers.	There is the potential for there to be more occurrence in estuary with grater residence times but is likely to be case specific.	Cell counts and toxin concentrations. Mean monthly from representative areas of estuary water column. Harmful Algal Bloom cell counts and toxins. Mean monthly from representative areas of estuary water column. Requires development. Applicability of current NOF Cyanobacteria – planktonic	N
43.	Estuary Trophic Index Score	The ETI Score is an integrated measure that can be indirectly linked to upstream management. NZ estuaries with a poorer ETI score are generally in developed catchments with high nutrient and sediment loadings. Variables used to calculate the score provide a stronger link to catchment management.	Yes, the calculation of the score is dependent on estuary class.	Some of the variables used to calculate the score require further research and development. However, the index uses an array of facets when considering the health of an estuary giving a more holistic view. ETI score = normalised FPSR (final primary symptom rating) + FSIR (final supporting indicator rating) / 2 See ETI section for more detail.	Y
44.	Gross Eutrophic Zone [i.e. >50% macroalgal cover and gross eutrophic sediment conditions (mud>25%, aRPD <1cm).]; (ha, % of estuary area)	Work has demonstrated a correlation between modelled catchment loads and GEZ measures across a range of NZ estuaries. However, this work to date has been confined to SIDEs only.	Yes those with intertidal areas – SIDEs and SSRTRE.	Due to the possibility that GEZ areas can become self-reinforcing there is a rationale that any in a system is problematic and shouldn't be there. As these conditions are potentially irreversible (or at the least very difficult and costly) there needs to be early warning sign built in. This is where the ability to use other metrics with earlier signs, such as EQR, are fundamentally important. GEZ should not be considered a stand-alone metric for managing estuaries.	Y - New River Estuary: 2001, 2007-13, 2016. Jacobs River Estuary: 2003, 2007-2013, 2016. Waikawa Estuary: 2004, 2007, 2008, 2016. Haldane Estuary: 2004, 2016. + some for 2018, note also for Fortrose SSRTRE.

Appendix B – Further consideration of short-listed attributes and attribute state tables

The short list (i.e. attributes from Appendix A with data available) of attributes is further considered for its validity and applicability as an attribute for Southland estuaries. These attributes and their associated recommendations are summarised in Table 2.

Sediment quality/sedimentation attributes

Sediment organic content

Recommendation: Needs development.

As TOC is theoretically reflected in apparent redox potential discontinuity (aRPD) via oxidative processes it seems overly complex to integrate TOC with muddiness and will therefore not be considered further.

Sediment grain size (mud content/extent and total organic carbon)

Recommendation: Proposed attribute for a numeric freshwater objective.

Mud and sand habitats are often the dominant habitat type in New Zealand estuaries (Robertson et al., 2002). Changes in sediment grain size can be indicative of habitat change and type of sediment supply, and can occur as a result of terrestrial sediment (Hewitt et al., 2014). Although estuaries are a natural sink for sediments, the amount of land-derived fine sediments entering estuaries has increased due to human impacts associated with changes in land use (e.g., deforestation) (Zaiko et al., 2018). These sediments enter estuaries via the stream and river networks and eventually is deposited into estuaries if not flushed out into coastal waters (Robertson et al., 2002). Sediment mud content within estuaries can increase as a result of fine sediments entering estuaries, and hence can be used as a surrogate for sediment accumulation (Hewitt et al., 2014). It can also provide information regarding the condition or state of estuaries as elevated delivery to and retention of terrigenous mud (<63µm particle diameter) in estuarine systems can impair feeding, behavioural responses, larval recruitment, and trophic interactions in coastal food-webs (Norkko et al., 2002; Ellis et al., 2002; Cummings et al., 2003; Duarte et al., 2005; Jones et al., 2011; Vasconcelos et al., 2011).

Extensive National Estuary Monitoring Protocol (NEMP; Robertson et al. 2020) data from typical NZ shallow tidal lagoon and tidal river estuaries show that extensive areas of soft mud, elevated sedimentation rates, and high sediment mud contents are commonly associated with low seagrass cover, a degraded macroinvertebrate community and degraded sediment conditions if nutrients are excessive and soft mud areas are overlain with dense nuisance beds of opportunistic macroalgae (Robertson et al. 2016). In New Zealand sediment mud content is a major stressor of benthic animal communities, such as macrofauna (Robertson et al. 2015b) including mahinga kai species, and ecological responses to bed-sediment mud content are reasonably well understood (Gibbs and Hewitt 2004, Thrush et al 2004). Fine sediments can also become contaminated with elevated metals, elevated nutrients, organic matter, potentially disease-causing organisms and potentially toxic chemicals (Robertson et al. 2002). The tendency for sediment to become anoxic (oxygen deficient) is higher if the sediments are muddy and the interstitial spaces small (Robertson et al., 2015). Underfoot condition (muddiness) is also a key component in human preference and the value people place on marine environments (Batstone & Sinner, 2010).

There are some considerations when using mud content as an attribute for managing upstream effects. Besides catchment supply, sediment mud content can be influenced by natural processes, such as resuspension, remobilization by currents and water movement, occurring within estuaries and associated coastal waters (Hewitt et al., 2003). These natural processes can create within-year and between-year variability in sediment grain size without a strong predictable pattern being observed for some intertidal areas (Hewitt et al., 2014). Furthermore, any changes in hydrology via, reclamation, significant sediment accumulation can also confound how a system responds to catchment sediment supply and natural processes.

As mud is less of an issue in relation to eutrophication in moderately deep to deep estuaries, and information on its impacts is limited, thresholds were not proposed in the Estuary trophic index (Robertson et al., 2016) for these less susceptible estuary classes. Where significant sediment inputs are present in such estuaries, they should be considered on a site specific basis with appropriate management response.

<u>Thresholds:</u>

Individual site

Multiple studies (Norkko et al., 2002; Thrush et al., 2003; Gibbs and Hewitt, 2004; Anderson, 2008; and Sakamaki and Nishimura, 2009; Hailes and Hewitt, 2012; and Rodil et al., 2013) have highlighted the importance of sediment mud content as a reliable predictor of the distribution and abundance of NZ estuarine macrofauna.

However, derived thresholds and ratings from these studies potentially lack strong regional transferability and are limited in terms of the number/type of taxa with Southland similarities. As such, their use in assessing estuary condition at any particular site needs to be supported by information that indicates that:

i. the estuary in question fits within the upper North Island estuary classification used to produce the ratings/thresholds,
ii. that due regard is given to differences in taxa i.e. those that have not yet been rated for sensitivity.

A review of monitoring data from 25 typical NZ estuaries (SIDEs) (Robertson et al., 2016b; includes Wriggle database 2009-2014 [includes all Southland data], Robertson, 2013 [includes Southland data], Robertson et al., 2015b [includes Southland data],) confirmed a 'high' risk of reduced macrobenthic species richness when mud values were >25-30% mud (Figure 5). This is supported statistically, see canonical analysis of the principal coordinates (CAP) for the effect of mud content in (Figure 5) by the increasing dissimilarity in the macrobenthic community as mud contents increase above 25-30% mud. These findings are supported by other studies which show that sediments become "cohesive" or sticky once the % mud content increases above approximately 20-30% mud depending on such factors as the clay content (Houwing, 2000).

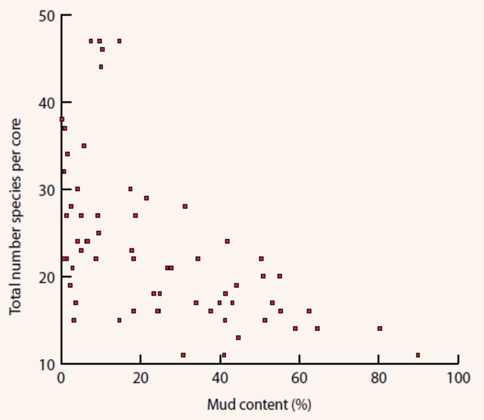

Figure A17. Sediment mud content and number of macrobenthic species per core from 12 estuaries scattered throughout NZ, and representing most NZ shallow, short residence time estuary types (Wriggle Coastal Management database 2009-14).

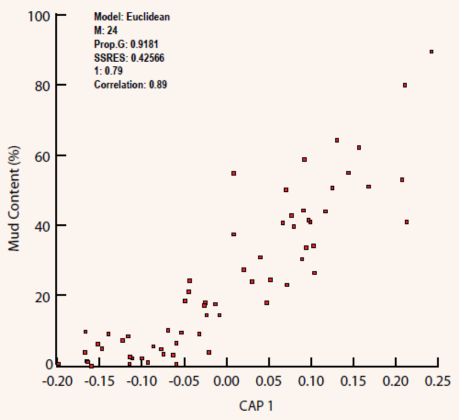

Figure A18. Canonical analysis of the principal coordinates (CAP) for the effect of sediment mud content (exclusively) on the macroinvertebrate assemblages from 25 typical NZ estuaries (i.e. CAP1) among sites.
Note: M = the number of PCO axes used for the analysis, Prop.G = the proportion of the total variation in the dissimilarity matrix explained by the first m PCO axes, SSRES = the leave-one-out residual sum of squares, 1 = the squared canonical correlation for the canonical axis, Correlation = the correlation between the canonical axis and the sediment mud content or pollution gradient.

Figure 5. Sediment mud content and macrofaunal species. Excerpt from Robertson et al. (2016b).

Robertson et al., (2015b) used organic enrichment, grain size and macroinvertebrate data from 135 sites in 25 estuaries scattered throughout NZ (some of which occur in Southland) encompassing 0.1 - 92.3% mud content. This study showed that macroinvertebrate diversity and abundance was greatest at low mud content and produced mud and organic sensitivity ratings for NZ estuarine macroinvertebrates with the utilisation of generalised additive models.

Sediment mud gradient alone accounted for most of the variation in macrobenthic assemblages among sampling locations (Spear-man correlation = 0.31). When combined with the TOC gradient, the correlation decreased (Spearman correlation= 0.27). However, the results also confirmed sediment mud content and TOC as co-varying (R^2 = 0.71; P = 0.001) and to be key drivers of the macroinvertebrate community (metals were not found to be key drivers but all sites had metals concentrations below ANZECC ISQG toxicity thresholds). There was found to be successful delineation of benthic condition along stressor gradients of sediment mud and total organic carbon contents.

The results indicate more diverse and abundant macrofaunal communities occur in low to intermediate mud concentrations (<25–30%) compared to areas where mud content exceeds 30%. Overall, the findings indicate that NZ estuarine sites with 2 - 25% mud content support a diverse and abundant macroinvertebrate assemblage and low organic enrichment (<1.2% total organic carbon) compared to systems characterised by >25% mud (Robertson et al., 2015b). Since large suspension-feeding bivalves are key species that influence nutrient cycling and productivity in sediment habitats (Lohrer et al., 2004; Thrush et al., 2006) and positively influence functional diversity of associated macrofaunal communities (Savage et al., 2012), any direct negative effects of sedimentation on these species may lead to detrimental changes in estuarine ecosystem functioning (Thrush et al., 2006; Barbier et al., 2011). Deriving thresholds based on macroinvertebrate community should therefore give consideration to species thresholds. The three suspension-feeding bivalves (*Paphies australis, Austrovenus stutchburyi* and *Cyclomactra ovata*) included in the analysis were found to have an optimum range of 0-40%. In the Robertson et al., (2015b) study of the 39 taxa, the optimum range upper limit for taxa was captured by the following mud content thresholds: <25% (17 taxa), <20% (14 taxa), <15% (11 taxa) and 10% (2 taxa). Bethelsen et al., (2019) review found some similar thresholds. Robertson et al., (2016b) carried out further analysis and assessment biotic indices and species sensitivity to disturbances using a variety of indices and regression trees. The trees identified sediment mud content as the dominant abiotic driver of benthic condition up to ∼30% mud, beyond which (i.e. in the ∼30–95% mud range) TOC became the focal stressor with disturbance thresholds at ∼1.2% and 3% TOC. This breakpoint coincides with relevant thresholds of TOC calculated for estuaries in other parts of the globe (e.g. Hyland et al., 2005; Sutula et al., 2014), is likely to be linked with oxygen depletion and build-up of toxic by-products (ammonia and sulphide) associated with the breakdown of organic materials (Hyland et al., 2005).

As TOC is theoretically reflected in apparent redox potential discontinuity (aRPD) via oxidative processes it seems overly complex to integrate TOC with muddiness and will therefore not be considered further.

This is based on the macroinvertebrate community response and not the seagrass response. Seagrass degradation is expected to be primarily caused by reduced water clarity, and hence light availability, as a result of resuspension and elevated suspended sediment input loads, as well as degraded sediment conditions (Robertson et al., 2016). Work in the US observed the preferred sediment mud content for seagrasses is 0.4% - 30% mud content (Batiuk et al., 2001). Preliminary findings from NZ estuary monitoring data (Wriggle reports 2002 - 2013), tend to support this range, for example extensive broad scale mapping of seagrass cover for 45 typical NZ tidal lagoon and tidal river estuaries (shallow, residence time <3 days) indicate that seagrass cover is absent or less than 1% cover for estuaries with greater than 20-30% of the estuary area as soft mud (i.e. >25% mud content) (Robertson et al., 2016). Extensive high density seagrass (*Zostera*) beds are found at 0.3 - 0.6% mud content in Freshwater Estuary, Stewart Island and in Waikawa Estuary, seagrass beds are present at 10% mud content but often absent in the extensive 25 - 80% mud content zone (Robertson et al., 2016). Due to the fact that multiple other variables (such as light regime, velocity and water quality; Kemp et al., 2004) may be driving the outcome for seagrass it is difficult to justify use of this species as the basis for setting thresholds. However, it is important to be cognisant of mud content being a likely key driver of seagrass bed health and resilience.

The recommended bandings for mud content are shown in Table 4.

Table 4. Attribute state option table for mud content.

Value	Ecosystem health
Freshwater body type	Tidal lagoon estuaries (SIDES) and tidal river estuaries (SSRTRE)
Attribute group	Southland attribute
Attribute name	Mud content
Attribute unit	% mud content*
Attribute band and description	**Numeric attribute state**
	3-year median[1]
A Little to no stress on aquatic organisms and seagrass beds. Ecological communities are healthy and resilient	≤5
B Minor stress on aquatic organisms, particularly sensitive species.	5 and ≤15
C Moderate stress on a number of aquatic organisms and risk of some species being lost.	>15 and ≤25
Proposed minimum acceptable state	**25**
D Significant, persistent stress on a range of macroinvertebrates. A likelihood of local extinctions of keystone species and loss of ecological integrity.	>25
*Measurement applies to individual sites within an estuary. Criteria to ensure monitored sites are adequately representative of the estuary are to be developed. [1] To be based on an annual monitoring regime.	

Estuary Spatial mud extent

Information on the relationship between the spatial distribution of these supporting indicators, and overall biological impacts, is very limited. Thresholds for mean mud content for whole of estuary will be very hard to defensibly support (and are unlikely to be readily achievable for management purposes), but could provide general narrative guidance. Whole estuary assessments could include measures such as % mud content (mean of whole estuary) and % estuary area comprising soft mud (soft mud being for example mud content >25%).

The term 'Total Soft Mud' is defined as the combination of the 'soft mud' and 'very soft mud' indicators in the National Estuary Monitoring Protocol (NEMP) (Robertson et al., 2002). Further refinement and validation of these has been conducted by Robertson et al., (2016b, 2017b):

- Soft Mud. A mixture of mud and sand, the surface appears grey-brown (may have a black anaerobic layer below) and when a human walks on it they sink 2-5cm
- Very Soft Mud. A mixture of mud and sand, the surface appears grey-brown and may have a black anaerobic layer below and when a human walks on it they sink >5cm

Based on the results from a selection of typical NZ tidal lagoon and tidal river estuaries the percent mud content of "total soft mud" generally equates to estuarine sediments with a % mud content in the 25 - 100% range (Robertson et al., 2016). This is in the range (i.e. the range above which sediments become "cohesive" or sticky, and experience significant shifts in macroinvertebrate communities). Therefore, mapping the extent of total soft muds in an estuary (i.e. using the NEMP broad scale mapping methodology) provides a strong indication of the spatial extent of mud related environmental effects.

Converting site specific macroinvertebrate response to a spatial whole estuary scale requires a process similar to the one undertaken for sedimentation (e.g., literature review, data mining, work-shopping, consultation, peer review). In other words, there is a strong relationship between increasing site specific sediment mud content and persistent ecological degradation but the relationship between the spatial extent of muddy sediment and overall biological impacts is still being established for NZ estuaries. It is worth noting that sediment mud content change is a key indicator and known driver of ecological shifts and can occur without being detected by other indicators (Zaiko et al., 2018).

Therefore, the spatial extent of muddy (>25% mud content) areas as an attribute is best suited as a change from baseline attribute. The recommended bands are shown in Table 5.

Table 5. Attribute state option table muddiness.

Value	Ecosystem health
Freshwater body type	Tidal lagoon estuaries (SIDES) and tidal river estuaries (SSRTRE)
Attribute group	Southland attribute
Attribute name	Mud extent
Attribute unit	m^2 of intertidal area
Attribute band and description	**Numeric attribute state**
	Comparison to baseline monitoring
Pass No likely further deterioration of ecology due to increased mud cover.	Decrease or no change*
Fail Likely deterioration of ecology due to increased mud cover.	Increase
Muddiness is defined as having >25% mud content. *Change is calculated from earliest available monitoring assessment.	

Note that no bottom line has been proposed as it is difficult to establish the spatial coverage of mud that is 'excessive' (D band) along with reference conditions. Therefore, this attribute may be more suited as a narrative or for the formation of a target (or part of). As this matter then becomes a value judgement it should be further deliberated by decisions makers to establish an acceptable system. Rate of change could also be considered as a means to track progress and health of an estuary.

Method:

Individual site

There are no current national standards for grain size (mud content) but sampling protocols, analytical procedures and guidelines do exist. Mud is defined as grain sizes <62.5 µm and expressed as a proportion of total sediments. The Southland estuary monitoring programme samples intertidal areas, follow guidelines described in Robertson et al., (2002). Currently sampling is conducted in most cases annually; no higher frequency monitoring is conducted due to prohibitively expensive costs. Without continuous monitoring detection of changes due to natural phenomenon, such as the ENSO is not possible. Though, it should be noted that this consideration may be only minor for Southland estuaries, especially those which have high sediment catchment supply and current accumulation rates.

To optimise sampling strategy, grain size sampling has been aligned with collecting data on other benthic attributes and/or state variables (e.g., sediment quality characteristics, macrofauna, and sedimentation rate). To date analysis has not been done to assess the effect of varying frequency and replication though this would verify if appropriate sampling design is in place. Though the intertidal habitat is relatively venerable it may not represent all upstream effects. Thus there is rationale to explore the validity of sub tidal sampling which is not currently conducted. Further consideration should also be given to differences in tidal range which can also have a strong effect on sediment mud content (Deloffre et al., 2007).

Monitoring requires measurement of grain size (enables calculation of mud content) concentration in upper 2cm of sediment as set out in the NEMP (Robertson et al., 2002). This happens usually annually Oct-March but could be done less frequently for systems that are considered in a good state. This is done using laboratory accredited measurements. See Inorganic compounds in sediment (metals/metalloids) section for sampling method details.

Estuary Spatial extent

There are broad scale mapping methodology protocols contained within the NEMP (Robertson et al., 2002). This can be used to measure the spatial extent of mud content; or representative sites may be used and the site specific threshold applied. The mapping of 'total soft mud' is a subjective appraisal of substrate and it should be noted that other factors may influence this appraisal (depth of sinking), such as muds within a gravel matrix (Robertson et al., 2016)

There has been some quasi-validation (Robertson et al., 2017b), however more development is required. Therefore, spatial mapping of mud content is limited by current measurement methods which are broad scale and therefore less accurate. To measure a whole estuary with laboratory accredited measures would currently be prohibitively expensive.

Sediment nutrient concentrations (N, P)

Recommendation: Needs development.

As part of the state of environment estuarine monitoring sampling is taken from the top 2cm of the sediment at specified sites on a scheduled basis. The analysis measures concentrations of total nitrogen and total phosphorus along with total organic carbon (see TOC section) and grain size (see grain size section). Robertson (2013) and Robertson et al. (2015) demonstrate consistent relationships between TN and TOC and biota for a wide range of NZ estuaries. However, information on the relationship between the spatial distribution of these supporting indicators, and overall biological impacts, is very limited, which is reflected in the lack of thresholds to apply. Notwithstanding, conditions that cause persistent ecological degradation (e.g. to macrofauna) indicate significant adverse impacts are occurring, and like the primary indicator macroalgae, a measure of the spatial distribution is also required in addition to the concentrations in order to determine an overall estuary rating for that indicator. If development is undertaken in the future for this variable considerations for representativeness and spatial extent (site of whole estuary) will be needed (similar to RPD).

Depth of RPD (REDOX Potential Discontinuity) in sediments / Areal extent of hypoxic/anoxic bottoms

Recommendation: Proposed attribute for a numeric freshwater objective.

With increased organic matter loading oxygen supply generally decreases (Hargrave, 2008). Given sufficient organic matter supply sulphate reduction becomes the major metabolic pathway predominating over oxic (aerobic) respiration (Hargrave, 2008). Changes in oxic conditions in surface sediments affect the taxonomic composition of both microbial and faunal communities (Nilsson et al., 1991; Diaz and Rosenberg, 1995; Nilsson and Rosenberg, 2000; Gray et al., 2002; Diaz et al., 2004 and Robertson et al., 2016). As hypoxic conditions progress there is a corresponding increase in sulphide concentrations (Hargrave, 2008). Total S^{2-} presence creates toxic biological effects for benthic fauna by interference with aerobic respiration (Grieshaber and Völkel, 1998). H_2S inhibits cytochrome C oxidase in the electron transport system and HS^- binds with the ferric (Fe^{3+}) ion of the cytochrome to prevent oxygen release by oxyhaemoglobin. This reduced taxonomic composition has been linked to reduced availability of forage for fish, birds and other invertebrates (Sutula et al. 2014). Macrofauna have a higher tolerance to total S^2 than fish and taxa inhabiting mud bottoms where concentrations may be higher and are less sensitive than those on hard or sandy substrates (Bagarino, 1992). Gray et al., (2002) reviewed published literature and concluded that fish are more sensitive to hypoxic conditions than crustaceans and echinoderms, with annelids and molluscs less sensitive, respectively.

The depth of sediment oxygenation (the zone where conditions change from oxidizing to reducing) is termed the Redox Potential Discontinuity (RPD).

Chemically, anoxic sediments accumulate sulphides (which give sediments a black colour) and ammonium, which are highly pervasive causes of sediment toxicity to aquatic life (Losso et al., 2007, Machado et al., 2004). A shallow RPD layer forces most macrofauna towards the sediment surface to where oxygen is available. In sandy, porous, non-eutrophic sediments, the RPD layer is usually relatively deep (>3cm) and is maintained primarily by current or wave action that pumps oxygenated water into the sediments. In finer silt/clay sediments, physical diffusion limits oxygen penetration to <1cm (Jorgensen and Revsbech, 1985) unless bioturbation by infauna oxygenates the sediments. The tendency for sediments to become anoxic is much greater if the sediments are muddy and therefore interstitial spaces small. Pearson and Rosenberg (1978) developed a useful organic enrichment tool that indicates the likely macrofauna community that is supported at a particular site based on the measured RPD depth. This tool has been used extensively to date, in a multi-indicator approach, to help interpret intertidal monitoring data and its relationship to organic enrichment in Southland (and NZ) estuaries (Robertson et al 2016b).

The RPD can be measured as apparent RPD (aRPD) which is the depth where sediment changes colour; or measured directly as RPD using hand-held ORP meters which are a redox potential electrode coupled to a millivolt meter (Rosenberg et al. 2001).

The visual-aRPD depth measure (often done in situ when sediments are intertidal and with digital imaging if subtidal) relies on the assumption that in the absence of oxygen, the microbial sulphate reduction results in the precipitation of Fe-sulphides, producing a grey/green or black sediment coloration. aRPD has been the primary method used to measure RPD depth in NZ estuaries to date. It is a recommended indicator in the NEMP (Robertson et al., 2002), but with the proviso that it only be used by experts trained using both visual and meter approaches. The aRPD essentially looks for a “colour break,” that is, the maximum colour difference below the sediment-water interface at which lighter-coloured (tan, brown, beige, yellow, or red), more-oxidized surface sediments transition into darker-coloured (grey, black, or blue-black), more-reduced deeper sediments. The depth of the aRPD is easily measured and has been found to be an extremely useful parameter in characterizing certain biogeochemical aspects of the sedimentary environment. For instance, the aRPD represents the depth at which iron exists as coloured, insoluble, ferric hydroxides, which dissolve into solution as iron monosulphides in a reducing environment, e.g., in the presence of sulphate reduction (Teal et al., 2009). The aRPD is a highly subjective and variable measure between providers and care is needed when analysing existing data.

For direct measures of RPD using a hand-held the electrode is inserted to different depths into the sediment and the extent of reducing conditions at each depth recorded (RPD is the depth at which the redox potential is ~0 mV, Fenchel and Riedl, 1970, Revsbech et al.,1980, Birchenough et al., 2012, Hunting et al., 2012). Robertson et al., 2016 recommend use of RPD over aRPD as the former provides a strong indication of the macrofaunal response to stress from reducing conditions, whereas visual (aRPD) measures ‘provide a relatively weak indication’, unless the aRPD is at 0cm. However, there are no current standard operating procedures defined for collecting ORP measures in NZ estuaries and thus any available data are likely be highly variable between providers and estuaries. ORP measurements also have a large number of caveats, such as drainage of porewater, tidal state, poor vertical resolution in depth scale of sampling (needs to millimetre not centimetre), variance in probe types, in-situ vs extracted core measurements, variance in interpretation of Eh profiles and time taken in field; this suggests the interim ETI ratings based on applying absolute mV values to define bands is somewhat currently flawed and requires further methodological development.

Assuming developed and acceptable methodology is applied to directly measured RPD via handheld probes the criteria from Hargrave et al., (2008) is appropriate (Figure 6); with a threshold of -150mV applied when measured at 1cm depth.

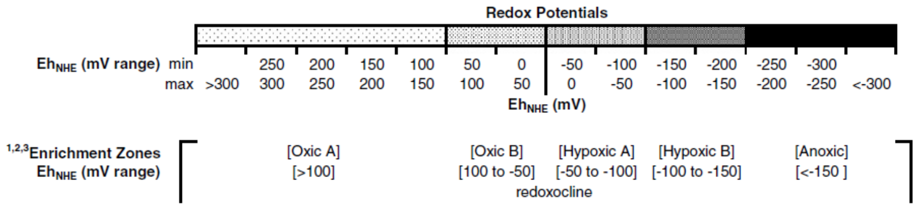

Figure 6. RPD measurements excerpt from Hargrave (2008)

More Southland specific criteria development is needed to progress this particular measure as an attribute, especially considering the lack of substantial data and time consuming method for measurement.

Some research has found the aRPD to strongly correlate to the true RPD depth (Grizzle and Penniman, 1991; Rosenberg et al., 2001), which is the depth where Eh (measured sediment reduction/oxidation potential) is zero. However, some of these measures were for multimetre measures rather than centimetre of millimetre values. Both measurements are very useful (Grizzle et al 1991), however data for aRPD can be collected and processed in such a short time for many sites/areas. For NZ, data needs to be collected in a standardised way, which addresses and acknowledges the caveats highlighted and a relationship developed between aRPD and RPD. Theoretically, aRPD is a more stable and reliable measure of integrated sediment oxygenation. Concerns especially arise when the RPD is deeper and the aRPD/RPD relationship deteriorates, though deeper RPD levels are of less environmental concern. Additionally, as aRPD is based on a visual observation this is interpretive, especially of concern if assessor changes and/or the catchment has volcanic material present resulting in darker sediment and thus potentially masking aRPD measures. For Southland limited data is available to do this aRPD vs RPD assessment. The aRPD measurements in Southland available have been consistently collected by trained technicians.

Sand vs mud

Sandy sediments are dominated by different porewater processes and rates to those that contain mud (FDGC, 2012). Therefore, aRPD and RPD depths manifest differently in these substrates and should result in deeper RPD levels due to greater advective flow. On this basis, applying mud thresholds to sandy environments is considered setting a low bar which is non-precautionary and could result in compromised outcomes. However, it is considered to be more favourable to have something in the absence of further supporting evidence for the development of a sandy environment specific threshold/s. Resultantly the same thresholds can be applied to sandy and muddy environments in the interim.

The proposed threshold is equivalent to diffusional category for aRPD i.e. between 0 and 1 cm depth (Table 6).

Table 6: aRPD excerpt from FGDC (2012)

aRPD Depth Values	aRPD Depth (centimeters)
Zero	0.0
Diffusional	>0.0 to 1.0
Shallow	>1.0 to 2.0
Moderate	>2.0 to 3.5
Deep	>3.5 to 5.0
Very Deep	>5

As aRPD has some limitations especially as the depth increases the upper limit threshold has been restricted to 30mm. The depth upon which aRPD can be reliably be measured in the field. This results in a slight modification to the FGDC (2012) application of numbers. All available sites which have aRPD measures were collated from 2019 data (Figure 7) to test the spread of sites across potential bands for this variable. The last band was further divided into for 5mm to further discern between depths within that category. A nominal depth of 5mm was selected and added.

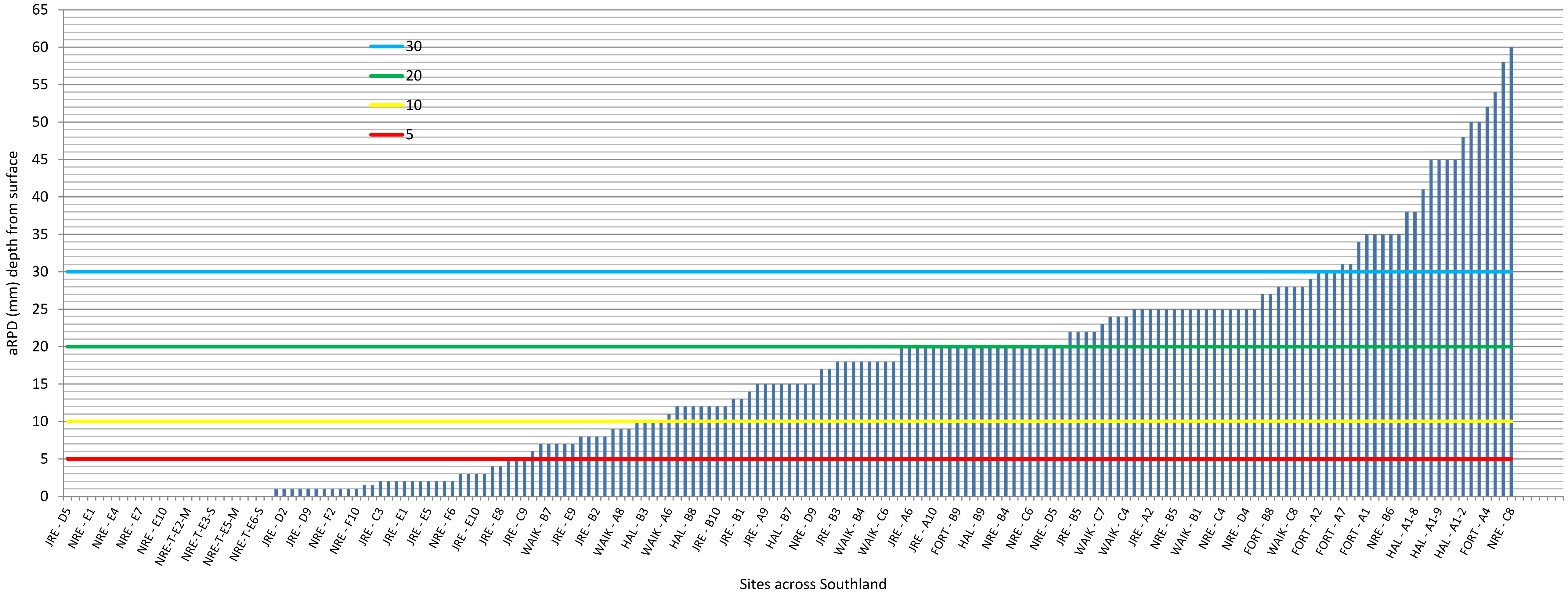

Figure 7. aRPD (mm) depth from surface with different breakpoints added and sites added from 2019 data.

The recommended attribute table for aRPD is as follows:

Table 7. Attribute state option table for sediment oxygen levels.

Value	Ecosystem health
Freshwater body type	Tidal lagoon estuaries (SIDES) and tidal river estuaries (SSRTRE)
Attribute group	Southland attribute
Attribute name	Sediment oxygen levels
Attribute unit	aRPD mm (depth to apparent Redox Potential Discontinuity in millimetres)
Attribute band and description	**Numeric attribute state**
	Median[1]
A+ Little to no stress on aquatic organisms and seagrass beds. Ecological communities are healthy and resilient, similar to natural reference conditions.	≥30
A Minor stress on aquatic organisms and seagrass.	≥20 and <30
B Moderate stress on a number of aquatic organisms exceeding preference levels for some species. A moderate risk of losing sensitive macroinvertebrate species due to oxygen stress.	≥10 and <20
C Significant, persistent stress on a range of macroinvertebrates. A likelihood of local extinctions of keystone species and loss of ecological integrity.	≥5 and <10
Proposed minimum acceptable state	**5**
D Severe loss of macroinvertebrates, a shift in the community structure and reduction in available habitat, loss of ecological integrity in addition to a fundamental shift in biogeochemical processes.	<5

[1] Measurement applies to individual sites within an estuary with a minimum of 10 measurements per site. Measurements should be taken by skilled and experienced personnel. aRPD is a variable measure between providers and care is needed when analysing data from more than provider.

Method:

Individual site

aRPD has been the primary method used to measure RPD depth in NZ estuaries to date. It is a recommended indicator in the NEMP (Robertson et al., 2002), but with the proviso that it only be used by experts trained using both visual and meter approaches.

Spatial scale

The spatial measures and criteria for defining oxygenation at this spatial scale are not currently consistent or well accepted and require further work. Measures can be conducted at representative sites, as transects, stratified randomly sampled or equidistant and interpolated. All these approaches may derive slightly different outputs and at differing levels of monitoring cost and effort. A measure of spatial distribution is required in order to determine an overall estuary rating for this indicator. It is recommended that this be pursued as a priority.

Inorganic compounds in sediment (metals/metalloids)

Recommendation: Proposed attribute for a numeric freshwater objective.

Although metals/metalloids (herein metals) occur naturally to some extent, their prevalence within estuarine sediments can increase due to human-induced changes in land-use (e.g., agriculture, urban development, waste water and commercial/industrial activities) (Zaiko et al., 2018). Land-derived metals can be flushed into streams/rivers and deposited into estuarine sediments, which act as a sink for contaminants (Robertson et al., 2002). In New Zealand estuaries, the concentrations of different metals in sediments are typically correlated (Robertson et al., 2002). Sediment metals can provide information regarding the condition or state of estuaries and link to values such as those associated with ecosystem health. At certain concentrations, sediment metals can be toxic to benthic organisms (ANZECC, 2000), and benthic organisms can also contribute to the bioaccumulation of metals in estuarine food webs (Robertson et al., 2002). Furthermore, metals can bind with fine sediments, which may cause additional stress to benthic organisms living in muddy areas. Some metals in sediments are also generally well correlated with organo-chlorine contaminants (Hewitt et al., 2014). Some additional considerations are potential sources of metal bound sediment are anthropogenic activities not associated with land-runoff such as storm water, waste water, commercial activities and boating (antifouling paints). Influence of these impacts/processes on metal concentrations may confound upstream effects (Zaiko et al., 2018).

There are no current national standards for metals but sampling protocols, analytical procedures and guidelines do exist. The Southland estuary monitoring programme samples intertidal areas, follow guidelines described in Robertson et al., (2002). The sampling design is constrained by the cost and capacity of council to sample more than is presently conducted. It has been noted that between three and five replicate samples are required to adequately assess concentrations of lead, copper and zinc (cited in Hewitt et al., 2014).

To optimise sampling strategy, metals sampling has been aligned with collecting data on other benthic attributes and/or state variables (e.g., sediment quality characteristics, macrofauna, and sedimentation rate). To date analysis has not been done to assess the effect of varying frequency and replication though this would verify if appropriate sampling design is in place. Though the intertidal habitat is relatively venerable it may not represent all upstream effects. Thus there is rationale to explore the validity of sub tidal sampling which is not currently conducted.

Thresholds:

There are ANZECC (2000) trigger values based on toxic effects to organisms for metals in sediments. However, there is some evidence to suggest that ecological effects occur at metal values lower than national (e.g., ANZECC 2000) low guidelines (Hewitt et al., 2009; Rodil et al., 2013; Tremblay et al., 2017), which are higher than international effects range-low guidelines (e.g., Long and Morgan, 1990), based on equivalent principles. Auckland Regional Council have also developed Environmental Response Criteria against which sediment metals concentrations can be compared (Auckland Regional Council, 2004). There are currently no Southland-specific criteria and substantial evidence to supporting the selection of ANZECC (2000) lower guidelines isn't available. On this basis the selection of ISQG-low (ANZECC 2000) is justified to establish thresholds.

The development for bands therefore logically follows as using ISQG-low ANZECC (2000) as the equivalent of a national bottom line. Break points with half (0.5) and quarter (0.25) forming the lower bands. This creates and even spread for bands A to D. This can be done for metals contained within the ANZECC (2000) guidelines: Arsenic (As), Copper (Cu), Cadmium (Cd), Chromium (Cr), Mercury (Hg), Nickel (Ni), Lead (Pb) and Zinc (Zn); these metals being the ones with data available.

Method:

The NEMP (Robertson et al., 2002) provides the method for consistent sampling at 'fine scale sites' there is a relatively rich data set for these variables for Southland. In summary, within a permanent 12 grid at the site, three 3 composite replicates (Replicate 1 = composite quadrats 1 – 3, Replicate 2 = composite quadrats 5 - 7 and Replicate 3 = composite quadrats 9 – 12) are taken of sediment characteristics along with macroinvertebrate cores (quadrats 1-10). Sediment is collected by scraping of the top 2cm of sediment in an area that has not been disturbed

The recommended table is as follows:

Table 8. Attribute state option table toxicants (metals/metalloids) in sediment.

Value	**Ecosystem health**
Freshwater body type	Estuaries and open coast
Attribute group	Southland attribute
Attribute name	Toxicants in sediment
Attribute unit	mg/kg (milligrams per kilogram dry weight)
Attribute band and description	**Numeric attribute state**
	Median[1]
A Very low risk of harm to aquatic species.	≤25% of DGV
B Low risk of harm to aquatic species.	>25 and ≤50% of DGV
C <10% probability of harm to aquatic species.	>50 and ≤100% of DGV
Proposed minimum acceptable state	**100% of DGV**
D >10% probability of harm to aquatic species.	>100% of DGV

The numeric attribute state is based on the ANZECC interim Default Guideline Value (DGV) and in the sediment quality guidelines (2018): https://www.waterquality.gov.au/anz-guidelines/guideline-values/default/sediment-quality-toxicants

As at October 2019, the DGV for toxicants in sediment are:

- Antimony DGV is 2.0 mg/kg
- Arsenic DGV is 20 mg/kg
- Cadmium DGV is 1.5 mg/kg
- Chromium DGV is 80 mg/kg
- Copper DGV is 65 mg/kg
- Lead DGV is 50 mg/kg
- Nickel DGV is 21 mg/kg
- Silver DGV is 1.0 mg/kg

[1]Upto three years of data. Note that background concentrations may be elevated in some areas due to geology.

Measured sediment deposition (Sedimentation rate)

Recommendation: Proposed attribute for a numeric freshwater objective.

Suspended sediment and fine sediment deposition (e.g., particles <0.0625 mm in diameter) are recognised as significant threats to estuaries and coastal environments in many parts of the world (e.g., McKnight, 1969; Woods and Armitage, 1997; Thrush et al., 2004). In some estuaries, particularly shallow intertidally dominated ones, land disturbance in the catchment can result in increased fine sediment mobilisation, resulting in significant mud deposition zones in the upper estuary tidal flats (Robertson et al., 2016). An added consideration is that fine sediment loads are often accompanied by elevated nutrient loads. Their combined effect can cause sediments to become eutrophic (Robertson and Stevens, 2012, 2013, Robertson et al., 2016). The resulting "soft mud/macroalgae cocktail" exacerbates sediment deoxygenation, production of sulphides, with a resultant degraded macrobenthos. High rates of sedimentation are capable of altering estuarine habitats, modifying ecosystem functions and decreasing a broad range of ecosystem services. Extensive national estuary monitoring protocol data from typical NZ shallow tidal lagoon and tidal river and estuaries show that extensive areas of soft mud, elevated sedimentation rates, and high sediment mud contents are commonly associated with a degraded macroinvertebrate community, and particularly so where nutrients are excessive and soft mud areas are overlain with dense nuisance beds of opportunistic macroalgae (Robertson et al., 2016). The early identification and management of excessive sediment deposition is therefore critical for managing eutrophication impacts.

For these reasons, mud is considered a key attribute for management and a useful supporting indicator for the assessment of estuary trophic status (i.e., if soft muds are present then the estuary is more prone to eutrophic sediments, Zaiko et al., 2018).

Sedimentation rate is the most direct measure of catchment sediment load issues. However, there is high variation within and between estuaries for this measure, and long term lag uncertainty regarding sediment delivery and retention in estuaries. There are also issues separating total sediment loads from fine sediment loads, the latter having significantly greater adverse environmental influence in estuaries. These uncertainties can be confounded by changes to hydrology e.g. New River estuary.

Sedimentation rate as a standalone measure would unlikely be sufficient for managing sediment effects in estuaries (Townsend and Lohrer, 2015). However, it may provide benefit as a foundation for a broader framework that includes other elements related to sediment stress, such as suspended sediment concentration (SSC), bed sediment particle size distribution (for mud content), and the areal extent of muddy sediment in an estuary. Such indicators will monitor the infilling rate, whether there has been a shift to finer sediments, and the spatial extent of any changes. Ideally, supporting state variables should therefore include monitoring of plants and animals so that the effects of mud changes on key biota (e.g., macroinvertebrates, fish, seagrass) can be gauged, as well as ensuring water clarity is not adversely impacted by suspended fine sediments.

The main uncertainty for management of sediment exists in estimates of input loads via predicted sediment yields from different land use categories and land management initiatives. Better links between estuarine sedimentation and catchment processes will facilitate a clearer understanding of erosion pathways and thereby improve targeted management responses in estuaries. Sediment deposition metrics need to be related to specific estuary conditions and a sufficient monitoring interval is needed to establish robust trends. Management also requires robust estimates of sediment inputs and the ability to predict change in response to management initiatives. Note that catchment sediment load estimates are difficult and expensive to validate. There are many unknown or poorly defined influencing factors including specific rates of sediment delivery following different types of land disturbance, sediment bed load erosion, sediment retention within estuaries, long-term cycles and influences related to climate cycles (e.g., el Ñino/la ñina), climate change (increased storm intensities), and human flow related changes (e.g., irrigation, flood control, dams).

Measurement options are relatively simple (sediment plates, lidar, bathymetric surveys, dated cores) and can be used to validate models if needed, though these options except sediment plates are expensive. These measures can be employed over entire estuaries or parts of estuaries to provide larger scale and complementary measures for sediment plates. A combination of methods spanning a range of temporal and spatial scales is probably the best option for building more robust management assessments. An important factor in determining methodology is cost vs uncertainty. Note there is no current specific integrated catchment scale sediment programme with specific measures/estimates of sediment supply and sedimentation in Southland.

No national standards exist although Townsend and Lohrer (2015) developed recommendations for estuary guidelines for a default value of 2mm of sediment accumulation per year above the natural annual sedimentation rate (NSR) for the estuary, or part of estuary, at hand. The natural sedimentation rate is defined as the rate under native-forested catchment. It is included in the default guideline value as a baseline to account for estuaries or parts of estuaries with naturally high rates of sedimentation. The NSR is the sedimentation rate for the estuary in its natural state (i.e., pre-human vegetation cover and wetland presence).

To derive 'natural sedimentation rates' radioisotopic dating of sediment cores can be used if available. As of 2019 coring information is available for New River estuary and Waikawa estuary. Grain size analyses rely on standard wet sieving analytical methods; whilst coring can use a combination of dating methods (isotope tracing, caesium-137, lead-210, carbon-dating, pollen-dating). However, these cores may not extend far enough back in time to derive natural conditions.

It is important to note that an estuary with an "overall" average sedimentation rate below a set guideline may still contain multiple sites where the levels are exceeded, while the inclusion of estuary areas with low sedimentation will reduce and 'dilute' the magnitude of the overall sedimentation rate, potentially obscuring instigation of necessary management responses, hence the 'annual sedimentation rate for the estuary, or part of estuary' to address this. This is expected to provide protection to sediment macrofauna in deposition zones from physical impacts (Townsend and Lohrer, 2015). It does not take into account 'indefinite resilience' which refers to the ability of an environment to absorb a given amount of a stressor in perpetuity. Additionally, different estuaries with different catchment geologies and erosion rates have a different natural sensitivity to sediment inputs, and consequently a universal rate of 2mm/yr may not reflect an appropriate management threshold in all estuaries.

On this basis the ETI (Robertson et al., 2016) has proposed estuary thresholds based on the natural sedimentation rate (NSR). The NSR is estimated as the current sedimentation rate (CSR) multiplied by the natural state sediment load (NSL)/current sediment load (CSL). The proposed threshold is the mean sedimentation rate no greater than five times the natural sedimentation rate (i.e., CSR >5 x NSR mm/yr). Catchment models (e.g., CLUES) can be used to estimate NSL and CSL. CSR can also be directly measured using sediment plates and/or bathymetric methods. A more robust approach would be to use hydrodynamic modelling methods to predict estuary retention and to replace NSL and CSL with retained NSL and retained CSL. These options are only applicable for some systems where the necessary data is available and these thresholds require refinement and also need to be related to different estuary classes. Robertson et al., (2016) also proposed thresholds for a proportion of the estuary area with sedimentation rates above a certain level. Again, it is considered that these thresholds need further consideration.

<u>Method:</u>

Guidance for estuary sedimentation plate monitoring can be found in Townsend and Lohrer (2015), and the NEMP (Robertson et al. 2002). Environment Southland has conducted monitoring in various systems over the last 20 years with varying configurations carry out monitoring in the following manner:
General sediment plate installation procedure has changed over time. The installation procedure from Jan-2018 is presented in the "Sediment Plate Standard Operating Procedure" (A396219) and should be followed for all future sediment plate installations. Previous to this time sediment plates were installed with pegs closer to the plate – this created some potential for scouring of the surface by algae wrapped around the pegs; a metal warratah was used under the plate – this prevents movement of the plate and allows it to be found more easily via metal detector if lost.

The installation and measurement procedure can be found for plate sites and transects in the run guides for Estuaries.

Spatial and temporal considerations

Plate sites have been located to best represent deposition areas. With additional sites being added through time. These locations and the sampling design will need to be carefully considered and interpreted because, as previously mentioned; sediment does not accumulate evenly or universally across the spatial extent of an estuary. When monitoring sedimentation, it is recommended to avoid exposed areas and focus on depositional zones and mid-estuarine areas where sediment can potentially accumulate; whilst also considering localised effects such as micro-channelling and wave erosion. It is recommended to avoid averaging sedimentation across all sites in the search for a single univariate statistic for the whole estuary. While this may have appeal (for example, for addressing the question, “does the ‘whole’ estuary exceed or fall below ANZECC guidelines?”), it is difficult to interpret this statistic meaningfully (Townsend and Lohrer, 2015). A better approach is to examine estuarine sites individually, or by category, and then initiate a proportionate management response following a review of the data. It is also important to express confidence intervals on estimates of deposition rates and their temporal/spatial changes.

The duration of monitoring records for measured changes in bed height is also critical in assessing mean annual sedimentation rates, as is the need to relate changes to significant influences e.g., flood frequency and magnitude, within estuary redistribution, land use changes. Guidance from Townsend and Lohrer (2015) indicate the requirement for monthly to quarterly frequency for buried plates. Currently only annual sampling is conducted by Environment Southland. In areas where there is high annual deposition (e.g., 30 mm/y) most methods would be able to detect the exceedance of 2mm/yr quite readily. However, if an estuary has an average sedimentation rate of 3 mm/y, while the sedimentation monitoring method has an accuracy of +/- 20mm, then ~7 years would likely be required to reliably demonstrate change (Townsend and Lohrer 2015). On this basis, a minimum of 5 years’ worth of data should be used for high annual deposition accumulation and assessment of data availability/suitability be made to accompany lower deposition areas.

<u>Thresholds</u>

Sedimentation should not be >2mm + natural sedimentation rate (NSR) of sediment accumulation per year for the estuary, or part of estuary, at hand. Historic data is used to determine if a significant trend can be determined that is below or above 2mm. A trend slope needs to be significant and >[2mm + NSR] to ‘fail’. The estimates for NSR for Southland estuaries that could be applied to Southland estuaries are <1mm/yr for SIDEs and <0.2 mm/yr for SSRTREs (Roberts and Ward 2020).

The recommended bands for sedimentation rate are shown in Table 9.

Table 9. Attribute state option table for sedimentation rate.

Value	Ecosystem health
Freshwater body type	Tidal lagoon estuaries (SIDES) and tidal river estuaries (SSRTRE)
Attribute group	Southland attribute
Attribute name	Sedimentation rate
Attribute unit	mm/year (millimetres per year)
Attribute band and description	**Numeric attribute state**
	Slope[1]
Pass No discernible effect on ecology.	≤2 + NSR[2]
Fail Adverse effects on ecology.	>2 + NSR[2]
[1] Trend slope tested to determine if statistically significantly different (90%) from 2 mm/y + natural state rate. Slope determined from all plate data points for the site. Minimum of 5 years to be used to determine slope. Sites are >25% mud content. Rate may be calculated from post significant erosion events and periods. [2]Natural State rate; for SSRTRE 0.2mm/yr and SIDES 1mm/yr.	

Note that Townsend and Lohrer (2015) propose a default guideline value of 2mm per year above the natural sedimentation rate. This is based on evidence that shows adverse effects on ecology. One of the supporting studies (Lohrer et al. 2004a) assessed the thickness of mud deposit on the invertebrate community and found negative impacts from increasing deposited mud thickness on diversity (number of taxa) and number of individuals. This would suggest that that the threshold (pass/fail) proposed may be towards the top of any banding. The Lohrer et al. (2004a) study experimented with deposits up to 7mm in thickness, thus the lowest banding is not likely to be more than 10mm/yr for SIDEs or SSTREs. It is also worth noting that there will be multi-stressors (such as pH, nutrient concentration and organic matter content) having an effect. In the absence of a banding system this attribute may be more suited as a narrative and should be considered further within social considerations.

Water Quality variables

Water nutrient concentrations (N, P, C)

Recommendation: Needs development.

Dissolved nutrients are monitored by Invercargill city council in New River estuary only (WQ report for New River (1991 -2015) - A329715). However, in general if sampling is to occur the water column needs to monitored at representative sites measuring dissolved inorganic and organic nutrients. This will likely need to include bottom waters and consider benthic/pelagic coupling and nutrient flux to be meaningful. The hydrodynamics of the particular estuary will also need to be out into context which will potentially involve modelling and river data to be of relevance. Each estuary will be different and require specific data and thresholds that have yet to be developed.

Water Chlorophyll-a (Chlorophyll-*a*)

Recommendation: Proposed attribute for a numeric freshwater objective.

Water chlorophyll-*a* is a photosynthetic pigment which serves as a proxy for phytoplankton biomass (Zaiko et al., 2018). Phytoplankton are a food source for many estuarine species and play an important role in estuarine and coastal nutrient cycling (Cloern et al., 2014). Chlorophyll-*a* can increase with nutrient loading from upstream sources due the proliferation of phytoplankton; high concentrations of chlorophyll-*a* is a sign of eutrophication (Boyer et al., 2009).

However, it can be difficult the separate the response of chlorophyll-*a* to different stressors. For example, as phytoplankton and other primary producers require light to photosynthesis, chlorophyll-*a* can respond negatively to lower light levels due to sediment loading (Cloern et al., 2014). Further, chlorophyll-*a* concentration is spatially and temporally variable within and between estuaries. Smaller estuaries with reduced tidal flushing may be more susceptible to measurable changes in chlorophyll-*a* (i.e., phytoplankton biomass) compared to larger frequently flushed estuaries (Zaiko et al., 2018).

There is a wealth of experience and studies that exist globally supporting phytoplankton chlorophyll-*a* as a reliable response indicator (e.g. Sutula, 2011), though residence times needs to be accounted for. Robertson et al., (2016) recommended use of chlorophyll-*a* as a primary symptom indicator for subtidal dominated estuaries (residence time weeks rather than days), and ICOLLs during their closed phase. Chlorophyll-*a* is used in the national objective framework (NOF; NPSFM 2017) for Lake attributes and was determined using the literature for the Tropic Level Index to determine break points. There is general consensus that tidal lagoon type estuaries do not have sufficiently long residence times of <3 days for water column primary productivity (reflected by Chlorophyll-*a*) to manifest. This is despite there being evidence to the contrary (Robertson et al., 2017).

<u>Threshold:</u>

The Estuary trophic Index (ETI, Robertson et al., 2016) proposed using thresholds which are based on Revilla et al., (2010; Table 10) who determined condition breakpoints (poor/ moderate/ good/ high/ reference) for Basque Country Cantabrian (Spain) estuaries under the Water framework directive (WFD). The key strength in the WFD approach is that they have a large data set contributing to their results and so their thresholds are well supported. In NZ we generally have insufficient national scale sample replication to get reliable numbers or to build local chlorophyll-*a* thresholds. However New River estuary has long term extensive sampling conducted by Invercargill City Council.

The estuaries are separated into salinity classes oligo/meso/polyhaline estuaries which accounts for some of the variability in susceptibility of estuaries (Revilla et al., 2010).

Table 10. Excerpt from Revilla et al. 2010.

Water category	Salinity stretch	Reference condition (μg L^{-1})	High/ Good (μg L^{-1})	Good/ Moderate (μg L^{-1})	Moderate/ Poor (μg L^{-1})	Poor/ Bad (μg L^{-1})
CW	Euhaline	<u>2.33</u>	<u>3.5</u>	<u>7.0</u>	10.5	14.0
TW	Euhaline	2.67	4.0	8.0	12.0	16.0
TW	Oligo/Meso/Polyhaline	5.33	8.0	12.0	16.0	32.0

Note: Oligohaline 0.5-5ppt salinity, Mesohaline >5-18ppt, Polyhaline >18-30ppt and Euhaline >30ppt salinity.

Robertson et al., (2016) did not form bands exactly according to Revilla et al., (2010) so reformulating these breakpoints is more appropriate than replicating directly.

The recommended bands for phytoplankton are shown in

Table 11:

Table 11. Attribute state option table for phytoplankton in water.

Value	Ecosystem health		
Freshwater body type	Estuaries and open coast		
Attribute group	Southland attribute		
Attribute name	Phytoplankton		
Attribute unit	mg chlorophyll-*a* /m^3 (milligrams chlorophyll-*a* per cubic metre)*		
Attribute band and description	**Numeric attribute state**		
	Open coast[1]	Estuaries (saline)[1]	Estuaries (less saline)[1]
	90th percentile[2]		
A Estuary ecological communities are healthy and resilient, similar to natural reference conditions.	≤3.5	≤4	≤8
B Estuary ecological communities are slightly impacted by additional algal and/or plant growth arising from nutrient levels that are elevated above natural reference conditions.	>3.5 and ≤7.0	>4 and ≤8	>8 and ≤12
C Estuary ecological communities are moderately impacted by additional algal and plant growth arising from nutrient levels that are elevated well above natural reference conditions. Reduced water clarity is likely to affect habitat available for native macrophytes.	>7.0 and ≤10.5	>8 and ≤12	>12 and ≤16
Proposed minimum acceptable state	**10.5**	**12**	**16**
D Estuary ecological communities have undergone or are at high risk of a regime shift to a persistent, degraded state (without native macrophyte / seagrass cover), due to impacts of elevated nutrients leading to excessive algal and/or plant growth, as well as from losing oxygen in bottom waters.	>10.5	>12	>16
*Chlorophyll-*a* from representative sites of estuary water column. [1] Coastal waters and saline estuaries are defined by having salinity >30 ppt and less saline estuaries <30 ppt salinity. [2] based on monthly measurements over 3 years.			

To date only New River estuary has data available collected (1991-2015) and shown to have high chlorophyll-*a* measures over long period of time (Robertson et al., 2017b). Water quality samples were collected correspondingly and show a clear decline in water quality across the system over time and a severe decrease in more recent years, especially at Ōreti beach which has exceptionally high concentrations for a beach coastal environment (Robertson et al., 2017b).

Methods:

Methodology for sampling and analysis is described in Revilla et al., (2010).

Habitat Variables

Areal extent of seagrass / Percent cover of seagrass

Recommendation: Proposed attribute for a numeric freshwater objective.

Seagrass (*Zostera muelleri*) provides critical habitat for various life stages of Kai species, and is also often associated with some shellfish species (Cornelisen et al., 2017). It is vulnerable to excessive nutrients and increased turbidity in the water column that can lead to light limitation; and excessive macroalgal growths associated with nutrient loading that can cause smothering.

Seagrass plays an important role in NZ estuarine ecology and is a well-documented as a keystone species. They attenuate and assimilate nutrients and sediment, and provide high value habitat for a wide range of biota (Robertson et al., 2016). The presence of extensive beds in good condition generally indicates low/moderate nutrient and mud inputs, combined with good water clarity, whereas die-off and absence is generally indicative of excessive nutrient and mud inputs and eutrophic conditions or poor water clarity (Robertson et al., 2016). Suspected causes of decline need to be verified by additional appropriate measures.

Seagrass is a useful indicator of state and biodiversity species. However, there are difficulties in defining natural state coverages as there is likely high natural variability between estuaries and years. There are multiple drivers for seagrass loss (e.g. sediment smothering, water clarity, N toxicity, flood or wave scouring, macroalgal or epiphytic smothering, grazing (fish, swans), temperature and desiccation). Long term cycles of expansion and contraction of sea grass bed area have been recorded in northern estuaries. This is not well understood but could affect the interpretation of changes in cover over time (Cornelisen et al., 2017). This all makes it difficult for use as an attribute suitable for catchment scale management i.e. linking to catchment aspects to be managed and determining eutrophication and non-eutrophication related stressors. It is unrealistic to expect a reliable condition gradient of expected seagrass cover that matches nutrient and/or sediment loads in the various NZ estuary classes (Robertson et al., 2016).

Threshold:

Despite this variability in response to nutrient and sediment loads, it is appropriate to develop estuary-specific thresholds using data on the estuary's seagrass cover prior to catchment development as the reference threshold, i.e., set thresholds based on the extent cover occurs naturally in a particular estuary (Robertson et al., 2016). If pre-development seagrass cover is unknown, then best estimates can be obtained from inferences based on known cover in similar estuaries with minimal catchment development, or early aerial photographs where available. In absence of such information the earliest known coverage can be applied i.e. earliest monitoring information.

Robertson et al., (2016, Table 12) recommended thresholds based on deviations from a natural state reference condition, or from a measured baseline. The magnitude of the deviations is based on expert opinion rather than strong evidence and should be considered interim. They should be used in association with other indicators (e.g. mud, macroalgae and sediment conditions).

Bands could be set based on % loss, noting that % changes can change very quickly when small areas are being considered. This places a very high level of emphasis on mapping accuracy, establishing a baseline, and requires clear definitions on what terms like 20% cover means. Further consideration should also be given to shoot vs root biomass measures and how these reflect overall seagrass bed health.

Table 12. Seagrass bands, excerpt from Robertson et al. 2016.

Table 12. Seagrass interim thresholds for screening shallow lagoon type estuaries.

Band	A	B	C	D
Ecological Quality	No stress caused by the indicator on any aquatic organisms.	A minor stress on sensitive organisms caused by the indicator.	Moderate stress on a number of aquatic organisms caused by the indicator exceeding preference levels for some species and a risk of sensitive macroinvertebrate species being lost.	Significant, persistent stress on a range of aquatic organisms caused by the indicator exceeding tolerance levels. A likelihood of local extinctions of keystone species and loss of ecological integrity.
SAV Extent % of Estimated Natural State Cover (ENSC)	100 % of ENSC	>95-99% of ENSC	85-95%of ENSC	<85% of ENSC

Method:

There are some possible methodological issues regarding the consistent mapping of density and biomass with available data. There is strong reliance on aerial photography which may not always be available and up to date for Southland. Without further investment Southland will be constrained to using the historic methods that have been applied to date. Ideally research should be undertaken to develop a model that predicts the potential of any NZ estuary in its natural state for high density seagrass growth, by accounting for both eutrophication and non-eutrophication related variables. Such a model would be capable of predicting numeric nutrient load criteria to support healthy seagrass beds. Current data and methods have applied a >20% coverage approach (Robertson, 2017b).

The recommended bandings are shown in Table 13.

Table 13. Attribute state option table for seagrass.

Value	Ecosystem health
Freshwater body type	Tidal lagoon estuaries (SIDES) and tidal river estuaries (SSRTRE)
Attribute group	Southland attribute
Attribute name	Seagrass coverage (>20%)
Attribute unit	% of estimated natural state cover[1]
Attribute band and description	**Numeric attribute state**
	Annual assessment
A Minimal stress on aquatic organisms.	100[2]
B Minor stress on sensitive organisms.	>95-99
C Moderate stress on a number of organisms with risk of some being lost.	85-95
Proposed minimum acceptable state	**<85**
D Significant, persistent stress on a range of organisms. macroinvertebrates. A likelihood of local extinctions of keystone species and loss of ecological integrity.	<85

[1]In the absence of known natural state cover earliest monitoring information.

[2] If baseline is establish for more recent times gain is seagrass may exceed 100%.

Composition and areal extent of dominant saltmarsh types

Recommendation: Needs development. Propose as attribute for a narrative freshwater objective.

Saltmarsh is a useful indicator of state and biodiversity. However, there are difficulties in defining natural state coverages, likely high natural variability between estuaries, and methodological issues regarding consistent mapping. There is likely to be little response to saltmarsh in response to sediment or nutrient reduction at a catchment scale unless there are extreme inputs currently causing saltmarsh burial. Sea Level Rise (SLR) and land drainage are the most obvious pressures to consider, with secondary aspects such as terrestrial weed incursions and the inability of saltmarsh to migrate inland in response to SLR needing consideration. A likely best case narrative is "no further loss from an established baseline due to sediment or nutrient drivers". Bands can be set based on % loss, noting that % changes can change very quickly when small areas are being considered. There is historic monitoring information available for this attribute which could serve as a baseline for change.

Areal extent of opportunistic macroalgae (EQR calculated from the Opportunistic Macroalgal Blooming Tool)

Recommendation: Proposed attribute for a numeric freshwater objective.

Blooms of rapidly growing macroalgae can have deleterious effects on intertidal and shallow subtidal communities, and cause an undesirable imbalance with effects such as: blanketing of the surface causing a hostile physico-chemical environment in the underlying sediment, sulphide poisoning of infaunal species, anoxic gradient at the water sediment interface, effects on birds including changes in the feeding behaviour of waders, smothering of seagrass beds (Duarte 1995, Taylor et al., 1995, Valiella et al., 1997, Sutula et al., 2012), excessive algal growths, or rafts of floating or detached weed causing interference with water users, aesthetic effects such as nuisance odours, or deposition in bathing waters. Where excessive macroalgae cause extreme sediment anoxia (measured by redox potential) there is an accompanying exclusion of normal communities of benthic macrofauna (e.g., Grizzle and Penniman, 1991); increased production of sulphides which can be toxic to rooted macrophytes (Lamers et al., 2013, Holmer and Bondgaard, 2001; Viaroli et al., 2008; Geurts et al., 2009; Green et al., 2014), and release of dissolved phosphorus and ammonium that exacerbate eutrophication (e.g., Søndergaard et al., 2003). Opportunistic macroalgae are species that survive well in conditions in which other species often struggle to survive or compete (Borum and Sand-Jensen, 1996). Blooms in NZ estuaries principally contain species of green algae *Ulva* (this includes taxa formerly known as *Enteromorpha*) and *Cladophora*, red algae *Gracilaria*, and brown algae (e.g. *Ectocarpus*, *Pilayella*, *Bachelotia*). These bloom-forming species are a natural component of intertidal ecosystems (Adams, 1994), but they only grow to bloom proportions when nutrient levels are elevated (Sutula et al., 2011) and sufficient light reaches the bed of the estuary (or the water column where macroalgae are suspended). As a consequence, they generally only reach nuisance conditions in shallow estuaries, or the margins of deeper estuaries. The macroalgal response to nutrient loads generally increases with water residence times (Painting et al., 2007), either of the whole estuary (as is often the case for many NZ short residence time estuaries), or part of the estuary (e.g. a poorly flushed upper estuary arm where nutrient-rich muds accumulate), or in 'backwaters' where drifting suspended macroalgae can accumulate (e.g. Avon-Heathcote Estuary; Bolton-Ritchie and Main, 2005).

There is some evidence this response may also be significantly attenuated by the presence of fringing saltmarsh, due to reductions in nutrient loading through processes such as denitrification (Valiela et al., 1997). Other factors that can influence the expression of macroalgal growth are the presence of suitable attachment strata, and physical and hydrodynamic conditions e.g. temperature (desiccation), fetch (wind driven waves), currents (scouring) e.g. Hawes and Smith (1995).

The WFD-UKTAG (Water Framework Directive – United Kingdom Technical Advisory Group, 2014) approach for opportunistic macroalgal condition is a relatively comprehensive rating tool that is currently used on NZ estuaries and is recommended for use in the ETI (Robertson et al., 2016). It is supported by extensive studies of the macroalgal condition in relation to ecological responses in a wide range of estuaries. The Opportunistic Macroalgal Blooming Tool (OMBT) is a comprehensive 5 part multimetric index that incorporates species composition, macroalgal cover, biomass, and entrainment within sediment to calculate an ecological quality rating (EQR). It is currently used in broad scale assessment of estuary condition by many regional councils in NZ.

The OMBT has been developed to classify data over the maximum growing season so sampling should target the peak bloom in spring-summer (Oct-March), although peak timing may vary among water bodies, therefore local knowledge is required to identify the maximum growth period (Zaiko et al., 2018). Sampling is not recommended outside the spring-summer period due to seasonal variations that could affect the outcome of the tool and possibly lead to misclassification; e.g., blooms may become disrupted

by stormy autumn weather and often die back in winter. Sampling is best carried out during spring low tides in order to access the maximum area of the Available Intertidal Habitat (AIH), (Zaiko et al., 2018).
The OMBT has been developed with thresholds to define ecological quality status based on extensive European data. The NZ macroalgal data assessed to date are largely consistent with the established UK-WFD thresholds, but the threshold for significant sediment related impacts appears to occur at a lower macroalgal biomass in NZ than in the UK-WFD (Wriggle, unpublished - Zaiko et al., 2018). Because the OMBT is designed to allow for specific changes such as this to be incorporated, NZ specific thresholds can easily be incorporated. However, a full assessment of available data is needed to apply this in a nationally consistent manner. Currently, available data are scattered throughout individual reports and there has been no collation of national data. Like most sampling there is also potential for variation in the application of the sampling design, particularly in terms of spatial extent and number of replicates, ensuring representative sampling sites are selected, and that criteria used to set thresholds of impact reflect the entire gradient of response to nutrient loads (low/pristine to high/degraded).

Threshold:

Persistent blooms of macroalgae can have negative impacts on both ecological and aesthetic values, and can be indicative of excessive nutrients and/or deteriorating sediment conditions. The Opportunistic Macroalgal Blooming Tool (OMBT - WFD-UKTAG 2014) thresholds have received extensive recent review and are considered highly appropriate for use in NZ's dominant estuary classes (i.e. shallow, intertidal dominated estuaries where macroalgae could reach nuisance levels), because they include both biomass and spatial measures.

This attribute is probably the most directly relevant and well supported of the attributes for SIDEs relating to eutrophication but does rely on a series of related state variables for its calculation.

The OMBT approach uses multiple metrics (Table 14) to calculate the index band for ecological quality rating (EQR), (Table 15).

Table 14. Multimetrics used to calculate ecological quality rating (Robertson et al., 2016)

OMBT Quality Status	High	Good	Moderate	Poor	Bad
EQR (Ecological Quality Rating)	≥0.8 - 1.0	≥0.6 - <0.8	≥0.4 - <0.6	≥0.2 - <0.4	0.0 - <0.2
% cover on Available Intertidal Habitat (AIH)	0 - ≤5	>5 - ≤15	>15 -≤25	>25 - ≤75	>75 - 100
Affected Area (AA) of >5% macroalgae (ha)*	≥0 - 10	≥10 - 50	≥50 - 100	≥100 - 250	≥250
AA/AIH (%)*	≥0 - 5	≥5 - 15	≥15 - 50	≥50 - 75	≥75 - 100
Average biomass ($g.m^2$ wet weight) of AIH	≥0 - 100	≥100 - 500	≥500 - 1000	≥1000 - 3000	≥3000
Average biomass ($g.m^2$ wet weight) of AA	≥0 - 100	≥100 - 500	≥500 - 1000	≥1000 - 3000	≥3000
% algae >3cm deep in sediment (entrained)	≥0 - 1	≥1 - 5	≥5 - 20	≥20 - 50	≥50 - 100

*N.B. Only the lower EQR of the 2 metrics, AA or AA/AIH is used in the final EQR calculation.

Table 15. Ratings for EQR (Robertson et al., 2016)

Band	A	B	C	D
Ecological Quality	Ecological communities (e.g. bird, fish, seagrass, and macroinvertebrates) are healthy and resilient. Algal cover <5% and low biomass (<50gm^{-2}wet weight) of opportunistic macroalgal blooms and with no growth of algae in the underlying sediment. Sediment quality high (e.g. RPD in Band A).	Ecological communities (e.g. bird, fish, seagrass, and macroinvertebrates) are slightly impacted by additional macroalgal growth arising from nutrients levels that are elevated. Limited macroalgal cover (5-20%) and low biomass (50-200gm^{-2}wet weight) of opportunistic macroalgal blooms and with no growth of algae in the underlying sediment. Sediment quality transitional (e.g. RPD in Band B).	Ecological communities (e.g. bird, fish, seagrass, and macroinvertebrates) are moderately to strongly impacted by macroalgae. Persistent, high % macroalgal cover (25-50%) and/or biomass (>200-1000g/m^{2}wet weight), often with entrainment in sediment. Sediment quality degraded (e.g. RPD in Band C).	Ecological communities (e.g. bird, fish, seagrass, and macroinvertebrates) are strongly impacted by macroalgae. Persistent very high % macroalgal cover (>75%) and/or biomass (>1000g/m^{2}wet weight), with entrainment in sediment. Sediment quality degraded with sulphidic conditions near the sediment surface(e.g. RPD in Band D).
Open Estuaries: EQR (Ecological Quality Rating)	≥0.8 - 1.0	≥0.6 - <0.8	≥0.4 - <0.6	0.0 - <0.4

There are likely to be issues with regard to catchment level management due to potential lag times in the response to nutrient changes, particularly if there are sediment bound nutrient issues (legacy effect). Expression may be influenced by other drivers e.g. flood scouring, channel flushing, wind-driven waves, temperature. The drivers of tipping points that result in macroalgal expression are not always predictable. New River estuary was theoretically saturated with nutrients but not expressing significant problems before rapid deterioration (Robertson et al., 2017b). This indicates co-driver/s to nutrients e.g. temperature and stable growing conditions, may be necessary to tip to significant nuisance conditions.
Currently, the data supporting a relationship between macroalgae and estuary trophic status in NZ estuaries is limited to a relatively small number of studies, but all confirm adverse impacts to sediment physico-chemical and biota along similar lines to those found in overseas studies (Zaiko et al., 2018). In order to provide a more robust basis upon which to base the metrics used in the OMBT (WFD-UKTAG 2014) ecological quality rating for macroalgae, it is recommended that the ecological response thresholds for macroalgae be more thoroughly assessed, over all estuary classes (but particularly those prone to macroalgal blooms i.e., shallow, intertidal dominated estuaries and ICOLLs). The studies should focus on opportunistic macroalgal effects on biota (e.g., macroinvertebrates, fish, seagrass), and physico-chemical parameters (e.g., sediment redox potential, sulphur, organic carbon, nutrients and bacteria).

Note that the EQR approach does not take into account the reduction in macroalgal biomass evident in parts of the Aparima (Jacobs river) and Waihōpai (New River) estuaries that is likely driven by extreme sediment anoxia and high sulphide levels (Robertson et al., 2017b).

<u>Method:</u>

There are no NZ monitoring standards for macroalgae. Macroalgae was also not included as a primary symptom of eutrophication in the NEMP so no specific methods were developed for its enumeration. While it has commonly been recorded where it is a dominant surface cover, NEMP spatial mapping does not include the measures of estuary wide percentage cover, biomass or entrainment that are required by the OMBT. Broad scale spatial mapping described in the NEMP requires updating to reflect subsequent advances in the protocol. The OMBT has been developed with thresholds to define ecological quality status based on extensive European data. This includes a full description of the metrics used and calculations required. The measures are taken annually between October and early March (Feb-March in Southland) to capture peak growth. % cover, biomass, entrainment of macroalgae are all measured (Robertson et al., 2016). More detail in methods can be found in WFD-UKTAG (2014) and Stevens (2018). Supporting indicators are an important subcomponent of any measurements and are required to understand the implications of macroalgal expression. In any particular substrate type, sediment oxygenation, organic content and sediment nutrient concentrations are all key measures in assessing the likely impact and duration of macroalgal growths.

The WFD-UKTAG (2014) recognize the specialist skillset needed to maintain consistency in macroalgal monitoring using the OMBT, and undertake this work using a specialist provider at a national level, rather than using multiple regional providers. For Southland, the same provider has conducted the monitoring throughout. Though some caution may be applied to first collection of data. Adequate training is required to consistently assess and enumerate broad scale macroalgal condition. Field sampling requires the ability to consistently define representative patches of macroalgal cover and biomass, and balance replication needs with practical considerations in terms of sampling within the limited tide window. However, there are opportunities to explore using drones and remote sensing tools; these are likely compliment rather than replace the need for field sampling though. Further development of field measurement standardisation of biomass, percentage cover, and entrainment for NZ estuaries would be beneficial and allow interregional comparison. The recommended table is as follows:

Table 16. Attribute state option table for Macroalgae (EQR).

Value	Ecosystem health
Freshwater body type	Tidal lagoon estuaries (SIDES) and tidal river estuaries (SSRTRE)
Attribute group	Southland attribute
Attribute name	Macroalgae
Attribute unit	EQR (Ecological Quality Rating)
Attribute band and description	**Numeric attribute state**
	Index[1]
A Ecological communities are healthy and resilient. Algal growth of opportunistic species low.	≥0.8
B Ecological communities are slightly impacted. Algal growth of opportunistic species limited.	≥0.6 and <0.8
C Ecological communities are strongly impacted. Algal growth of opportunistic species high.	≥0.4 and <0.6
Proposed minimum acceptable state	**0.4**
D Ecological communities are strongly impacted. Algal growth of opportunistic species very high.	<0.4
[1] Described and explained by Estuary Trophic Index Tool 2 (Robertson et al. 2016).	

Macrofauna variables

Biodiversity of macrofauna (measures incl. biodiversity, multivariate indices, trait based index)/ Traits based macrofauna index/ Evenness of macrofauna / Multivariate macrofauna indices

Recommendation: Needs development.

Due to their relatively sedentary life style and sensitivity to changes in pressures, soft sediment macrofauna can indicate and integrate complex environmental conditions and therefore considered useful for representing benthic community health in response to contaminants, nutrients, organic enrichment, deposition rates, turbidity and changes in muddiness (if representative sites are surveyed), (Zaiko et al., 2018). As an estuary progresses along the gradient of increasing eutrophication and muddiness, the benthic macroinvertebrate community responds to lowering oxygen and increasing toxicity by shifting towards smaller, more stress tolerant species. These are not as efficient at bioturbation, which limits oxygen penetration into the sediments and effectively minimise the zone of coupled nitrification/denitrification in the sediments (Pearson and Rosenberg 1978, Sutula, 2011). They are also often less efficient in providing other ecosystem services, e.g., secondary production, biofiltration or provisioning. However, developing macrofauna-based attributes might be complicated by the high variability of natural conditions in estuaries and multivariate response of the macrofauna communities (Zaiko et al., 2018).

Sample sieving and specimens picking approaches are rather consistent; the organisms are sorted under microscope, identified to the lowest taxonomic level possible, enumerated. However, "lowest taxonomic level" can vary significantly among the labs and taxonomists. Some councils identify taxa internally, while small and cryptic animals sent to external taxonomic experts. Many taxa are identified to relatively broad levels of taxonomic resolution (Family, Class, Order or even Phylum), however, this is consistent with international practice. All of Environment Southland data is identified by an external taxonomist. Since 2014, some monitoring protocols are following QA procedures developed by Hewitt et al., (2014) for regional councils. In addition, care must be given to where and how representative samples should be collected i.e. from most impacted 10% of the estuary, randomised set of samples, replicates or composites, or a targeted (subjective) sample collection from "representative" habitat. There are issues of taxonomic sufficiency for some groups (e.g. Amphipoda) which are poorly described and commonly (but variably) grouped, variance also occurs between providers with regard to resolution of identifications, and a lack of national consistency in data quality and QAQC. This all results in limited ability to pool regional data collectively, thus hampering development of these indices.

Within NZ, there have been several approaches to the development of macroinvertebrate/estuary condition relationships based on the response of NZ species to estuarine variables. The most common environmental variables for which taxa responses have been identified are: mud content (Norkko et al., 2002; Robertson et al., 2015), heavy metals (Rodil et al., 2013), and organic enrichment (Robertson, 2013; Robertson et al., 2015). However, such sensitivity analyses potentially lack strong regional transferability and are limited in terms of the number of taxa with assigned ratings. As such, their use in assessing estuary condition at any particular site needs to be supported by information that indicates that: i. the estuary in question fits within the original estuary classification used to produce the ratings, ii. that due regard is given to taxa that have not yet been rated for sensitivity, iii. that the ratings are only used to assess sensitivity to the original assessment variable e.g. sediment mud content, and iv. that appropriately representative gradients were used in the original assessment for the estuary now being assessed.

These indices result in a single number which summarises the complex estuary condition and is statistically supported by a wide range of physical, chemical and biological measures. The development of these indices reflects the facts that biological communities are a product of their environment, and organisms can be grouped according to different habitat preferences and pollution tolerance. Most of the estuarine biotic indices are only used in a limited way at present, but AMBI and M-AMBI, BQI (and its various adaptations), B-IBI, and Infaunal Trophic Index (ITI) are currently widely used throughout the world (Borja et al., 2012). However, a recent review (Borja et al., 2012) concluded that no single biotic index can correctly assess the estuary macroinvertebrate condition, and that a multi-criteria approach is favoured. Use of a multi-metric approach is required to gain a true indication of the factors driving a particular macroinvertebrate assemblage, particularly the inclusion of indicators of eutrophication and toxicity. The use of any specific index should reflect the question being asked. For southland a comprehensive assessment of the different indices and their application has not been carried out to date.

Thresholds

For eutrophication-related macrofauna response, thresholds have been recommended for the RI_AMBI for the SIDE type estuaries (ETI Tool 2; Robertson et al., 2016a) and on national scale (Robertson et al., 2016b); and also within the Auckland region for BHM (Anderson et al., 2006). However, threshold values may need to be calibrated for different stressors/ specific estuary/ estuary class/ bioregion to ensure that differences in natural variability are accounted for. Deriving standardized thresholds is impeded by the high natural variability of macrofauna communities in estuaries, limited data on reference conditions and stressor-specific response on a national scale (Berthelsen et al. in press; Zaiko et al., 2018).

Given that taxonomy and QA/QC challenges are resolved it is recommended that an assessment explore using different indices appropriate to the three major estuary stressors in Southland i.e. muddiness, eutrophication and toxicity and their performance. A multi-criteria approach using physical, chemical and biotic indicators is recommended. The approach should use quantitatively derived estuarine sensitivities for NZ taxa supported by results analysing changes in species richness, individual species abundances (particularly in relation to their mud/enrichment tolerance groupings), mud, TOC, metals concentrations, and redox potential. In particular analyses should consider:

- *TOC concentration:* versus NZ hybrid AMBI; versus species richness; versus macroinvertebrate community similarity.
- *Mud concentration:* versus NZ hybrid AMBI; versus species richness; versus macroinvertebrate community similarity.
- If metal (or some other toxin) concentrations from anthropogenic sources are elevated above biologically stressful levels, then include these data as a potential explanatory variable. Though there is likely to be limited representative data set which has elevated metals/metalloids across different estuaries.
-

The analyses should consider the applicability of currently available and developed indices from multiple sources, nationally and internationally.

Combined Index variables

Estuary Trophic Index Score (ETI)

Recommendation: Needs development.

The ETI is a screening tool for assessing estuary state. It is not intended to be used as a management tool because there will be residual uncertainty in scores that precludes it from being a stand-alone index for assessing catchment management changes. This partly relates to the method accuracy of individual metrics (most are discussed above), as well as potential time lags between catchment changes and observed responses. At a high level, maintaining existing bands should be a minimum objective, but aspirational targets can also be set. The ETI requires many of the thresholds to be further developed and methodological issues to be addressed more comprehensively. Despite this, the ETI is a powerful tool for conveying multiple stressor responses within a single index and has the added benefit in that it can be used as a predictive tool to indicate what type and magnitude of changes are needed to alter existing state. It is not fully developed for all estuary classes and there are limitations with the input data available at a national scale (e.g. Coastal Explorer) that reduces the confidence in regional and national comparisons that can currently be made. It has benefits in that changes and additions can be made to the indicators and bands used in the tool without compromising its overall score. However, this means that past scores may change as a result of improvements to the thresholds underpinning the ETI.

The ETI requires supporting metrics to calculate and as some of these variables need development it is recommended at this time that further development is needed before the Index is applied as a Southland attribute.

Gross Eutrophic Zone

Recommendation: Proposed attribute for a numeric freshwater objective.

As highlighted in previous sections (Areal extent of opportunistic macroalgae and depth to aRPD), excessive macroalgae can cause extreme sediment anoxia. The resultant conditions displace most sensitive estuarine animals (including shellfish), and sediment anoxia will cause the release of Phosphorus bound in the sediment under oxic conditions. This release from Fe-oxyhydroxides under anoxic conditions can also contribute to algal growth. More bioavailable Ammonia will also be realised from these sediments, which is much more readily available to fuel macroalgal growth. A cycle of increasing habitat deterioration can then establish that is likely to be difficult to reverse. This unleashes the potential for reinforcing processes and continued legacy issues. These conditions are most likely to be present in the sheltered tidal flats of an estuary which are often those most favourable for the growth of high value seagrass habitat (Stevens, 2018a).

Gross eutrophic conditions should not be present in short residence time estuaries like New River (SIDE), Jacobs River (SIDE) and Fortrose (SSRTRE) (Stevens, 2018a&b). The persistent and extensive presence of these areas in estuaries provides a clear signal that the assimilative capacity of the estuary is being exceeded (Zaiko et al., 2018). They represent the physical expression of problem conditions that are likely to be hard to reverse, and may become self-reinforcing due to feedback loops promoting anoxic release of sediment bound nutrients so are ideally characterised early and limited to very small areas.

These zones of extreme sediment degradation (also called "Gross Nuisance Areas (GNAs[3])" – Robertson et al., 2016) are currently used in the ETI as an indicator of excessive opportunistic macroalgae (including epiphytes) that are associated with anoxic sediment. Widespread monitoring of NZ shallow estuaries indicates that excessive macroalgal cover in poorly flushed parts of these estuaries can result in GEZ (Zaiko et al., 2018). Similar GNAs occur in shallow coastal lagoons or ICOLLs where conditions are not too turbid e.g., Waituna Lagoon.

In extreme cases sediment toxicity conditions are unsuitable for the growth of macroalgae growth, as is evident in parts of New River estuary (Stevens, 2018).

<u>Threshold:</u>

A review of monitoring data from 25 typical NZ estuaries (shallow, short residence time estuaries) supports an opportunistic macroalgal biomass "exhaustion" threshold of approximately 1000-2000g.m^{-2} (ww) above which there was a major shift in the chemistry of the underlying sediment to surface anoxia (aRPD at the surface), elevated TOC (>1.5%) and a degraded macrofaunal community (Wriggle Coastal Management database 2009-2014; Zaiko et al., 2018). Such conditions have been used to identify GEZs. Based on the measured detrimental impact on macrofauna in NZ tidal lagoons, it has been estimated that if GEZs cover >15% of the estuary area or >30ha, then estuary ecological condition is seriously impaired (Zaiko et al., 2018). These areal thresholds of an estuary should be considered interim and further development is likely needed.

Due to the possibility that GEZ areas can become self-reinforcing there is a rationale that any in a system is problematic and shouldn't be there. As these conditions are potentially irreversible (or at the least very

[3] For clarity these areas will be referred as gross eutrophic zones (GEZ)

difficult and costly) there needs to be early warning sign built in. This is where the ability to use other metrics with earlier signs, such as EQR, are fundamentally important. GEZ should not be considered a stand- alone metric for managing estuaries.

In principle once a threshold has been crossed consideration should be given to the persistency of that exceedance. Exceedance will likely be clearly enduring through time or fluctuating around the threshold. This fluctuation may indicate that marked ecological improvement of a system is more achievable and therefore greater consideration for efforts and approaches may differ to otherwise. It also poses some policy, social and management considerations as how to address target compliance. There is no clear scientific approach to addressing this challenge.

Robertson et al., (2016) undertook an exploration of the relationship between gross eutrophic zones and catchment load. This preliminary work provided compelling evidence for a catchment load ecological condition connection i.e. an N areal load gradient clearly exists within and between Southland estuaries against which eutrophic expression can be assessed. The relationship for sediment and Phosphorus is much less promising, which is to be expected as fine sediment (as associated Phosphorus) has been accumulating in the estuaries over time and is therefore not solely dependent on recent fine sediment loads. Further development is needed to progress this relationship, especially if the intention is for application to catchment management (Robertson et al., 2016; Ward and Roberts, 2018).

<u>Method:</u>

The use of GEZ (or High Enrichment Condition) metrics have significant merit as they are features that should not be present in most estuaries so are relatively easy to delineate. Gross eutrophic zones have been delineated as areas which meet three eutrophic criteria: combined symptoms of high mud content (>25% mud), a shallow aRPD (<1 cm) and high macroalgal growth (>50% cover) (Stevens, 2018). Though slight variances on this combination have been used: high mud content, surface sediment anoxia (<1 cm), elevated organic matter and nutrient concentrations (Robertson and Stevens, 2013). Therefore, the progression of these measures as an attribute requires a consistent definition.

Gross eutrophic zones (GEZs) are areas identified as:

- mud content (> 25 %);
- a shallow aRPD/RPD (< 1 cm), and
- high macroalgal growth (> 50 % cover)*

*unless evidence exists that sediment toxicity has prevented persistence and/or establishment.

The mud content and aRPD values are based on the thresholds contained within the corresponding sections. The threshold for low oxygenation has been set at <1 cm opposed to surface to account for shallow surface oxygenation from overlying macroalgae, fresh deposition and recent wave action. Ideally a biomass threshold should be used to determine high macroalgal growth. International work has derived biomass threshold values between 700-1450g.m^{-2} ww (Green et al., 2014; McLaughlin et al., 2013; Sutula et al., 2014 and Scanlan et al., 2007), though these are based on preliminary work and for other system types such as Australian ICOLLs. A review of monitoring data from 25 typical NZ estuaries (shallow, short residence time estuaries) supports an opportunistic macroalgal biomass "exhaustion" threshold of approximately 1000 - 2000g.m^{-2} ww above which there was a major shift in the chemistry of the underlying sediment to surface anoxia (aRPD at the surface), elevated TOC (>1.5%) and a degraded macrofaunal community (Wriggle Coastal Management database 2009-2014; Zaiko et al. 2018). Further consideration should be given to interlacing EQR biomass measures into the last component of GEZ/GNA, high macroalgal growth.

Preliminary findings from 2018 field data across New River, Jacobs River and Fortrose estuaries support the use of >50% cover as measures are predominately above the aforementioned 1000g.m^2 wet weight (Figure 8).

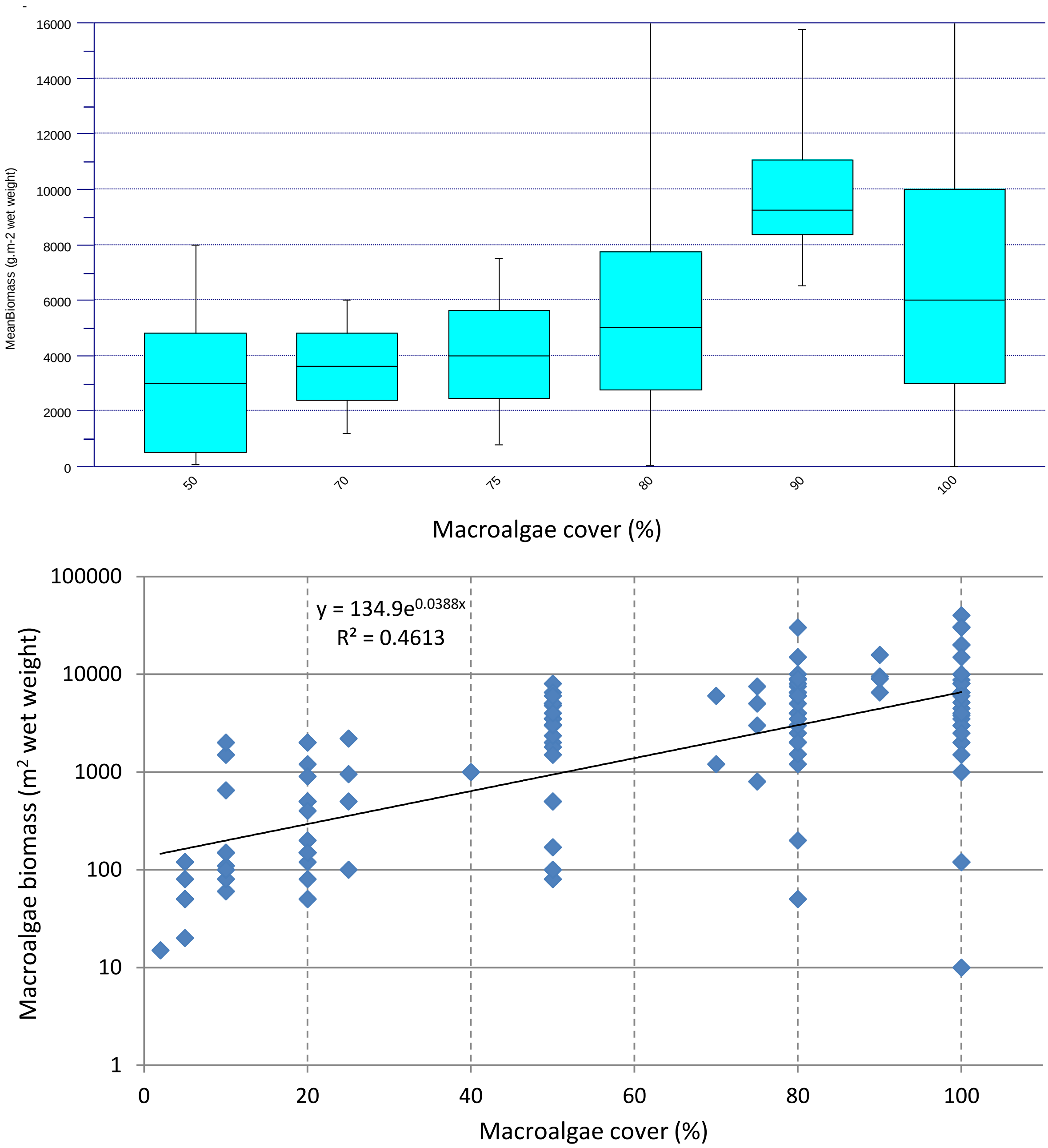

Figure 8. Macroalgae cover versus biomass field measures from 2018 field work for New River Estuary, Jacobs River and Fortrose estuaries combined.

There is the potential for further development to allow partial or full automated definition of these areas from individual GIS layers may be one way of standardising boundaries, but direct mapping is likely to be the most accurate. Additionally, integration with other algal metrics, grain size and sedimentation should be considered, see 'Future monitoring' in main body of report.

The recommended table is as follows:

Table 17. Attribute state option table for gross eutrophic zone

Value	Ecosystem health	
Freshwater body type	Tidal lagoon estuaries (SIDES) and tidal river estuaries (SSRTRE)	
Attribute group	Southland attribute	
Attribute name	Gross eutrophic zone	
Attribute unit	GEZ[1]	
Attribute band and description	**Numeric attribute state**	
	% cover across intertidal area	Area (ha)
A Within zones high stress on aquatic organisms and loss of habitat. Minimal impact on aquatic organisms outside these zones.	≤1	≤0.5
B Within zones high stress on aquatic organisms and loss of habitat. Minor impact on aquatic organisms outside these zones.	>1 and ≤5	>0.5 and ≤5
C Within zones high stress on aquatic organisms and loss of habitat. Moderate impact on aquatic organisms outside these zones. High risk of GEZ areas expanding and becoming self-reinforcing.	>5 and ≤10	>5 and ≤20
Proposed minimum acceptable state	**10**	**20**
D Within zones high stress on aquatic organisms and loss of habitat. Significant impact on aquatic organisms outside these zones. GEZ areas expanding and becoming self-reinforcing with severe loss of ecological integrity for whole estuary.	>10	>20
[1]GEZ is defined as areas consisting of >25% muddiness and aRPD depth of <1cm and 'high (>50%)' macroalgae cover. This definition may be reviewed if substantial areas of hydrogen sulphide toxicity prohibit macroalgae growth. The numeric bands proposed in this table are not based on literature thresholds. Nominal values have been proposed based on premise that any GEZ area in an estuary constitutes degradation.		

Faecal indicator organisms

Water that is contaminated by faecal matter can contain a range of disease-causing microorganisms such as viruses, bacteria and protozoa. Exposure to these microorganisms can result in a range of health issues with gastro-enteritis, hepatitis A, cryptosporidiosis, campylobacteriosis and respiratory problems being just a few.

E. coli

Recommendation: Proposed attribute for a numeric freshwater objective.

Note there are two tables. One for monthly sampling year round and one for popular bathing sites i.e. weekly in summer.

This is the preferred indicator organism in freshwater environments as it is generally straightforward and easy to measure. Survival time in freshwater is four to six weeks and therefore indicates recent contamination of faecal origin. Although it is not possible to distinguish *E. coli* from human, animal or avian origin this is of no consequence since all of the *E. coli* from these sources can act as carriers of human disease. Since multiplying in freshwater is rare, *E. coli* can give a quantitative assessment of contaminant levels. Though *E. coli* survivability in saline waters is limited the presence of *E. coli* may occur due to recent high flows and recent discharges. The NPSFM (2017) contains a four metric criteria for assessing *E. coli* in Lakes and Rivers. The same approach can be taken for estuaries and coast where appropriate data is available i.e. monthly data. The recommended table is as follows:

Table 18. Attribute state option table for *E. coli*.

Value	Human health for recreation			
Freshwater body type	Estuaries and open coast			
Attribute group	Southland attribute			
Attribute name	*Escherichiacoli* (*E. coli*)			
Attribute unit	*E. coli*/100 mL (number of *E. coli* per hundred millimetres)			
Attribute band and description	**Numeric attribute state**			
	% exceedances over 540 cfu/100 mL	% exceedances over 260 cfu/100 mL	Median concentration (cfu/100 mL)	95th percentile of *E. coli*/100 mL
A For at least half the time, the estimated risk is <1 in 1,000 (<0.1% risk). The predicted average infection risk is 1%*.	<5%	<20%	≤130	≤540
B For at least half the time, the estimated risk is <1 in 1,000 (<0.1% risk). The predicted average infection risk is 2%*.	5 to 10%	20 to 30%	≤130	≤1,000
C For at least half the time, the estimated risk is <1 in 1,000 (<0.1% risk). The predicted average infection risk is 3%*.	10 to 20%	20 to 34%	≤130	≤1,200
D 20 to 30% of the time the estimated risk is ≥50 in 1,000 (>5% risk). The predicted average infection risk is >3%*.	20 to 30%	>34%	>130	>1,200
E For more than 30% of the time the estimate risk is ≥50 in 1,000 (>5% risk). The predicted average infection risk is 7%*.	>30%	>50%	>260	>1,200
*The predicted average infection risk is the overall average infection to swimmers based on a random exposure on a random day, ignoring any possibility of not swimming during high flows or when a surveillance advisory is in place (assuming that the *E. coli* concentration follows a lognormal distribution). Actual risk will generally be less if a person does not swim during high flows. [1] Attribute state should be determined by using a minimum of 60 samples over a maximum of 5 years, collected on a regular basis regardless of weather and flow conditions. However, where a sample has been missed due to adverse weather or error, attribute state may be determined using samples over a longer timeframe. [2] Attribute state must be determined by satisfying all numeric attribute states.				

Additionally, the proposed NPSFM (2019) contains an additional table for *E. coli* at primary contact sites. This requires weekly data over the summer period; the proposed period for summer is November to March inclusive. The currently available data only covers December to March inclusive weekly sampling.

The recommended table is as follows:

Table 19. Attribute state option table for *E. coli* at popular bathing sites.

Value	Human health for recreation
Freshwater body type	Primary contact in estuaries and open coast
Attribute group	Southland attribute
Attribute name	*Escherichia coli* (*E. coli*) at popular bathing sites
Attribute unit	*E. coli*/100 mL (number of *E. coli* per hundred millimetres)
Attribute band and description	**Numeric Attribute State**
	95th percentile during the bathing season#
A <0.1% risk of Campylobacter infection. Risk of less than one case of Camplylobacter infection in every 1,000 exposures.	≤130
B 0.1 to 1% risk of Campylobacter infection. Risk of up to one case of Campylobacter infection in every 100 exposures.	>130 to ≤260
C 1 to 5% risk of Campylobacter infection. Risk of up to one to five cases of Campylobacter infection in every 100 exposures.	>260 to ≤540
National guideline for primary contact*	**540**
D >5% risk of Campylobacter infection. Risk of at least one case of Campylobacter infection in every 20 exposures.	>540
The narrative attribute state description assumes "% of time" equals "% of samples" *National bottom line proposed in the Essential Freshwater Package amendments to the NSPFM #Using weekly monitoring data	

Enterococci

Recommendation: Proposed attribute for a numeric freshwater objective.

Note there are two tables. One for monthly sampling year round and one for popular bathing sites i.e. weekly in summer.

Since *E. coli* and faecal coliforms generally do not survive well in saline waters it is preferable to use enterococci as the faecal contamination indicator for marine environments. Enterococci are more indicative of human sourced faecal contamination due to their abundance in human faeces and, like *E. coli*, they are easily cultured in the laboratory. It should be noted, however, that enterococci can also be present in the environment due to natural decomposition processes and therefore can come from sources other than faecal contamination. In order to apply equivalent tables to that of *E. coli* the same approach has been taken as the four metric table for *E. coli*, however two columns are only necessary as the same outcomes will occur using 4 metrics. The equivalent thresholds have been derived from available guidelines (MfE 2003). Again this is only applicable to sites with sufficiently appropriate data i.e. monthly data.

The recommended table is as follows:

Table 20. Attribute state option table for enterococci.

Value	Human health for recreation	
Freshwater body type	Estuaries and open coast	
Attribute group	Southland attribute	
Attribute name	enterococci	
Attribute unit	enterococci/100 mL (number of enterococci per hundred millilitres)	
Attribute band and description	**Numeric Attribute State**	
	95th percentile (cfu/100 mL)*	% exceedances over 280 enterococci/100 mL*
A Estimated GI risk is <1% and AFRI is <0.3% from a single exposure. The estimated GI risk is >10% and AFRI risk is >4% less than 5% of the time.	≤40	≤5
B Estimated GI risk is 1 - 5% and AFRI is 0.3 - 2% from a single exposure. The estimated GI risk is >10% and AFRI risk is >4% between 5 and 10% of the time.	>40 and ≤200	>5 and ≤10
C Estimated GI risk is 5 - 10% and AFRI is 2 - 4% from a single exposure. The estimated GI risk is >10% and AFRI risk is >4% between 10 and 20% of the time.	>200 and ≤500	>10 and ≤20
Proposed minimum acceptable state	**500**	**20**
D Estimated GI risk is >10% and AFRI is >4% from a single exposure. The estimated GI risk is >10% and AFRI risk is >4% more than 20% of the time.	>500	>20
*Using monthly monitoring data GI is gastrointestinal illness and AFRI is acute febrile respiratory illness		

An equivalent table to *E. coli* at primary contact sites (

Table 19) has been derived for enterococci at popular bathing sites (

Table 21) using the values contained in the guidelines (MfE 2003). Currently enterococci sampling is conducted weekly over the summer period, December to March inclusive.

The recommended table is as follows:

Table 21. Attribute state option table for enterococci at popular bathing sites.

Value	Human health for recreation
Freshwater body type	Primary contact in estuaries and open coast
Attribute group	Southland attribute
Attribute name	enterococci at popular bathing sites
Attribute unit	enterococci/100 mL (number of enterococci per hundred millilitres)
Attribute band and description	**Numeric Attribute State**
	95th percentile during the bathing season#
A Estimated GI risk is <1% and AFRI is <0.3% from a single exposure. The estimated GI risk is >10% and AFRI risk is >4% less than 5% of the time.	≤40
B Estimated GI risk is 1 - 5% and AFRI is 0.3 - 2% from a single exposure. The estimated GI risk is >10% and AFRI risk is >4% between 5 and 10% of the time.	>40 and ≤200
C Estimated GI risk is 5 - 10% and AFRI is 2 - 4% from a single exposure. The estimated GI risk is >10% and AFRI risk is >4% between 10 and 20% of the time.	>200 and ≤500
Proposed minimum acceptable state	**500**
D Estimated GI risk is >10% and AFRI is >4% from a single exposure. The estimated GI risk is >10% and AFRI risk is >4% more than 20% of the time.	>500
*Using weekly summer monitoring data GI is gastrointestinal illness and AFRI is acute febrile respiratory illness	

	As			Cd			Cr			Cu				
	2010	2016	2019	2010	2016	2019	2010	2016	2019	2010	2016	2019	2010	
Site A														
Site B														
e A			2018 2019 (6)			2018 2019 (6)	2009 (3)		2018 2019 (6)	2009 (3)		2018 2019 (6)	2009 (3)	
e B			2018 2019 (6)	2009 (3)		2018 2019 (6)			2018 2019 (6)			2018 2019 (6)		
ite A				2009 2010 (6)			2009 2010 (6)			2009 2010 (6)				
ite B				2009 2010 (6)	2015 (10)		2009 2010 (6)	2015 (10)		2009 2010 (6)	2015 (10)			20
ite C					2015 (10)			2015 (10)			2015 (10)			20
e A1			2019 (3)	2009 2010 (6)		2019 (3)	2009 2010 (6)		2019 (3)	2009 2010 (6)		2019 (3)		
e B			2019 (3)		2015 (10)	2019 (3)		2015 (10)	2019 (3)		2015 (10)	2019 (3)		20
e C					2015 (10)			2015 (10)			2015 (10)			20
Site A			2018 2019 (6)			2018 2019 (6)			2018 2019 (6)			2018 2019 (6)		
Site B			2018 2019 (6)		2015 (10)	2018 2019 (6)		2015 (10)	2018 2019 (6)		2015 (10)	2018 2019 (6)		20
Site C			2018 2019 (6)			2018 2019 (6)			2018 2019 (6)			2018 2019 (6)		
Site D			2018 2019 (6)		2015 (10)	2018 2019 (6)		2015 (10)	2018 2019 (6)		2015 (10)	2018 2019 (6)		20
Site E			2018 2019 (6)			2018 2019 (6)			2018 2019 (6)			2018 2019 (6)		
ite B			2018 2019 (6)	2010 (3)	2015 (10)	2018 2019 (6)	2010 (3)	2015 (10)	2018 2019 (6)	2010 (3)	2015 (10)	2018 2019 (6)	2010 (3)	20
ite C			2018 2019 (6)	2010 (3)		2018 2019 (6)	2010 (3)		2018 2019 (6)	2010 (3)		2018 2019 (6)	2010 (3)	
ite D			2018 2019 (6)	2010 (3)		2018 2019 (6)	2010 (3)		2018 2019 (6)	2010 (3)		2018 2019 (6)	2010 (3)	
ite E			2018 2019 (6)			2018 2019 (6)			2018 2019 (6)			2018 2019 (6)		
ite F			2018 2019 (6)		2015 (13)	2018 2019 (6)		2015 (13)	2018 2019 (6)		2015 (13)	2018 2019 (6)		20
te A			2019 (3)	2008 (3)		2019 (3)	2008 (3)		2019 (3)	2008 (3)		2019 (3)	2008 (3)	
te B			2019 (3)	2008 (3)	2015 (10)	2019 (3)	2008 (3)	2015 (10)	2019 (3)	2008 (3)	2015 (10)	2019 (3)	2008 (3)	20
te C					2015 (10)			2015 (10)			2015 (10)			20
Site D			2018(1)			2018(1)			2018(1)			2018(1)		
Site E			2018(1)			2018(1)			2018(1)			2018(1)		
Site G			2018(1)			2018(1)			2018(1)			2018(1)		

Appendix C - Estuarine and Coastal analytical details

The state of ecosystem health, habitat and water quality attributes at a site, estuary, estuary class or FMU, at a given baseline year (2010, 2016 and 2019) was assessed using data from the preceding three years.

Details of specific statistical requirements are shown in Table 22 and data periods used to represent the different attributes in

Table 23, **Error! Reference source not found.**, **Error! Reference source not found.** and **Error! Reference source not found.**.

Most of the data is collected on an annual basis so is not monthly or weekly except that of phytoplankton for New River estuary and microbiological data. Minimum statistics have been specified but not stringently adhered to otherwise much of the data would be excluded. For some periods there is no data available so the closest preceding time period was used.

It is important to note the following:

1. Not all attributes have been assessed for each site, estuary, estuary class or FMU and for every baseline, as the number and frequency of monitoring needed to cover estuaries in Southland is unfeasible. Focus has therefore been on high risk (highly modified/developed land use in catchment and high sensitivity) estuaries.
2. There was a hiatus in most of the estuarine monitoring from 2013 to 2016.
3. Waiau estuary (not lagoon which is classed within Lakes and lagoons) has no available data.
4. Some attributes are at a site scale and some at an estuarine scale.
5. EQR is a measure of macroalgal response and therefore can theoretically detect a response in the estuarine system to increased pressures. GEZ/GNA measures detect the condition of a system once it has exceeded its assimilative capacity, taking into account sediment oxygen state and algal cover. Measures for EQR and GEZ/GNA should therefore considered as fundamentally different but also tools to use in a gradient of deteriorating state.
6. For microbial data values which were below the limit of detection a general rule was applied; the below detect was replaced with a numerical value equivalent to a half fraction of the detection limit. Studies have shown that the application of this rule is not suitable particularly for regression and correlation analysis (Helsel, 2006), however the purpose of this report was not to assess trends but to report on state.
7. Analysis for phytoplankton has yet to be done for the periods of 2010 and 2016.
8. For microbial coastal/marine sites there is a decline in WQ using cfu measures up to 2007. Following this the method was changed to MPN; however, no cross over time for the methods exists. The measure of cfu was re-established in 2014. Therefore, only one period may be calculated using the data.
9. For a small period both *E. coli* cfu and enterococci cfu were measured at popular bathing sites (summer weekly). For sites that displayed little to no issues sampling were ceased or not started for *E. coli*. These are Colac Bay at Colac Bay Road opp marae, Monkey Island at Frentz Road, Ōreti Beach at Dunns Road, Riverton Rocks at Mitchells Bay North, Halfmoon Bay at Bathing Beach, Halfmoon Bay at Elgin Terrace, Porpoise Bay at Camping Ground, Awarua Bay at Tiwai Pumphouse. Those sites which showed some pollution were retained and continued to be sampled for both enterococci and *E. coli*: Bluff Harbour at Morrison Beach, Jacobs River Estuary d/s Railway Br East, New River Estuary at Water Ski Club and New River Estuary at Omaui. Note that *E. coli* will therefore have less data available for the ongoing sites due to the more recent introduction.
10. The main analyte for use against guidelines is Faecal coliforms but *E. coli* can be measured at the same time for no additional cost. For a small period both *E. coli* cfu and enterococci cfu have been measured at shellfish gathering waters (monthly data). enterococci has not been measured or ceased (due to low detected levels) at select sites: Riverton Rocks at Mitchells Bay. Some sites continue to be measured for both *E. coli* and enterococci: Bluff Harbour at Ocean Beach, Monkey Island at Frentz Road, Colac Bay at Bungalow Hill Road, Jacobs River Estuary d/s Fish co-op, New River Estuary at Whalers Bay, New River Estuary at Mokomoko Inlet, Toetoes Harbour at Fortrose.

Table 22. Statistical details for estuarine and coastal attributes.

Scale	Attribute	Statistic	Minimum Requirements	Additional Conditions

Site	**Mud content (% mud)**	Annual Median	Up to 3 years of annual sampling n = 3 composite samples /yr	Sampling is conducted according to the estuary protocol by taking 3 composite samples for a site.
Site	**aRPD (cm below surface)**	Annual median	Annual data (n=10)	Sampling is conducted according to the estuary protocol by taking 10 measures in conjunction with macrofaunal samples.
Site	**Toxicants in sediment (As, Cd, Cr, Cu, Pb, Ni) (mg/Kg)**	Annual Median	Up to 3 years of annual sampling n = 3 composite samples /yr	Sampling is conducted according to the estuary protocol by taking 3 composite samples for a site.
Site	**Sediment rate (mm/yr)**	>2mm or <2mm if pass trend significance test (90%).	Filter sites according to if muddy or not (>25% mud content). Use all available annual data, 4 plates per site. Slope calculated using all individual data points to determine slope. Rate may be calculated from post significant erosion events and periods.	Slope characteristics used to run two criteria, slope significance test and test if <2mm/yr rate. Model coefficient was compare against threshold (2mm) using t-test; using R-Script. Not significant (significance test above) = IN (Indeterminate). If Significant (significance test above) and be <2mm to PASS; and be >2mm to FAIL.
Estuary	**Mud extent (m^2 of intertidal area)**	Area >25% mud content has increased/decreased	Annual assessment for the estuary.	Change is calculated from most recent and current assessment.
Site	**Phytoplankton in water (mg chlorophyll-*a* /m^3)**	90^{th} percentile	Twice a Month data, minimum 5 years. Sites are identified as >30ppt or <30ppt salinity using median conductivity for the data used.	Chl-*a* measure done at ICC lab: Acetone extraction. Fluorometer. In line with APHA 10200. ICC lab is a non-accredited lab. 90^{th} percentile was calculated using excel. Data has not been filtered according to low and high flow, one of each has been recorded per month.
Scale	**Attribute**	**Statistic**	**Minimum Requirements**	**Additional Conditions**
Estuary	**EQR**	No units. The score is based on biomass and cover of macroalgae.	Annual assessment for the estuary.	Refer to the New Zealand Estuary Trophic Index (Wriggle Coastal Management Ltd and NIWA, 2015) for guidance on how to calculate EQR.
Estuary	**GEZ**	% cover and Area (Ha) of the estuary	Annual assessment for the estuary.	GEZ is defined as areas consisting of >25% muddiness and aRPD depth of <1 cm and 'high' macroalgae cover. This definition may be reviewed if substantial areas of hydrogen sulphide toxicity prohibit macroalgae growth.
Site	**E. coli (cfu/100ml)**	Median, 95^{th} percentile	5 years of monthly sampling, ideally n = 60	NPS-FM recommends a minimum of 60 samples over 5

		% exceedances over 540 E. coli/100mL % exceedances over 260 E. coli/ 100mL	Some sites have less data available.	years, where a sample is missed the state may be determined over a longer timeframe. Calculations were done via R-Script. Using the hazen method to calculate the 95^{th} percentile.
Site	**E. coli at popular bathing sites (cfu/100ml)**	95^{th} percentile	5 years (seasons) of weekly data over the summer, ideally n = 80 to 85. Some sites have less data available.	The Proposed NPS-FM (2019) has no recommendations for minimum data used. However, the Microbial guidelines recommend using 5 years' worth of data. Calculations were done via R-Script. Using the hazen method to calculate the 95^{th} percentile.
Site	**enterococci (cfu/100ml)**	95th percentile % exceedances over 280 enterococci/100 mL	5 years of monthly sampling, ideally n = 60 Some sites have less data available.	NPS-FM recommends a minimum of 60 samples over 5 years for E. coli. The same principles have been applied to enterococci. Calculations were done via R-Script. Using the hazen method to calculate the 95^{th} percentile.
Site	**enterococci at popular bathing sites (cfu/100ml)**	95^{th} percentile	5 years (seasons) of weekly data over the summer, ideally n = 80 to 85. Some sites have less data available.	The Microbial guidelines recommend using 5 years' worth of data. Calculations were done via R-Script. Using the hazen method to calculate the 95^{th} percentile.

Table 23. Data used for attributes at site scale.

Site	As			Cd			Cr			Cu			Pb		
	2010	2016	2019	2010	2016	2019	2010	2016	2019	2010	2016	2019	2010	2016	2019
Awarua Bay Estuary Site A															
Awarua Bay Estuary Site B															
Bluff Harbour Site A															
Bluff Harbour Site B															
Fortrose Estuary Site A			2018 2019 (6)			2018 2019 (6)	2009 (3)		2018 2019 (6)	2009 (3)		2018 2019 (6)	2009 (3)		2018 2019 (6)
Fortrose Estuary Site B			2018 2019 (6)	2009 (3)		2018 2019 (6)			2018 2019 (6)			2018 2019 (6)			2018 2019 (6)
Freshwater Estuary Site A				2009 2010 (6)			2009 2010 (6)			2009 2010 (6)					
Freshwater Estuary Site B				2009 2010 (6)	2015 (10)		2009 2010 (6)	2015 (10)		2009 2010 (6)	2015 (10)			2015 (10)	
Freshwater Estuary Site C					2015 (10)			2015 (10)			2015 (10)			2015 (10)	
Haldane Estuary Site A1			2019 (3)	2009 2010 (6)		2019 (3)	2009 2010 (6)		2019 (3)	2009 2010 (6)		2019 (3)			2019 (3)
Haldane Estuary Site B			2019 (3)		2015 (10)	2019 (3)		2015 (10)	2019 (3)		2015 (10)	2019 (3)		2015 (10)	2019 (3)
Haldane Estuary Site C					2015 (10)			2015 (10)			2015 (10)			2015 (10)	
Jacobs River Estuary Site A			2018 2019 (6)			2018 2019 (6)			2018 2019 (6)			2018 2019 (6)			2018 2019 (6)
Jacobs River Estuary Site B			2018 2019 (6)		2015 (10)	2018 2019 (6)		2015 (10)	2018 2019 (6)		2015 (10)	2018 2019 (6)		2015 (10)	2018 2019 (6)
Jacobs River Estuary Site C			2018 2019 (6)			2018 2019 (6)			2018 2019 (6)			2018 2019 (6)			2018 2019 (6)
Jacobs River Estuary Site D			2018 2019 (6)		2015 (10)	2018 2019 (6)		2015 (10)	2018 2019 (6)		2015 (10)	2018 2019 (6)		2015 (10)	2018 2019 (6)
Jacobs River Estuary Site E			2018 2019 (6)			2018 2019 (6)			2018 2019 (6)			2018 2019 (6)			2018 2019 (6)
New River Estuary Site B			2018 2019 (6)	2010 (3)	2015 (10)	2018 2019 (6)	2010 (3)	2015 (10)	2018 2019 (6)	2010 (3)	2015 (10)	2018 2019 (6)	2010 (3)	2015 (10)	2018 2019 (6)
New River Estuary Site C			2018 2019 (6)	2010 (3)		2018 2019 (6)	2010 (3)		2018 2019 (6)	2010 (3)		2018 2019 (6)	2010 (3)		2018 2019 (6)
New River Estuary Site D			2018 2019 (6)	2010 (3)		2018 2019 (6)	2010 (3)		2018 2019 (6)	2010 (3)		2018 2019 (6)	2010 (3)		2018 2019 (6)
New River Estuary Site E			2018 2019 (6)			2018 2019 (6)			2018 2019 (6)			2018 2019 (6)			2018 2019 (6)
New River Estuary Site F			2018 2019 (6)		2015 (13)	2018 2019 (6)		2015 (13)	2018 2019 (6)		2015 (13)	2018 2019 (6)		2015 (13)	2018 2019 (6)
Waikawa Estuary Site A			2019 (3)	2008 (3)		2019 (3)	2008 (3)		2019 (3)	2008 (3)		2019 (3)	2008 (3)		2019 (3)
Waikawa Estuary Site B			2019 (3)	2008 (3)	2015 (10)	2019 (3)	2008 (3)	2015 (10)	2019 (3)	2008 (3)	2015 (10)	2019 (3)	2008 (3)	2015 (10)	2019 (3)
Waikawa Estuary Site C					2015 (10)			2015 (10)			2015 (10)			2015 (10)	
Waimatuku Estuary Site D			2018(1)			2018(1)			2018(1)			2018(1)			2018(1)
Waimatuku Estuary Site E			2018(1)			2018(1)			2018(1)			2018(1)			2018(1)
Waimatuku Estuary Site G			2018(1)			2018(1)			2018(1)			2018(1)			2018(1)

Site	Hg			Ni			Zn			% mud			aRPD		
	2010	2016	2019	2010	2016	2019	2010	2016	2019	2010	2016	2019	2010	2016	2019
Awarua Bay Estuary Site A															
Awarua Bay Estuary Site B															
Bluff Harbour Site A															
Bluff Harbour Site B															
Fortrose Estuary Site A			2018 2019 (6)	2009 (3)		2018 2019 (6)	2009 (3)		2018 2019 (6)			2018 2019 (6)			2019(10)
Fortrose Estuary Site B			2018 2019 (6)			2018 2019 (6)			2018 2019 (6)	2009(3)		2018 2019 (6)	2009(10)		2019(10)
Freshwater Estuary Site A				2009 2010 (6)			2009 2010 (6)			2009 2010 (6)			2010(10)		
Freshwater Estuary Site B		2015 (10)		2009 2010 (6)	2015 (10)		2009 2010 (6)	2015 (10)		2009 2010 (6)			2010(10)	2015(10)	
Freshwater Estuary Site C		2015 (10)			2015 (10)			2015 (10)						2015(10)	
Haldane Estuary Site A1			2019 (3)	2009 2010 (6)		2019 (3)	2009 2010 (6)		2019 (3)	2009 2010 (6)		2019 (3)			2019(10)
Haldane Estuary Site B		2015 (10)	2019 (3)		2015 (10)	2019 (3)		2015 (10)	2019 (3)			2019 (3)		2015(10)	2019(10)
Haldane Estuary Site C		2015 (10)			2015 (10)			2015 (10)						2015(10)	
Jacobs River Estuary Site A			2018 2019 (6)			2018 2019 (6)			2018 2019 (6)			2018 2019 (6)			2019(10)
Jacobs River Estuary Site B		2015 (10)	2018 2019 (6)		2015 (10)	2018 2019 (6)		2015 (10)	2018 2019 (6)			2018 2019 (6)		2015(10)	2019(10)
Jacobs River Estuary Site C			2018 2019 (6)			2018 2019 (6)			2018 2019 (6)			2018 2019 (6)			2019(10)
Jacobs River Estuary Site D		2015 (10)	2018 2019 (6)		2015 (10)	2018 2019 (6)		2015 (10)	2018 2019 (6)			2018 2019 (6)		2015(10)	2019(10)
Jacobs River Estuary Site E			2018 2019 (6)			2018 2019 (6)			2018 2019 (6)			2018 2019 (6)			2019(10)
New River Estuary Site B		2015 (10)	2018 2019 (6)	2010 (3)	2015 (10)	2018 2019 (6)	2010 (3)	2015 (10)	2018 2019 (6)	2010 (3)		2018 2019 (6)	2010(10)	2015(10)	2019(10)
New River Estuary Site C			2018 2019 (6)	2010 (3)		2018 2019 (6)	2010 (3)		2018 2019 (6)	2010 (3)		2018 2019 (6)	2010(10)		2019(10)
New River Estuary Site D			2018 2019 (6)	2010 (3)		2018 2019 (6)	2010 (3)		2018 2019 (6)	2010 (3)		2018 2019 (6)	2010(10)		2019(10)
New River Estuary Site E			2018 2019 (6)			2018 2019 (6)			2018 2019 (6)			2018 2019 (6)			2019(10)
New River Estuary Site F		2015 (13)	2018 2019 (6)		2015 (13)	2018 2019 (6)		2015 (13)	2018 2019 (6)		2016 (3)	2018 2019 (6)		2015(10)	2019(10)
Waikawa Estuary Site A			2019 (3)	2008 (3)		2019 (3)	2008 (3)		2019 (3)			2019 (3)	2008(10)		2019(10)
Waikawa Estuary Site B		2015 (10)	2019 (3)	2008 (3)	2015 (10)	2019 (3)	2008 (3)	2015 (10)	2019 (3)			2019 (3)	2008(10)	2015(10)	2019(10)
Waikawa Estuary Site C		2015 (10)			2015 (10)			2015 (10)						2015(10)	2019(10)
Waimatuku Estuary Site D			2018(1)			2018(1)			2018(1)	2018(1)		2018(1)			2008(1)
Waimatuku Estuary Site E			2018(1)			2018(1)			2018(1)	2018(1)		2018(1)			2008(1)
Waimatuku Estuary Site G			2018(1)			2018(1)			2018(1)	2018(1)		2018(1)			2008(1)

Site	Phytoplankton (Chlorophyll-a)			*E.coli*			*E.coli* at popular bathing sites			enterococci			enterococci at popular bathing sites		
	2010	2016	2019	2010	2016	2019	2010	2016	2019	2010	2016	2019	2010	2016	2019
New River Estuary Omaui Beach	To be done	To be done	Jun 2016-Jul 2019 (74)												
New River Estuary Awarua Farm	To be done	To be done	Jun 2016-Jul 2019 (68)												
New River Estuary Lagoon tip outlet	To be done	To be done	Jun 2016-Jul 2019 (73)												
New River Estuary Stead Street	To be done	To be done	Jun 2016-Jul 2019 (74)												
New River Estuary Dunns Road	To be done	To be done	Jun 2016-Jul 2019 (74)												
Ōreti Beach	To be done	To be done	Jun 2016-Jul 2019 (73)												
New River Estuary Ski club	To be done	To be done	Jun 2016-Jul 2019 (74)												
New River Estuary Mcoys Beach	To be done	To be done	Jun 2016-Jul 2019 (70)												
Bluff Harbour at Ocean Beach						2016 – 2019 (42)						2014 – 2019 (63)			
Colac Bay at Bungalow Hill Road						2015 – 2019 (56)						2014 – 2019 (63)			
Jacobs River Estuary d/s Fish Co-op						2014 – 2019 (59)						2014 – 2019 (63)			
Monkey Island at Frentz Road (south)						2016 – 2019 (41)						2014 – 2019 (63)			
New River Estuary at Mokomoko Inlet						2016 – 2019 (42)						2014 – 2019 (63)			
New River Estuary at Whalers Bay						2016 – 2019 (42)						2014 – 2019 (63)			
Riverton Rocks at Mitchells Bay						2016 – 2019 (34)						2014 – 2017 (43)			
Toetoes Harbour at Fortrose						2016 – 2019 (42)						2014 – 2019 (63)			
Awarua Bay at Tiwai pumphouse															2013 – 2019 (54)
Bluff Harbour at Morrison Beach									2015 – 2019 (55)						2013 – 2019 (102)
Colac Bay at Colac Bay Road opp marae									2015 – 2017 (21)						2013 – 2019 (102)
Halfmoon Bay at bathing beach															2013 – 2019 (96)
Halfmoon Bay at Elgin Terrace															2013 – 2019 (97)
Jacobs River Estuary d/s Railway Br East									2015 – 2019 (55)						2013 – 2019 (101)
Kawakaputa Bay at Wakapatu Road															2013 – 2019 (101)
Monkey Island at Frentz Road									2015 – 2017 (20)						2013 – 2019 (101)
New River Estuary at Omaui									2015 – 2019 (55)						2013 – 2019 (102)
New River Estuary at Water Ski Club									2015 – 2019 (55)						2013 – 2019 (101)
Ōreti Beach at Dunns Road									2015 – 2017 (21)						2013 – 2019 (93)
Porpoise Bay at camping ground															2013 – 2019 (54)
Riverton Rocks at Mitchells Bay North									2015 – 2017 (21)						2013 – 2019 (102)

Estuary	Macroalgae (EQR)			GEZ			Muddiness of intertidal area		
	2010	2016	2019	2010	2016	2019	2010	2016	2019
Fortrose Estuary	2013	2016	2018	2013	2016	2018	2003 compare 2013	2013 compare 2016	2016 compare 2018
Freshwater Estuary									
Haldane Estuary	2004	2016		2004	2016			2004 compare 2016	
Jacobs River Estuary		2016	2018	2008	2016	2018	2003 compare 2008	2013 compare 2016	2016 compare 2018
New River Estuary	2007	2016	2018	2007	2016	2018	2007 compare 2012	2013 compare 2016	2016 compare 2018
Waikawa Estuary	2008	2016		2009	2016		2004 compare 2009	2009 compare 2016	