Advice on Indicators, Thresholds and Bands for Estuaries in Aotearoa New Zealand

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Salt Ecology
Report 141

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Advice on Indicators, Thresholds and Bands for Estuaries in Aotearoa New Zealand

Introductory Report

prepared by

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for

Ministry for the Environment

June 2024

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For the environment
Mō te taiao

GLOSSARY

AA	Affected Area (OMBT metric)
ADCP	Acoustic Doppler current profiler
AIH	Available Intertidal Habitat (OMBT metric)
AMBI	AZTI Marine Biotic Index (macroinvertebrate index)
ANZECC	Australian and New Zealand Environment and Conservation Council
ANZG	Australian and New Zealand Guidelines for Fresh and Marine Water Quality (2018)
aRPD	Apparent Redox Potential Discontinuity (assessed visually)
As	Arsenic
ASH	Available Salt marsh Habitat
AVS	Acid-volatile sulfide
BHM	Benthic Health Model (macroinvertebrate index)
BHQ	Benthic Habitat Quality
Cd	Cadmium
CLUES	Catchment Land Use for Environmental Sustainability (NIWA model)
Cr	Chromium
CTD	Conductivity, Temperature, Depth
Cu	Copper
DGV	Default Guideline Value (ANZG)
DIN	Dissolved Inorganic Nitrogen
DO	Dissolved Oxygen
DOP	Degree of Pyritization
DSDE	Deeper Subtidal Dominated, longer residence time Estuary
EG	Eco-Group (used in AMBI)
EQR	Ecological Quality Ratio
ETI	Estuary Trophic Index
EVA	Ecological Vulnerability Assessment
GIS	Geographic Information System
GPS	Global Positioning System
GV-high	Guideline Value-High (ANZG)
HAB	Harmful Algal Blooms
Hg	Mercury
ICOLL	Intermittently closed/open lakes and lagoons estuary
LCDB	Land Cover Data Base
LiDAR	Light Detection And Ranging; remote sensing method for measuring bed height
LoD	Limits of Detection
LOI	Loss On Ignition
MDL	Method Detection Limit
MfE	Ministry for the Environment
MHWN	Mean High Water Neap (tide height)
MHWS	Mean High Water Spring (tide height)
MPI	Ministry for Primary Industries
NEMP	National Estuary Monitoring Protocol
NEMS	National Environmental Monitoring Standards
Ni	Nickel
NIWA	National Institute of Water and Atmospheric Research

NH_3	Ammoniacal nitrogen
$N0_2$-	Nitrite nitrogen
$N0_3$-	Nitrate nitrogen
NNE	Nutrient Numeric Endpoint
NOF	National Objectives Framework
NPS-FM	National Policy Statement for Freshwater Management
OMBT	Opportunistic Macroalgae Blooming Tool
Pb	Lead
pRPD	Probe Redox Potential Discontinuity (assessed with instrumentation)
QA/QC	Quality Assurance/Quality Control
RPD	Redox Potential Discontinuity
SIDE	Shallow Intertidal Dominated Estuary
SLR	Sea Level Rise
SOE	State of Environment (Monitoring)
SPI	Sediment Profile Imaging
SQGVS	Sediment Quality Guideline Values
SRP	Soluble Reactive Phosphorus
SSRTRE	Shallow, Short Residence-time Tidal River Estuary
SVOC	Semi Volatile Organic Compounds
TBI	Traits Based Index (macroinvertebrate index)
TKN	Total Kjeldahl Nitrogen
TN	Total Nitrogen
TOC	Total Organic Carbon
TON	Total Oxidized Nitrogen
TP	Total Phosphorus
TS	Total Sulphur
TSS	Total Suspended Solids
WFD	Water Framework Directive
Zn	Zinc

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We are also grateful to Cawthron Institute staff Susie Wood and Laura Biessy for reviewing, the cyanobacteria indicator.

As part of a concurrent MfE project undertaking a 'stocktake' of 55 potential ecological indicators, we acknowledge the provision of draft outputs on salt marsh and seagrass from Anna Berthelsen (Cawthron Institute), and cyanobacteria in coastal waters from Laura Biessy and Susie Wood (Cawthron Institute), and the use of information on other indicators prepared by NIWA staff.

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SUMMARY

In March 2024, MfE contracted Salt Ecology and NIWA to assess a suite of 19 commonly used estuarine ecological indicators as outlined in the table to the side. The project brief was to:

- Document the rationale for inclusion of proposed indicators, and any caveats associated with their use;
- Review existing thresholds and associated bands for each indicator, and provide advice on ecologically relevant thresholds and associated bands for non-compulsory use in estuarine monitoring programmes in Aotearoa New Zealand;
- Indicate the degree of certainty and scientific robustness associated with any proposed thresholds (numeric or narrative) and, where appropriate, specify additional work needed to improve them.

Habitat indicators (estuary-wide)
Macroalgae (opportunistic species)
Mangrove forest extent and quality
'Mud elevated' (>25% mud) sediment extent
Salt marsh extent and quality
Seagrass extent and quality
Shellfish bed extent and quality
Sediment indicators (site-specific)
Sedimentation rate
Macrofauna (community composition)
Microalgae (chlorophyll-*a* and phaeophytin)
Mud content
Nutrients (sediment N and P)
Organic matter
Depth to Redox Potential Discontinuity
Sulphur and sulphides
Trace metals
Water column indicators (site-specific)
Cyanobacteria
Dissolved oxygen
Nutrients (water column N and P)
Phytoplankton (chlorophyll-*a* concentration)

The primary aim for developing non-compulsory thresholds and associated bands is to assist decision-makers and communities to better interpret monitoring results, understand stressor-response relationships, articulate the quality they wish to protect/achieve, and understand the extent to which current conditions are supporting objectives. The thresholds presented herein, and summarised in tables on the following page, are not intended for use as individual regulatory targets or to assess compliance, although they may be developed for these purposes over time.

Detailed advice for each indicator has been presented as a stand-alone appendix authored by the respective subject expert(s). This is to facilitate future indicator-specific updates, allow the easy addition of new indicators, and enable a web-based reporting approach should that be considered in future.

There has been little scope to collate or analyse data to develop new thresholds. Preliminary data analyses to support literature findings have been undertaken where feasible, but otherwise the information supplied is drawn from existing thresholds and knowledge from the project team's own studies, other New Zealand studies, and limited review of international literature. Further, the current report does not attempt to prioritise or recommend specific indicator use within estuary monitoring programmes.

This project is a significant step toward a national approach for clear and consistent interpretation, communication and reporting on the ecological health of estuaries, but it is not the endpoint. It is emphasised that the purpose of developing and refining thresholds is to assist councils to interpret SOE monitoring data and to guide timely and effective management actions. Hence, the level of scientific certainty is less critical than it would be for the development of regulatory or compliance thresholds.

RECOMMENDATIONS

- Prioritise (by national consensus) further development of indicators with direct links to management and which relate to the most ecologically damaging estuary stressors that Councils can manage or mitigate, i.e., fine sediment, nutrients, and habitat loss/displacement.
- Where appropriate, adopt a 5-band threshold structure (i.e., Very good, Good, Fair, Poor, Very poor).
- Collate existing national data and undertake analysis of relationships between indicators, stressors, and ecological responses as a high priority for supporting and refining proposed thresholds.
- Give initial priority to refining: (i) site-specific thresholds for sediment TOC, TN, mud and metals, as well as macrofauna BHM and AMBI, and (ii) estuary-wide thresholds for macroalgae, seagrass and salt marsh indicators.
- MfE provide specific guidance to Councils on how these thresholds should be applied or adopted.

Summary of **habitat (estuary-wide) indicator thresholds** proposed by subject matter experts relative to expected ecological quality status. See Technical Appendices for caveats and guidance on threshold use.

Indicator	Metric	Ecological Quality Status				
		Very Good	Good	Fair	Poor	Very Poor
Macroalgae	OMBT-EQR^	≥0.8 to 1.0	≥0.6 to <0.8	≥0.4 to <0.6	≥0.2 to <0.4	0.0 to <0.2
Mud-elevated	% of intertidal area with mud-elevated sediment*	<1%	1 to <5%	≥5 to <15%	>15%	>25%
	% increase of intertidal mud-elevated sediment from baseline*	≤0	>0 to <5%	≥5% to <10%	≥10% to <20%	≥20%
Salt marsh	% of available salt marsh habitat (ASH)#^	≥50%	≥25 to <50%	≥10 to <25%	≥5 to <10%	0 to <5%
	% loss from first accurately measured baseline*	≤0	>0 to <5%	≥5% to <10%	≥10% to <20%	≥20%
	% loss from estimated historical extent#	<0 to <20%	≥20% to <40%	≥40% to <60%	≥60 % to <80%	≥80% loss
Seagrass	% loss of dominant (>50% cover) seagrass from first accurate baseline*	≤0	>0 to <5%	≥5% to <10%	≥10% to <20%	≥20%
	% reduction in average %cover (density) (annual mean)#	>0 to ≤10%	>10% to ≤30%	>30% to ≤50%	>50 % to ≤70%	>70% loss
	% reduction in average %cover (density) (5-yr rolling mean)#	>0 to ≤5%	>5% to ≤15%	>15% to ≤25%	>25 % to ≤35%	>35% loss
Shellfish	% loss from estimated historical extent*	<0 to <10%	≥10% to <20%	≥20% to <50%	≥50 % to <75%	≥75% loss

*Thresholds based on expert judgement. #Based on WFD. ^Note scaling is high to low.

Summary of **site-specific sediment indicator thresholds** proposed by subject matter experts relative to expected ecological quality status. See Technical Appendices for caveats and guidance on threshold use.

Indicator	Metric	Ecological Quality Status				
		Very Good	Good	Fair	Poor	Very Poor
Accretion	SAR (mm/yr) if assumed natural SAR ≤1mm/yr	---------------------	0 to 1	≥1 to 3	≥3 to 10	≥10
	SAR (mm/yr) above natural SAR	---------------------	0	>0 to 2	≥2 to 10	≥10
Macroinvertebrates	AMBI score*	0 to 1.2	>1.2 to 3.3	>3.3 to 4.3	>4.3 to 5.0	>5.0 to 7
	TBI Score#^	---------------------	>0.4 to 1	≥0.3 to 0.4	0 to 0.2	---------------------
Mud content	% sediment mud content	<5	≥5 to <10	≥10 to <25	≥25 to <50	≥50
Nutrients	Sediment TN (mg/kg)	<250	≥250 to <800	≥800 to <1200	≥1200 to <2000	≥2000
Organic matter	TOC (%)	<0.5%	0.5 to 1.2%	>1.2% to 2%	>2%	---------------------
RPD	RPD depth (mm)^	>40	40 to >25	25 to 10	<10	---------------------
Total Sulphur	TOC:TS^	---------------------	>5	≤5 to 1.5	<1.5	---------------------
Trace Metals	Threshold 'rules' relative to ANZG (2018) SQGVs	<25% DGV	25 to <50% DGV	50% DGV to <DGV	DGV to <GV-High	≥GV-high
	As (mg/kg)	<5	5 to <10	10 to <20	20 to <70	≥70
	Cd (mg/kg)	<0.38	0.38 to <0.75	0.75 to <1.5	1.5 to <10	≥10
	Cr (mg/kg)	<20	20 to <40	40 to <80	80 to <370	≥370
	Cu (mg/kg)	<16	16 to <32.5	32.5 to <65	65 to <270	≥270
	Hg (mg/kg)	0.038	0.038 to <0.075	0.075 to <0.15	0.15 to <1	≥1
	Ni (mg/kg)	<5.25	5.2 to <10.5	10.5 to <21	21 to <52	≥52
	Pb (mg/kg)	<12.5	12.5 to <25	25 to <50	50 to <220	≥220
	Zn (mg/kg)	<50	50 to <100	100 to <200	200 to <410	≥410

* AMBI 7-point scale mapped to 5-band scale. #TBI score mapped to equivalent Ecological Quality Status band. ^Note scaling is high to low.

Indicator	Metric	Level of impact relative to other estuarine sites in New Zealand				
		Very low	Low	Moderate	High	Very High
Macroinvertebrates	BHM score	1 to <2	2 to <3	3 to <4	4 to <5	≥5

Summary of **site-specific water column indicator thresholds** proposed by subject matter experts relative to expected ecological quality status. See Technical Appendices for caveats and guidance on threshold use.

Indicator	Metric		Ecological Quality Status				
			Very Good	Good	Fair	Poor	Very Poor
Dissolved Oxygen	7-day mean minimum (mg/L DO)^		≥7.0	7.0 to ≥6.0	6.0 to ≥5.0	<5.0	---------------------
Potential TN (mg/m^3)	Corresponding to macroalgal OMBT bands		<175	175 to ≤335	335 to ≤ 495	>495	---------------------
Potential TN (mg/m^3)	Corresponding to phytoplankton bands	Euhaline (>30ppt)	<30	≥30 to 75	≥75 to 110	>110	---------------------
		Meso/polyhaline (≥5-30ppt)	<45	≥45 to 90	≥90 to 145	>145	---------------------
		Oligohaline (<5ppt)	<90	≥90 to 225	≥225 to 530	>530	---------------------
Potential TP (mg/m^3)	Corresponding to phytoplankton bands	Euhaline (>30ppt)	<4	≥4 to 10	≥10 to 15	>15	---------------------
		Meso/polyhaline (≥5-30ppt)	<6	≥6 to 12	≥12 to 20	>20	---------------------
		Oligohaline (<5ppt)	<12	≥12 to 30	≥30 to 75	>75	---------------------
Phytoplankton Biomass: Chlorophyll-a (mg/m^3)	90th percentile	Euhaline (>30ppt)	≤ 3	>3 to ≤8	>8 to ≤12	>12 to ≤16	>16
	90th percentile	Meso/polyhaline (≥5-30ppt)	≤ 5	>5 to ≤12	>12 to ≤16	>16 to ≤32	>32
	Annual median	Oligohaline (<5ppt)	≤2	>2 to ≤5	>5 to ≤12	>12 to ≤30	>30
	Annual maximum	Oligohaline (<5ppt)	≤10	>10 to ≤25	>25 to ≤60	>60 to ≤150	>150

^Note scaling is high to low.

1. INTRODUCTION

1.1 BACKGROUND

Monitoring the condition of estuaries is critical to their management, and is undertaken by most councils in Aotearoa New Zealand (hereafter New Zealand) as part of their State of the Environment (SOE) programmes using a variety of indicators. Much of the estuary SOE monitoring follows a widely-used National Estuary Monitoring Protocol (NEMP; Robertson et al. 2002). The NEMP is intended to provide resource managers nationally with a cost-effective, robust and standardised approach for monitoring the ecological status of estuaries in their respective regions, in particular intertidally-dominated estuaries with extensive tidal flats. The NEMP approach involves three main components:

- A regional prioritisation framework to identify which estuaries to monitor;
- A 'broad-scale' protocol for mapping estuary-wide intertidal habitats (e.g., substrate, salt marsh);
- A 'fine-scale' protocol for monitoring site-specific estuary sediment quality and associated biota.

The approach is intended to detect and understand changes in estuaries over time, with a particular focus on changes in habitat type (e.g., salt marsh or mud extent), as well as changes within habitats from the input of nutrients, fine (muddy) sediments and contaminants, which are key drivers of degraded estuary sediment condition.

As the NEMP has not been formally evaluated or revised since its first publication in 2002, Salt Ecology was contracted by MfE in 2022 to engage with scientists from regional councils, unitary authorities, and research providers, to collect high-level information on its current application (Roberts & Stevens 2023). Feedback identified, among other issues, inconsistency in sampling methods and analyses, high costs (for some indicators), the use of indicators that did not appear fit-for-purpose, a need to revise or add new indicators, uncertain linkages to drivers of change or management, and a lack of guidance for interpreting and consistently reporting state.

A specific recommendation was to review and update the NEMP, which is currently underway (Stevens et al. in prep.). There was also strong support for the NEMP approach to be extended, if feasible, to include guidance on non-compulsory thresholds and bands representing different states of ecological quality for commonly used estuarine indicators. Most ecological monitoring studies currently rely on expert interpretation of results, which may differ among studies and experts. Thresholds and bands have the potential to assist decision-makers and communities to:

i. Interpret monitoring results in a consistent way, which will facilitate understanding of temporal changes and enable comparisons among different locations;
ii. Better understand stressor-response relationships for NEMP indicators;
iii. Articulate the ecological quality objectives deemed worthy of protection/achievement;
iv. Understand the extent that current conditions are meeting such objectives.

While there was consensus that any recommended thresholds and bands should accompany the NEMP, most councils indicated that these would ideally sit within a separate guidance document that could be regularly updated as new information became available. This guidance is presented in the current report and represents an initial assessment of indicators and associated thresholds.

1.2 PROJECT OVERVIEW

In March 2024, MfE contracted Salt Ecology and NIWA to assess a suite of 19 commonly-used estuarine indicators selected by MfE (Table 1). These include indicators described in the NEMP (Robertson et al. 2002), and *ad hoc* extensions and improvements made to NEMP methods over time. Other indicators come from estuary monitoring tools that overlap with the NEMP (e.g., Estuary Trophic Index, ETI; Robertson et al. 2016a,b), initiatives proposed under National Planning Framework (NPF) and resource management reforms, and through the development of specific additional methods (e.g., CSIG seagrass monitoring protocol; Shanahan et al. 2023).

The project brief from MfE was to:

- Document the rationale for inclusion of proposed indicators, and any caveats associated with their use;
- Review existing thresholds and bands for each indicator metric, and provide advice on ecologically relevant thresholds and bands suitable for use in estuarine monitoring programmes in New Zealand;
- Indicate the degree of certainty and scientific robustness associated with any proposed thresholds (numerical or narrative) and, where appropriate, specify additional work needed to improve them.

Note that the current report does not attempt to prioritise or make specific recommendations on indicators for SOE programmes, and provides only a brief overview of monitoring methods (with references to more detailed information provided where available). The current report also does not represent an exhaustive list of estuary indicators, but includes the most commonly used ones in New Zealand. A concurrent MfE project (Lohrer et al. in prep.) is undertaking a 'stocktake' of 55 potential ecological indicators (across air, land, freshwater, estuaries, and coastal waters) to assess their potential utility in monitoring programmes, some of which may be applicable for threshold development. The Lohrer et al. (in prep.) report also provides substantial background on the indicators included in the current project and should be referred to for further detail.

Table 1. List of estuarine ecological indicators requested for assessment by MfE. The terms 'estuary-wide' and 'site-specific', are synonymous with the terminology relating to 'broad-scale' and 'fine-scale' indicators as used in the existing NEMP.

Habitat indicators (estuary-wide)
Macroalgae (opportunistic species)
Mangrove forest extent and quality
'Mud elevated' (>25% mud) sediment extent
Salt marsh extent and quality
Seagrass extent and quality
Shellfish bed extent and quality
Sediment indicators (site-specific)
Sedimentation rate
Macrofauna (community composition)
Microalgae (chlorophyll-*a* and phaeophytin)
Mud content
Nutrients (sediment N and P)
Organic matter
Depth to Redox Potential Discontinuity
Sulphur and sulphides
Trace metals
Water column indicators (site-specific)
Cyanobacteria
Dissolved oxygen
Nutrients (water column N and P)
Phytoplankton (chlorophyll-*a* concentration)

1.3 REPORT STRUCTURE

Each indicator (and associated metric/s) has been presented in a stand-alone appendix (Appendices A1 to A19) authored by subject experts listed in Table 2. This approach is intended to facilitate future updates on an indicator-by-indicator basis, allow the easy addition of new indicators over time, and enable a modular web-based reporting approach should that be considered in the future.

We aimed to achieve consistency in the content and quality of information provided through use of a template for each of the expert contributors to follow. For each indicator, expert/s prepared draft material which was internally reviewed within each organisation before submission to the project lead (Leigh Stevens; hereafter LS). Where appropriate, high-level feedback was provided by LS to each expert before drafts were sent for independent technical review to Mal Green (hereafter MG) at RMA Science. The technical review comments were then sent back to the subject experts for consideration, and indicators were finalised by subject experts. Following this review process, no changes were made to the final technical content submitted by the subject experts and the finalised indicators are included in the report appendices.

Below we provide a synthesis of the key findings and summarise recommendations made by the experts. We highlight the:

1. General rationale for inclusion of each indicator.
2. Indicators considered suitable for numeric thresholds.
3. Indicators that may require further development and/or are suited to a narrative threshold.
4. Indicators that require further development methodologically before thresholds can be considered, and/or indicators considered unsuitable for bands and thresholds.
5. Recommendations for further work needed to refine or update the above (e.g., comprehensive data analysis, research, etc.).

In undertaking the project there has been little scope to comprehensively review international literature, collate or analyse national data, or develop new thresholds. Preliminary data analyses to support literature findings have been undertaken where feasible, but otherwise the information supplied is drawn from existing work and knowledge from the project team's own studies, other New Zealand studies, and existing sources of information. For example, preliminary thresholds and associated bands for some of the estuary-wide and site-specific indicators were proposed as part of the Estuary Trophic Index (ETI) Toolbox project (Robertson et al. 2016a) and are revisited in this report, noting the ETI has received little review or validation. Preliminary thresholds and associated bands have also been proposed by Salt Ecology (e.g., Forrest et al. 2023; Stevens et al. 2023) for estuary-wide and/or site-specific indicators, based on ANZG (2018), FDGC (2012), Townsend and Lohrer (2015) and Stevens & Robertson (2014). Others have been assessed as part of council plans, the National Policy Statement for Freshwater Management (NPSFM), and the proposed National Planning Framework (NPF) and National Objectives Framework (NOF) initiatives (e.g., Managing Upstream; Cornelisen et al. 2017). The preference was to provide guidance on numeric thresholds and associated bands, accompanied by narrative descriptions. Where numeric thresholds were unavailable, MfE requested that narrative thresholds be included where appropriate.

2. INDICATORS AND THRESHOLDS

2.1 ECOLOGICAL INDICATORS

There are certain characteristics that make some ecological indicators more useful than others. These characteristics, described in detail by Sutula (2011), include;

- A clear link to beneficial monitoring uses;
- Predictive relationships with causal factors;
- Scientifically sound and practical measurement process;
- Acceptable signal-to-noise ratio that shows trends in ecological condition.

Further indicators would ideally:

- Be easy to understand (unambiguous to a non-technical audience);
- Provide an early warning of emerging problems;
- Be adaptable for use at a range of spatial scales;

Table 2. Technical experts contributing to each of the listed estuarine ecological indicators.

Appdx	Ecological Indicator	Author/s
	Habitat indicators (estuary-wide)	
A1	Macroalgae (opportunistic species)	Keryn Roberts[1]
A2	Mangrove forest extent and quality	Carolyn Lundquist[2]
A3	'Mud elevated' (>25% mud) sediment extent	Leigh Stevens[1]
A4	Salt marsh extent and quality	Leigh Stevens[1]
A5	Seagrass extent and quality	Leigh Stevens[1], John Zeldis[3]
A6	Shellfish bed extent and quality	Drew Lohrer[2], Carolyn Lundquist[2], Barrie Forrest[1]
	Sediment indicators (site-specific)	
A7	Sedimentation rate	Steph Mangan[3], Orlando Lam-Gordillo[2], Drew Lohrer[2]
A8	Macrofauna (community composition)	Barrie Forrest[1], Orlando Lam-Gordillo[2]
A9	Microalgae (chlorophyll-*a* and phaeophytin)	Steph Mangan[3]
A10	Mud content	Barrie Forrest[1], Leigh Stevens[1]
A11	Nutrients (sediment N and P)	Keryn Roberts[1]
A12	Organic matter	John Zeldis[3]
A13	Redox Potential Discontinuity	John Zeldis[3]
A14	Sulphur and sulphides	Keryn Roberts[1]
A15	Trace metals	Barrie Forrest[1], Don Morrisey[1]
	Water column indicators (site-specific)	
A16	Cyanobacteria	Keryn Roberts[1]
A17	Dissolved oxygen	John Zeldis[3]
A18	Nutrients (water column N and P)	Bruce Dudley[3], John Zeldis[3], David Plew[3]
A19	Phytoplankton (chlorophyll-*a* concentration)	Keryn Roberts[1], John Zeldis[3]

[1] Salt Ecology, Nelson; [2] NIWA, Hamilton; [3] NIWA, Christchurch

- Diagnose multiple causative factors;
- Show detectable trends in both directions (improving or degrading).

Not all indicators meet such criteria, however this does not disqualify them from use. Rather, it will often result in them being used in conjunction with other indicators as part of a weight-of-evidence approach. Indicators may therefore be usefully classified into three broad types as suggested by Sutula (2011):

Primary indicators, for which regulatory endpoints could be developed. Designation as a 'primary' indicator implies a high level of confidence, based on a wealth of experience and knowledge, about how the indicator causes or reflects an ecological response. An example of a primary indicator is the growth of opportunistic macroalgae in direct response to increased catchment nutrient inputs (e.g., Stevens et al. 2022).

Supporting indicators, provide supporting lines of evidence, but development of regulatory endpoints is not anticipated. However, use of the indicator and supporting evidence over time may sufficiently increase confidence such that it achieves 'primary' indicator status. An example of a supporting indicator is the measurement of sediment oxygenation, which may respond to multiple stressors, and could, for example, indicate adverse sediment impacts caused by organic matter enrichment.

Co-factor indicators, could be part of a routine monitoring programme, and are important for data interpretation and trend analysis, but are not used explicitly to make a diagnosis. Examples of co-factor indicators are river flow or rainfall data, which may, for example, explain changes in estuary flushing time and persistence of phytoplankton blooms.

In the current report we adopt the above terminology for the three broad types of indicators, although primarily address the first two.

2.2 ECOLOGICAL THRESHOLDS

2.2.1 General concepts

Most resource management systems are reactionary, responding to environmental stressors when they become noticeable rather than actively seeking them out before they cause problems (Kelley et al. 2014). Environmental management is more likely to meet its goals if it addresses thresholds of response to environmental stress explicitly (Kelly et al. 2015), although the definitive means of detecting an environmental threshold is to exceed it (Scheffer and Carpenter 2003). Effective management therefore requires some knowledge of specific ecological thresholds that, once crossed, move the system away from a 'desired state' (Groffman et al. 2006).

Groffman et al. (2006) describe the application of ecological threshold concepts in three main ways:

(1) Analysis of dramatic 'shifts in ecosystem state' or '**tipping points**' where a small change in a driver causes a marked change in ecosystem condition;

(2) Determination of '**critical loads**', which represent the amount of pollutant that an ecosystem can safely absorb before there is a change in ecosystem state and/or in a particular ecosystem function; and

(3) Analysis of '**extrinsic factor thresholds**', where changes in a variable at a large scale alter relationships between drivers and responses at a small scale.

2.2.2 Types of thresholds

In assessing the above, this report includes various numeric and narrative thresholds.

There are two types of **numeric threshold**. The first type is a particular value (or small range of values) of a **physical, measurable quantity** that corresponds to the boundary between different states or bands of ecological quality. An example is the ANZG (2018) sediment quality guideline values. A numeric threshold will normally be based on a strong scientific understanding of a direct ecological response to a stressor.

The second type of numeric threshold is a particular value (or small range of values) of a **nondimensional index** that corresponds to the boundary between different states or bands of ecological quality. The nondimensional index itself is typically composed of many constituent quantities, each with its respective units, arranged in a way to make the index nondimensional. An example is the 1-5 scale of the macroinvertebrate Benthic Health Model (BHM) that compares relative differences between estuaries.

A **narrative threshold** similarly marks the boundary between different states or bands of ecological quality. The difference is - unlike a numeric threshold - that not enough is known to precisely ascribe a value (or small range of values) to the threshold, or the indicator is more suited to a narrative description, for example both increases and decreases in mangrove extent can indicate a problem. As a result, the threshold is imprecise and is expressed in words as a narrative. Because the narrative threshold is imprecise, the bands that are delineated by the narrative threshold are also imprecise, for example, 'minor through to very high stress'.

Narrative thresholds (and the associated narrative bands) commonly describe the expected ecological outcome under different levels of pressure and are often used to assist in the interpretation of monitoring results and the identification of potential management priorities. In many instances they are refined over time as additional data are collected.

2.2.3 Challenges in defining thresholds

The most reliable and useful thresholds are those primary indicators for which a change in the measured value of the indicator is clearly linked to a cause. An example is the reduction in the areal extent of salt marsh habitat caused by anthropogenic pressures such as reclamation or vehicle damage. The cause of degradation (i.e., salt marsh loss) can be clearly determined, and thresholds for assessing change can be easily defined and agreed to (e.g., no further loss of salt marsh from the existing extent). In addition, the required management actions are clear (e.g., prevent further reclamation or vehicle access), and the beneficial outcomes of management will be certain (e.g., no further loss of salt marsh and/or recovery of degraded habitat).

Loss of estuarine salt marsh as a consequence of reclamation (left) and vehicle damage (right).

Other indicators, and associated thresholds, may be less clear-cut, indirect, non-linear or affected by multiple factors (including natural variability) making both the setting of thresholds, and management, far more complex. For example, excessive nutrient concentrations are known to cause nuisance algal growth (see adjacent photos), but direct measurements of water column or sediment nutrients can have high spatial and temporal variability and may not be good predictors of any ecological response.

The algal response to nutrients can also be affected by external factors such as physical scouring from floods, tides or waves, while other stressors (e.g., sea temperature change, rainfall frequency and severity, sediment oxygenation status) may also strongly influence how algal proliferations or related indicators respond at a site-specific scale.

Outputs from models (e.g., potential estuary nutrient concentrations) may correlate strongly with observed macroalgal extent and provide a useful proxy measure of expected state or vulnerability to change, even when there is uncertainty about the mechanisms underlying the relationship. Where results are less conclusive, a combination of indicators may be required to understand state, with less reliance placed on specific thresholds, and a 'weight of evidence' approach used across multiple indicators.

Extensive growths of nuisance macroalgae (*Gracilaria* spp.), New River Estuary, Southland.

Anoxic sediment and thick microalgal cover following die-off of nuisance macroalgae, New River Estuary, Southland.

2.2.4 Baseline state

For some indicators it may be appropriate to develop thresholds based on change from a '**baseline**' state. Appropriate cases include measures of habitat change (e.g., percentage loss of salt marsh), or change in sediment quality, in relation to a defined baseline. Anchoring thresholds to a baseline state has particular value in situations where meaningful absolute thresholds cannot be developed, or are highly uncertain in terms of the cause-effect relationship between the level of a stressor and the ecological response. For example, the natural extent of salt marsh or seagrass varies greatly across estuaries, and an absolute measure of extent is relatively meaningless in terms of understanding ecological condition. In contrast, a decrease in extent relative to a baseline state might be meaningfully indicative of ecological degradation.

Although it might seem an obvious choice, it is not always possible or practical setting the baseline to a 'natural' or 'pre-human-disturbance' state. The natural state is often difficult or impossible to discern due to a lack of data, or even to define, given that the natural state would be (or would have been) itself dynamic and subject to both short- and long-term variability. For some indicators, it might be preferable to set the baseline to some contemporary measured state, in which case thresholds may need to reflect the potentially degraded starting point (e.g., environmental conditions may already be compromised to a state unsuitable for biota that have historically existed there).

Hence, there are two main states that can be used as a baseline.

(1) A **natural state**, which is defined as the maximum potential physical extent of habitat, or the quality of sediment or water column metrics expected to be present under an unmodified catchment prior to human disturbance (i.e., equivalent to natural state).

(2) A **contemporary state** which may be derived from the first set of reliable measurements, accepting that these are unlikely to represent the 'true' natural state.

The means of deriving the baseline, and its reliability, will differ among indicators even where similar methods are used. For example, salt marsh extent can be estimated with reasonable confidence from historic aerial imagery and LiDAR, whereas for seagrass it is likely to be only the denser beds (~50% cover or greater) that will be discernible. If contemporary data are used to define the baseline it is important that they reflect relatively stable and representative conditions (e.g., do not reflect recent episodic storm or flood impacts). This may require multi-year sampling.

For many sediment quality indicators, defining a baseline state is problematic. Nonetheless, methods such as deep sediment coring, with analysis and dating of sediment layers (e.g., Handley et al. 2017, Hale et al. 2024), can assist in determination of 'natural' (pre-human) sediment conditions (e.g., mud and trace metal content).

Less-commonly used, but still legitimate and useful states that may also be used as baselines, are states estimated using **modelling**, and states estimated by **expert judgement**. Because these latter approaches have a relatively high level of uncertainty associated with them, baselines based on measured data are preferable.

There are also implications for using contemporary state as a baseline when setting management objectives. For example, if you have a highly degraded site, a threshold of 'no further loss/degradation' may be more readily achievable than for a more pristine site. Similarly, if most of a habitat has previously been lost, any further loss may be of far greater ecological significance. Finally, it is not aspirational to manage relative to a degraded baseline unless improvement to the degraded state is encouraged or required.

The relative extent of features being assessed is also important when interpreting results. For example, if an estuary has a large seagrass extent, a small percent decrease may represent a large area of loss. Conversely, if an estuary only has a very small seagrass extent, a small decrease may result in a very large percent loss.

Salt marsh extent, and temporal change, is a broad scale indicator included in the NEMP, Whanganui/Westhaven Inlet, Tasman.

Habitat change. Dense seagrass (foreground) and extensive beds of dying seagrass (background), Whanganui/Westhaven Inlet, Tasman.

The period between surveys is also important as a 5% loss over 1 year is a different rate of change to a 5% loss over 20 years. It is beyond the current scope to consider these matters in greater detail but is important that councils and providers keep these considerations in mind, with further work required if thresholds are to be scaled based on estuary size or state.

2.2.5 Estuary typology and catchment characteristics

The ability to predict responses to identified stressors and develop reliable thresholds will be strongly influenced by estuary size and typology, catchment geology, topography and land use activities, the extent of past modification, and the availability of data.

A general theme that emerges in many of the individual indicator assessments is the challenge in developing reliable thresholds where there is a low signal-to-noise ratio. For example, the high 'noise' caused by natural processes that affect estuary state can 'drown out' discernible responses to changes in anthropogenic pressures. Estuary typology and related factors are particularly relevant. For example, the ecological state of tidal river estuaries is strongly governed by interacting factors such as river flow variability, vulnerability to water column stratification, and the extent to which the estuary entrance becomes blocked off to tidal exchange (Forrest et al. 2024). In these types of systems, pronounced spatial and temporal variability in water column and substrate characteristics can make it difficult to discriminate between natural and anthropogenic drivers of ecological state (Forrest et al. 2024). These issues, and other aspects of estuary typology and catchment characteristics that are important for threshold development, are addressed as relevant in the technical appendices.

2.2.6 Use of thresholds

The previous sections highlight that there is no 'silver bullet' that allows a simple set of nationally consistent thresholds to be proposed. In many instances, multiple indicators will be needed to interpret results, and these should be applied at an estuary-specific scale and closely linked to the purpose of any monitoring and management being proposed.

Therefore, it is reiterated that thresholds presented in this report are not intended for use as individual regulatory targets or to assess compliance. Rather, the intent is to improve consistency in the type of approaches used to classify estuary state/condition, highlight the current status of different indicators and thresholds with regard to their potential for use, and recommend the type of additional work that may be required to improve them. This advice will sit alongside an updated NEMP, which will describe monitoring methods for each of the indicators.

2.2.7 Confidence in thresholds proposed

For the purposes of this report, thresholds proposed for consideration have been ascribed a confidence rating outlined in Table 3 reflecting the level of certainty associated with their application, alongside a description of how the proposed thresholds should be accepted and adopted. The criteria we use have been loosely based on the confidence levels used by Lohrer et al. (in prep.) in the assessment of 55 potential ecological indicators for their potential utility.

Table 3. Ratings applied to describe confidence levels in thresholds.

Rating	Description
Very high	Thresholds well established and based on comprehensive analysis/syntheses; multiple studies agree. Widely demonstrated utility. Further threshold development considered unnecessary other than ongoing review as additional monitoring data are collected.
High	Thresholds established and general agreement, but incomplete due to limited data/studies, particularly NZ-specific data. Thresholds considered preliminary and require refinement following review of existing NZ data and further collection of new data.
Fair	Thresholds based on expert judgement and some studies/data but conclusions inconsistent or potentially not applicable nationally. Thresholds considered suitable for guidance only, and potentially useful alongside other metrics. NZ data collection and analysis required to further develop thresholds or improve confidence.
Low	Thresholds based on a suggestion or speculation; no or limited evidence or data inconclusive. May be useful for further investigation into potential approaches, but substantial further development or data required. Suitable for use in interpretation alongside other metrics.
Undeveloped	No thresholds identified or proposed. No evidence to support development of thresholds, or very low likelihood that any proposed thresholds will be suitable for assessing estuary state. Not endorsed for further development.

3. INDICATOR RATIONALE

Tables 4 and 5 provide a high-level rationale for the inclusion of indicators by MfE for assessment in this report. Most are well described and have been widely used in monitoring programmes. However, as stated earlier, this report does not attempt to prioritise or make specific recommendations on indicator use within SOE monitoring programmes, and the inclusion of a specific indicator does not necessarily imply that it is considered either appropriate or should be a priority for use. In the Tables in this section, and in Section 4, the terms 'estuary-wide' and 'site-specific', are synonymous with the terms 'broad-scale' and 'fine-scale' applied to indicators in the NEMP.

Table 4. General overview of **habitat (estuary-wide) indicators** presented in this report.

INDICATOR	DESCRIPTION
Habitat indicators (estuary-wide)	
Macroalgae (opportunistic species)	Opportunistic macroalgae (e.g., species of *Gracilaria* and *Ulva*) are a symptom of eutrophication (nutrient enrichment). At nuisance levels, these algae can form mats across the intertidal area that can adversely impact underlying sediments and biota, fish, birds, seagrass, and salt marsh. The Opportunistic Macroalgal Blooming Tool (OMBT) is a multi-metric index that combines different metrics of macroalgae extent, cover and biomass and is an indicator of ecological condition.
Mangrove forest extent and quality	Mangrove forests provide a diversity of physical and ecological functions in New Zealand estuaries, contributing organic matter to food webs, providing habitat structure, mitigating against climate change through carbon sequestration, and serving as a natural defence against coastal hazards. Rapid expansion of mangroves is often an indicator of high rates of estuarine sedimentation. Areal extent is an efficient and cost-effective indicator of broad scale spatial and temporal changes. The size-structure of mangrove forests (e.g., tall, dwarf forms), can be useful when converting density-estimates to ecosystem services.
'Mud elevated' (>25% mud) sediment extent	The deposition of fine sediment (i.e., mud <63µm) can reduce habitat heterogeneity, concentrate contaminants, nutrients and organic matter, and lead to degradation of benthic communities by displacing sensitive species including shellfish, and smothering other habitats. Enrichment of muddy sediments (i.e., high TOC and nutrients) can additionally fuel algal growth and deplete sediment oxygen. The assumption is that increases in the spatial extent of 'mud elevated' (>25% mud) sediment will cause ecological damage.
Salt marsh extent and quality	Salt marsh (vegetation able to tolerate saline conditions where terrestrial plants are unable to survive) is important in estuaries as it is highly productive, naturally filters and assimilates sediment and nutrients, mitigates shoreline erosion, and provides habitat for a variety of species including insects, fish and birds. There is a high potential for displacement loss of salt marsh as a consequence of sea level rise. Decreases in the spatial extent of salt marsh will reduce these important values.
Seagrass extent and quality	Seagrass (*Zostera muelleri*) beds enhance primary production and nutrient cycling, stabilise sediments, elevate biodiversity, and provide nursery and feeding grounds for invertebrates and fish. Seagrass is vulnerable to muddy sediments in the water column (reducing light), sediment smothering (burial), excessive nutrients (primarily secondary impacts from macroalgal smothering), and sediment quality (e.g., low oxygenation). Seagrass responds to natural and human disturbances through changes in spatial extent, percent cover, density (number of plants), biomass and/or morphology (e.g., leaf length or width).
Shellfish bed extent and quality	Shellfish beds provide a diversity of physical and ecological functions including sediment bioturbation and oxygenation, sediment stabilisation, water filtering, and are a food source for fish, birds and humans. Extent and quality metrics can be used to track health at the population level, for example, by analysis and changes in biomass and population size structure (including recruitment). It is beyond present scope to address shellfish quality indicator thresholds at the individual level.

Opportunistic macroalgae in Moutere Inlet, Tasman 2019.

Table 5. General overview of **sediment and water column (site-specific) indicators** presented in this report.

INDICATOR	DESCRIPTION
Sediment indicators (site-specific)	
Sedimentation rate	Provides a measure of sediment deposition and erosion integrated over time (year or longer). Excess sedimentation can smother benthic biota. Sedimentation can be measured regularly at fixed sites using rods and plates, or infrequently using estuary-wide on-ground or remote survey methods (e.g., LiDAR, RTK).
Macrofauna (community composition)	The abundance, composition and diversity of infauna (including shellfish) living with the sediment are commonly-used indicators of estuarine health. Macrofauna provide a food source for birds and fish. By definition, macrofauna are usually regarded as those species retained on a 0.5mm mesh after sieving.
Microalgae (chl-*a* and phaeophytin)	Microalgal mats can be conspicuous (e.g., bright yellow or green colour on sediment surface) under enriched conditions, but lack demonstrated utility as a routine indicator. Measured as sediment chlorophyll-a and phaeopigments. Taxonomic composition is not typically undertaken due to clumped or patchy distributions.
Mud content	Mud content can be reliably measured by well-established laboratory methods. As well as mud itself being a stressor, the contaminant-holding capacity of sediments tends to increase with decreasing particle grain size. Targeting reductions in anthropogenic mud provides the main avenue for mitigating adverse ecological effects.
Nutrients (nitrogen and phosphorus)	Nitrogen, and to a lesser extent phosphorus, are important nutrients utilised in plant growth (e.g., macroalgae, seagrass, salt marsh and mangroves). Elevated nutrient concentrations can cause blooms in the growth of phytoplankton and macroalgae in estuaries and therefore play a key role in eutrophication.
Organic matter	Sediment organic matter is an indicator of eutrophication and includes carbon derived from plant and animal material. It is typically measured as Total Organic Carbon (%TOC). Production and decomposition of TOC can result in oxygen depletion and changes to other biogeochemical processes in sediments and overlying waters.
Redox Potential Discontinuity	The Redox Potential Discontinuity (RPD) is the boundary between oxic near-surface sediment and the underlying suboxic or anoxic sediment. The depth (from the sediment surface) to the RPD is used as a measure of the enrichment/trophic state of sediments.
Total sulphur and sulphides	The build-up of sulphides in sediment porewater is indicative of persistent anoxic conditions, a symptom of eutrophication. Sulphides can be toxic to fish and benthic macrofauna.
Trace metals	Arsenic, copper, chromium, cadmium, lead, mercury, nickel, and zinc are common toxic contaminants generally associated with human activities (although sometimes with natural sources), and are generally referred to collectively as trace metals or heavy metals. High concentrations may indicate a potential for toxic effects (ANZG 2018), and a need to investigate other anthropogenic contaminant types (e.g., pesticides, hydrocarbons).
Water column indicators (site-specific)	
Cyanobacteria	Cyanobacteria are a type of photosynthetic bacteria, commonly called blue-green algae. Some species produce toxins (cyanotoxins) that pose a health risk to humans and animals through ingestion (e.g., contaminated water and seafood), inhalation or dermal contact. Other negative effects of blooms include low dissolved oxygen, poor water clarity, benthic smothering, fish kills and altered biogeochemical cycling.
Dissolved oxygen	Oxygen levels are controlled by a balance between photosynthesis, aeration/ mixing and consumption by respiration. Oxygen is an indicator of the suitability of a water body for aquatic life. Depleted water column oxygen can adversely impact sediment-dwelling and water column communities, and is a primary cause of most fish kills. Low oxygen levels can also trigger the release of sediment bound nutrients to the water column, promoting secondary eutrophic symptoms (e.g., algal blooms).
Nutrients (nitrogen and phosphorus	Water column nutrient concentrations provide a metric that is sensitive enough to detect broad spatial and temporal changes in nutrient loads to estuaries, and eutrophication impacts of those loads. Nitrogen is expected to limit algal growth in most estuaries. Dissolved forms (ammoniacal-N, nitrate, nitrite) can be readily assimilated by algae. Ammoniacal-N has a temperature-dependent toxicity to fish and other aquatic organisms. Phosphorus is a key nutrient associated with the growth of plants and algae, especially in freshwaters. It is not a particularly useful indicator because phosphorus is not typically limiting to primary production in estuaries.
Phytoplankton (chl-*a* conc.)	Phytoplankton respond to nutrients and excess chlorophyll-*a* can be an indicator of phytoplankton blooms. Elevated nutrients and low flushing can facilitate rapid algal growth and high oxygenation from photosynthesis, but can deplete dissolved oxygen levels when algal blooms crash and die and the resulting organic matter is consumed by respiring animals.

4. SUMMARY OF TECHNICAL ASSESSMENTS

In this report, the intent is to document available knowledge relevant to New Zealand in order to assist decision-makers understand how commonly used monitoring indicators characterise estuary ecological condition, and to highlight the extent to which changes in indicators might be linked to thresholds for key stressors which can be used to inform management.

Summaries of indicators, measurement metrics (the specific methods used to assess indicators), recommendations, and comments relating to the use or development of thresholds are presented in Tables 6-8. Tables 9-11 summarise recommended numeric thresholds contained in the Technical Appendices. It is important that the caveats and guidance contained in the Technical Appendices are considered when applying any thresholds.

Table 6. Summary of **habitat (estuary-wide) indicators**, with associated metrics, type, and threshold confidence rating assigned by subject matter expert/s.

Indicator	Metric	Type	Confidence	Comment
Macroalgae	Opportunistic macroalgal abundance (OMBT-EQR)	Primary	High	Preliminary thresholds with well documented link between macroalgal blooms and increases in nutrient inputs and/or availability. Potential for regulatory thresholds to be established following analysis/review of New Zealand data.
Mangroves	% change in mangrove forest extent from baseline	Supporting	Fair	Narrative thresholds proposed, with potential for numerical thresholds following an assessment of existing New Zealand data.
	% change in proportion of tall vs dwarf mangroves	Supporting	Undeveloped	Metric a correlative indicator of mangrove quality but considered insufficiently developed to enable threshold development.
'Mud-elevated' (>25% mud) sediment	% of intertidal area with mud-elevated sediment	Supporting	Fair	Guidance only thresholds based on expert judgement, but require refinement based on an assessment of existing New Zealand data.
	% increase of intertidal mud-elevated sediment from first accurately measured baseline	Primary	High	Preliminary thresholds based on expert judgement and suited to the early detection of change within limits of method accuracy. Require refinement based on an assessment of existing New Zealand data.
Salt marsh	% of available salt marsh habitat (ASH)	Supporting	Fair	Thresholds for guidance only. Based on international metrics, but require validation based on an assessment of existing New Zealand data. Threshold development required for estuaries with mangroves.
	% loss from first accurately measured baseline	Primary	High	Preliminary thresholds based on expert judgement, but suited to the early detection of contemporary change. Refinement recommended following an assessment of existing New Zealand data.
	% loss from estimated historical extent	Supporting	Fair	Preliminary thresholds based on international guidance and expert judgement. Refinement recommended following an assessment of existing New Zealand data.
	Quality (multiple metrics)	Supporting	Undeveloped	High level narrative thresholds have potential to be developed as preliminary screening criteria to help determine if more detailed investigation is warranted.
Seagrass	% loss of dominant (>50% cover) intertidal seagrass from first accurately measured baseline	Primary	High	Preliminary thresholds based on expert opinion and intended as an early indicator of contemporary seagrass loss. Refinement recommended following an assessment of existing New Zealand data.
	% reduction in area-weighted average % cover (density) of inter-tidal seagrass >10% cover	Supporting	Fair	Guidance only thresholds based on international criteria which appear permissive based on observed temporal changes in New Zealand seagrass density. Assessment by repeat measurements can be related to anthropogenic disturbance.
	Quality (multiple metrics)	Supporting	Low	Thresholds for selected quality metrics proposed as preliminary screening criteria to help determine if more detailed investigation is warranted and to assess if potentially suitable for further development.
Shellfish beds	% loss from estimated historical extent	Supporting	Low	Preliminary thresholds based on expert judgement. Refinement recommended following an assessment of existing New Zealand data.
	Quality (Health)	Supporting	Low	Metric considered insufficiently developed to enable threshold development. Research into indicators of shellfish health should be encouraged.

Table 7. Summary of **site-specific sediment indicators**, with associated metrics, type, and threshold confidence rating assigned by subject matter expert/s.

Indicator	Metric	Type	Confidence	Comment
Sediment accretion rate (SAR)	Annual average change in sediment level (site-specific)	Supporting	Fair	Guidance only thresholds based on ANZECC estuarine sedimentation DGV. Further research required to better establish relationships between SAR and ecological health.
Macrofauna (community composition)	AZTI's Marine Biotic Index	Supporting	High	Preliminary thresholds based on widespread international acceptance and use of the index. Integrative indicator of multiple stressors. Further work required to develop reliable and agreed eco-groups for New Zealand taxa.
	National Benthic Health Models (BHM)	Supporting	Fair	Preliminary thresholds of state relative to other estuaries. Further testing and refinement needed, particularly using within-site time series data, where marked changes in mud or metals levels have occurred, to evaluate efficacy for council SOE monitoring.
	Traits Based Index (TBI)	Supporting	Fair	Preliminary thresholds with High confidence for use in Auckland and Waikato regions, but Fair elsewhere. Further validation needed to assess national scale application and improve threshold resolution. Currently most suitable for assessing within-site temporal change.
Microalgae	Sediment microalgae (chlorophyll-a and phaeopigments)	Supporting	Low	Not endorsed for threshold development. There are some studies/data, but large spatial and temporal variability make banding into thresholds inaccurate.
Mud content	Sediment mud content (%)	Primary	High	Preliminary thresholds with general agreement from multiple studies. Would benefit from an analysis of collated national data that focused on threshold development.
Nutrients (Sediment nitrogen and phosphorus)	Total nitrogen sediment concentration	Supporting	High	Preliminary thresholds with general agreement from multiple studies. Use as a supporting indicator for eutrophication alongside others (e.g., macroalgae, TOC, mud content, RPD depth).
	Total phosphorus sediment concentration	Supporting	Low	Not endorsed for threshold development as unlikely to be a major driver of estuary eutrophication. Further analysis/review of data from NZ and elsewhere is required to properly assess thresholds.
Organic matter	Total Organic Carbon (%TOC)	Supporting	High	Preliminary thresholds sensitive to broad spatial and temporal changes. Consider influence in driving eutrophication alongside other indicators (e.g., macroalgae, mud content, RPD depth.
Depth to RPD (Redox Potential Discontinuity)	Depth from the sediment surface to the RPD	Supporting	High	Preliminary thresholds for RPD depth indicate an effect on sediment health, but is conditioned by other estuary characteristics potentially operating independently or in concert, including grain size, organic content, and primary producer and faunal community compositions.
Sulphur and sulphides	TOC:TS	Supporting	Fair	Guidance only thresholds pending further data collection and analysis in New Zealand estuaries to determine appropriateness of proposed TOC:TS thresholds, and to determine if indicator should be restricted to depositional areas or applied estuary-wide.
	Degree of Pyritization (DOP)	Supporting	Low	Not endorsed. Inconclusive thresholds proposed in international literature and, to our knowledge, no local data are available to make a further assessment. Further, the complexity of the laboratory approach could potentially be cost prohibitive to councils.
Trace metals	Trace metal concentration in bed sediment	Primary	High	Preliminary thresholds based on ANZG (2018) guidelines. As adverse ecological effects could potentially manifest at concentrations <DGV, more in-depth analysis of New Zealand field data is recommended to validate 'Very good' to 'Fair' thresholds.

Table 8. Summary of **site-specific water column indicators**, with associated metrics, type, and threshold confidence rating assigned by subject matter expert/s.

Indicator	Metric	Type	Confidence	Comment
Cyanobacteria	Planktonic cyanobacteria (human health) biovolume or cell counts	Supporting	Very high (for ICOLLS) Fair (for other typologies)	Adopt ICOLL thresholds presented in the New Zealand Guidelines for Cyanobacteria in Recreational Freshwaters (in press). For other typologies, thresholds are feasible for human health indicators, but require a thorough review and further consideration of tidal state, mixing status, stratification, and depth to be reliable.
	Benthic cyanobacteria (human health) % cover	Supporting	Low Investigative	In principle, a useful human health indicator, however a data deficit in New Zealand estuarine and coastal waters currently limits its development.
Dissolved oxygen	Dissolved oxygen (DO)	Primary	High	Preliminary thresholds with general agreement from multiple studies. Uncertainties on precision of settings of DO thresholds and time and space scales over which they are assessed. Analysis/review of data from NZ and elsewhere recommended, particularly on oxygen tolerances for NZ native species.
Nutrients (water column nitrogen and phosphorus)	Potential nutrient concentrations: DIN, SRP, TN, TP	Primary	High	Modelled 'potential' (in the absence of processes that may produce or consume) nutrient concentrations relate well to spatial patterns of eutrophication effects, therefore provide a metric sensitive enough to predict impacts of nutrient loads to estuaries in New Zealand.
	Measured nutrient concentrations: DIN, SRP, TN, TP	Supporting	Fair	Guidance only. No nationally applicable, field-effects based guideline values developed for measured nutrient concentrations. Region-specific baselines recommended for open coastal waters with minimal anthropogenic influence to better define coastal nutrient loads to estuaries.
Phytoplankton (chlorophyll-*a* concentration)	Phytoplankton biomass	Primary	High	Preliminary thresholds with general agreement in the international literature, but limited local data. Additional data required across a range of estuary types (including euhaline systems) to assess the suitability of the proposed thresholds to New Zealand estuaries.

Phytoplanktom bloom beneath stratified freshwater surface layer, Ohau River Estuary, Manawatu.

Table 9. Summary of **habitat (estuary-wide) indicator thresholds** proposed by subject matter experts relative to expected ecological quality status. See Technical Appendices for caveats and guidance on threshold use.

Indicator	Metric	Ecological Quality Status				
		Very Good	**Good**	**Fair**	**Poor**	**Very Poor**
Macroalgae	OMBT-EQR^	≥0.8 to 1.0	≥0.6 to <0.8	≥0.4 to <0.6	≥0.2 to <0.4	0.0 to <0.2
Mud-elevated	% of intertidal area with mud-elevated sediment*	<1%	1 to <5%	≥5 to <15%	>15%	>25%
	% increase of intertidal mud-elevated sediment from baseline*	≤0	>0 to <5%	≥5% to <10%	≥10% to <20%	≥20%
Salt marsh	% of available salt marsh habitat (ASH)#^	≥50%	≥25 to <50%	≥10 to <25%	≥5 to <10%	0 to <5%
	% loss from first accurately measured baseline*	≤0	>0 to <5%	≥5% to <10%	≥10% to <20%	≥20%
	% loss from estimated historical extent#	<0 to <20%	≥20% to <40%	≥40% to <60%	≥60 % to <80%	≥80% loss
Seagrass	% loss of dominant (>50% cover) seagrass from first accurate baseline*	≤0	>0 to <5%	≥5% to <10%	≥10% to <20%	≥20%
	% reduction in average %cover (density) (annual mean)#	>0 to ≤10%	>10% to ≤30%	>30% to ≤50%	>50 % to ≤70%	>70% loss
	% reduction in average %cover (density) (5-yr rolling mean)#	>0 to ≤5%	>5% to ≤15%	>15% to ≤25%	>25 % to ≤35%	>35% loss
Shellfish	% loss from estimated historical extent*	<0 to <10%	≥10% to <20%	≥20% to <50%	≥50 % to <75%	≥75% loss

*Thresholds based on expert judgement. #Based on WFD. ^Note scaling is high to low.

Table 10. Summary of **site-specific sediment indicator thresholds** proposed by subject matter experts relative to expected ecological quality status. See Technical Appendices for caveats and guidance on threshold use.

Indicator	Metric	Ecological Quality Status				
		Very Good	**Good**	**Fair**	**Poor**	**Very Poor**
Accretion	SAR (mm/yr) if assumed natural SAR ≤1mm/yr	---------------------	0 to 1	≥1 to 3	≥3 to 10	≥10
	SAR (mm/yr) above natural SAR	---------------------	0	>0 to 2	≥2 to 10	≥10
Macroinvertebrates	AMBI score*	0 to 1.2	>1.2 to 3.3	>3.3 to 4.3	>4.3 to 5.0	>5.0 to 7
	TBI Score#^	---------------------	>0.4 to 1	≥0.3 to 0.4	0 to 0.2	---------------------
Mud content	% sediment mud content	<5	≥5 to <10	≥10 to <25	≥25 to <50	≥50
Nutrients	Sediment TN (mg/kg)	<250	≥250 to <800	≥800 to <1200	≥1200 to <2000	≥2000
Organic matter	TOC (%)	<0.5%	0.5 to 1.2%	>1.2% to 2%	>2%	---------------------
RPD	RPD depth (mm)^	>40	40 to >25	25 to 10	<10	---------------------
Total Sulphur	TOC:TS^	---------------------	>5	≤5 to 1.5	<1.5	---------------------
Trace Metals	Threshold 'rules' relative to ANZG (2018) SQGVs	<25% DGV	25 to <50% DGV	50% DGV to <DGV	DGV to <GV-High	≥GV-high
	As (mg/kg)	<5	5 to <10	10 to <20	20 to <70	≥70
	Cd (mg/kg)	<0.38	0.38 to <0.75	0.75 to <1.5	1.5 to <10	≥10
	Cr (mg/kg)	<20	20 to <40	40 to <80	80 to <370	≥370
	Cu (mg/kg)	<16	16 to <32.5	32.5 to <65	65 to <270	≥270
	Hg (mg/kg)	0.038	0.038 to <0.075	0.075 to <0.15	0.15 to <1	≥1
	Ni (mg/kg)	<5.25	5.2 to <10.5	10.5 to <21	21 to <52	≥52
	Pb (mg/kg)	<12.5	12.5 to <25	25 to <50	50 to <220	≥220
	Zn (mg/kg)	<50	50 to <100	100 to <200	200 to <410	≥410

* AMBI 7-point scale mapped to 5-band scale. #TBI score mapped to equivalent Ecological Quality Status band. ^Note scaling is high to low.

Indicator	Metric	Level of impact relative to other estuarine sites in New Zealand				
		Very low	**Low**	**Moderate**	**High**	**Very High**
Macroinvertebrates	BHM score	1 to <2	2 to <3	3 to <4	4 to <5	≥5

Table 11. Summary of **site-specific water column indicator thresholds** proposed by subject matter experts relative to expected ecological quality status. See Technical Appendices for caveats and guidance on threshold use.

Indicator	Metric		Ecological Quality Status				
			Very Good	**Good**	**Fair**	**Poor**	**Very Poor**
Dissolved Oxygen	7-day mean minimum (mg/L DO)^		≥7.0	7.0 to ≥6.0	6.0 to ≥5.0	<5.0	---------------------
Potential TN (mg/m^3)	Corresponding to macroalgal OMBT bands		<175	175 to ≤335	335 to ≤ 495	>495	---------------------
Potential TN (mg/m^3)	Corresponding to phytoplankton bands	Euhaline (>30ppt)	<30	≥30 to 75	≥75 to 110	>110	---------------------
		Meso/polyhaline (≥5-30ppt)	<45	≥45 to 90	≥90 to 145	>145	---------------------
		Oligohaline (<5ppt)	<90	≥90 to 225	≥225 to 530	>530	---------------------
Potential TP (mg/m^3)	Corresponding to phytoplankton bands	Euhaline (>30ppt)	<4	≥4 to 10	≥10 to 15	>15	---------------------
		Meso/polyhaline (≥5-30ppt)	<6	≥6 to 12	≥12 to 20	>20	---------------------
		Oligohaline (<5ppt)	<12	≥12 to 30	≥30 to 75	>75	---------------------
Phytoplankton Biomass: Chlorophyll-a (mg/m^3)	90th percentile	Euhaline (>30ppt)	≤ 3	>3 to ≤8	>8 to ≤12	>12 to ≤16	>16
	90th percentile	Meso/polyhaline (≥5-30ppt)	≤ 5	>5 to ≤12	>12 to ≤16	>16 to ≤32	>32
	Annual median	Oligohaline (<5ppt)	≤2	>2 to ≤5	>5 to ≤12	>12 to ≤30	>30
	Annual maximum	Oligohaline (<5ppt)	≤10	>10 to ≤25	>25 to ≤60	>60 to ≤150	>150

^Note scaling is high to low.

5. FINAL REMARKS AND RECOMMENDATIONS

5.1 GENERAL STATUS OF THRESHOLDS

Estuary-wide

Most of the broad-scale (estuary-wide) habitat-related indicators have well-established NEMP methods suited to consistent and cost-effective data collection and reporting. The exception is shellfish extent which requires characterisation of features not visible from the surface, and therefore requires intensive sampling approaches.

Thresholds are well-established for macroalgae with a demonstrated linkage to catchment nutrient inputs (see Stevens et al. 2022). Thresholds proposed for metrics relating to salt marsh, seagrass, mangroves and mud-elevated sediment extent are based primarily on a percent change from a contemporary or historical (between contemporary and natural) baseline. These are recommended for use as guidance only and all would benefit from the collation and review of existing New Zealand data to validate and refine thresholds.

The ETI Toolbox project (Robertson et al. 2016) suggested spatial extent thresholds (based on hectares of habitat, or a temporal change in hectares) for some metrics, but these have not been proposed here. Such an approach has limited value due to the wide variation in spatial extent that is known to occur under natural state conditions, but may have some utility where changes in absolute area are more meaningful for management purposes than percent change alone.

Site-specific

As above, most of the site-specific sediment indicators have well-established NEMP methods, and many thresholds are considered suitable for preliminary adoption. However, all of the site-specific indicators have limitations with regard to estuary-wide extrapolation, meaning site selection is a critical determinant of results. The use of estuary-wide randomised or stratified sampling approaches could largely address this issue, however the associated costs are potentially limiting for many councils, particularly for indicators for which high levels of sample replication are commonly required, e.g., macrofauna.

5.2 CONSISTENCY IN THRESHOLD SCALING

The Technical Appendices reveal inconsistencies in approaches to threshold setting that we consider desirable to resolve, as follows:

- The number of bands (and therefore thresholds) proposed differs among indicators. For example, five bands are proposed in many cases, with band descriptors of 'Very good', 'Good', 'Fair', 'Poor', and 'Very poor'. However, for some indicators, different bands are proposed (e.g., three bands for the macrofauna Traits-Based Index).
- Different indicators may be differently scaled. For example, macrofauna indicators range from 0-1 (TBI), 1-5 (BHM), and 0-7 (AMBI). It would be possible to standardise ranges. For example, the 'parent' (i.e., original) scale for the macrofauna biotic index AMBI is 1-7, which we have mapped to a 1-5 scale. However, we recognise that re-scaling would be inappropriate in some cases.
- The interpretation of effects relative to the minimum and maximum values in each indicator scale differs. In some cases the lowest value in the scale represents the poorest state (e.g., macrofauna TBI scores) and in other cases the best (e.g., macrofauna AMBI and BHM).
- It would be ideal to express percent change on a consistent time scale where the time period between surveys is variable. Thresholds of change may also need to be different for assessing short-term annual change and longer-term change e.g., from an historical baseline.
- It is recommended that the threshold between the Fair/Poor boundary equates to where there is a high risk of ecological function being significantly impaired, difficult to reverse, or close to where a tipping point (i.e., regime shift) is predicted.

For clarity in interpretation of ecological effects, it is desirable that these issues are addressed. In particular, our experience is that reports on estuary state can be confusing to readers when the number of bands differs among indicators and the interpretation of effects relative to the minimum and maximum values in each indicator scale differs. There is an appealing simplicity in presenting a single rating matrix in which each indicator is scored across the same number of bands, uses the same descriptors (i.e., 'good', 'poor', etc.), and for which minimum and maximum values are interpreted in the same way. This type of approach provides consistency and clarity, and a colour coding scheme can be used to assist with conveying a visual impression of estuary state. For example:

Descriptor	Very Good	Good	Fair	Poor	Very Poor
Numeric indicator	Lowest value	→	→	→	Highest value

5.3 PRIORITIES FOR FURTHER WORK

As part of this project, each subject expert was asked to include recommendations for further work (collated in Appendix 20). Because author recommendations for each Appendix have largely been made independently of other experts, it is necessary to determine priorities for ongoing effort. We suggest priority be given to indicators with direct links to management and which also relate to the most ecologically damaging estuary stressors that Councils can manage or mitigate, i.e., fine sediment, nutrients, and habitat loss/displacement. A short list of suggested priorities is presented below, but we recommend a more thorough appraisal of priorities be undertaken and agreed to by national consensus.

5.3.1 Collation of existing data

A common theme across indicators was a recommendation to collate existing data in a consistent manner to facilitate analysis. Substantial data have been collected, particularly with regard to sediment indicators, which is held by a number of different providers. While much of this is now included as part of standardised national reporting of estuarine monitoring data on the Land, Air, Water Aotearoa (LAWA) a more comprehensive collation of data is required to undertake a national analyses for the purpose of developing thresholds. Habitat scale data (e.g., seagrass, salt marsh, mangrove, mud extent) are less standardised in their collection, and are not currently included as part of LAWA reporting nor have they been collated at a national level.

5.3.2 Analysis of existing data

There is great utility in analysing existing data to improve understanding of method consistency and accuracy (and the ability to detect change), to refine proposed thresholds on the basis of measured changes, and to assess the responsiveness of indicators to changes in stressors and corresponding ecological condition. This is particularly important where thresholds are based on international guidance and require local validation. Data analysis will also inform key sampling and reporting requirements with regard to sample replication, stratification, frequency of sampling, and potential redundancy of some indicators.

5.3.3 Site consistency and data extrapolation

Guidance is needed on site selection to ensure consistency in the collection of site-specific indicators, or to highlight where direct data comparisons are potentially unsuitable. To address limitations regarding estuary-wide extrapolation from site-specific indicators, estuary-wide randomised or stratified sampling approaches need to be developed.

5.3.4 Refinement of thresholds

Priority indicators for further threshold development are considered to be sediment TOC, TN, mud and metals (at a site-specific scale), and macroalgae, seagrass and salt marsh (at an estuary-wide scale). The latter, in particular, have received relatively limited attention to date, and international thresholds appear to over-estimate seasonal variation observed in New Zealand seagrass and salt marsh.

Further assessment and development of percent change thresholds for habitat scale indicators (e.g., salt marsh, seagrass, mangroves or mud extent), is recommended, as is assessing the national applicability of proposed thresholds to estuaries with mangroves.

5.4 RECOMMENDATIONS

The current project is therefore a significant step toward a national approach for clear and consistent interpretation, communication and reporting on the ecological health of estuaries, but is not the endpoint.

Based on the discussion above, the following recommendations are made:

- Prioritise (by national consensus) further development of indicators with direct links to management and which relate to the most ecologically damaging estuary stressors i.e., fine sediment, nutrients, and habitat loss/displacement.
- Where appropriate, adopt a 5-band threshold structure (i.e., Very good to Very poor).
- Collate existing national data and undertake analysis of relationships between indicators, stressors, and ecological responses as a high priority for supporting and refining proposed thresholds.
- Give initial priority to refining: (i) site-specific thresholds for sediment TOC, TN, mud and metals, as well as macrofauna BHM and AMBI, and (ii) estuary-wide thresholds for macroalgae, seagrass and salt marsh indicators.
- MfE provide specific guidance to Councils on how these thresholds should be applied or adopted.

It is emphasised that the purpose of developing and refining thresholds is to assist councils to interpret SOE monitoring data and guide timely and effective management actions. Hence, the level of scientific certainty is less critical than it would be for the development of regulatory or compliance thresholds.

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APPENDIX 1. MACROALGAE

Author: Keryn Roberts (Salt Ecology)

Macroalgae, also known as seaweed, is ubiquitous in coastal and nearshore marine environments. Its rapid growth in response to fluctuations in nutrient concentrations, tangible link to manageable anthropogenic inputs (e.g., nutrients), and established method of measurement, make it a useful eutrophication indicator.

BACKGROUND

Macroalgae are large multicellular photosynthetic organisms that are among the most productive primary producers, undertaking photosynthesis to produce energy, oxygen and organic matter, and are an important food source at the base of the food web (Mann 1973; Sutula 2011). They also provide refuge for invertebrates, juvenile fish, crabs and other species (Sutula 2011; Borja et al. 2012).

Macroalgae thrive in nutrient-enriched waters and, when combined with suitable growing conditions, nuisance blooms of rapidly-growing species can occur (Scanlan et al. 2007; Sutula 2011; Borja et al. 2012; Sutula et al. 2014; Woodland et al. 2015; Lapointe et al. 2018). The most common nuisance species in New Zealand are the native red seaweed *Gracilaria* spp. (previously named *Agarophyton* spp.) and the bright green seaweed *Ulva* spp. (commonly known as 'sea lettuce'; Robertson et al. 2016). Effects of nuisance macroalgal blooms include aesthetic impacts from odour and deposition of drift algae on shorelines, interference in water use activities, and changes to the feeding behaviour of birds (WFD-UKTAG 2014). More significantly, smothering growths of macroalgae can create degraded sediment conditions (i.e., low sediment oxygenation), disrupt biogeochemical cycling, cause fish and invertebrate mortality, reduce biodiversity, and outcompete seagrass (Sutula 2011; Bittick et al. 2018 and references therein). Persistent beds of entrained macroalgae (i.e., growth within the sediment matrix) typically become dominated by soft, muddy sediments because near-bed current velocities decrease in macroalgal mats of increasing cover, promoting sediment deposition (Romano et al. 2003). The co-accrual of fine sediments can lead to secondary sediment-related adverse effects including changes in sediment nutrient and oxygen fluxes, decreased water clarity, smothering of seagrass beds, and impacts to sediment macrofauna (Thrush et al. 2004 and references therein).

Macroalgae is a useful eutrophication indicator because: (1) the link between intertidal macroalgal blooms and increases in nutrient inputs and/or nutrient availability have been well documented globally (e.g., Valiela et al. 1997; Valiela & Bowen 2002; Howarth 2008; Sutula et al. 2014; WFD-UKTAG 2014; Woodland et al. 2015) and in New Zealand (e.g., Robertson et al. 2017; Barr et al. 2020; Plew et al. 2020; Zeldis et al. 2020; Dudley et al. 2022; Stevens et al. 2022), and (2) it effectively integrates nutrient availability over a period of days to weeks, providing a more stable indicator of eutrophication than nutrient concentrations, for example, which can be episodic, highly variable in space and time, and influenced by consumption and production processes.

While nuisance macroalgal growth is regulated by nutrient concentrations, as discussed above, other factors including salinity, hydrology, physical disturbance, light availability, temperature, and grazing pressure are also important (Sutula 2011; WFD-UKTAG 2014; Plew et al. 2020). As a result, macroalgae can be spatially and temporally variable. To effectively utilise macroalgae as an indicator requires documenting change over both time and space to strengthen the ability to detect trends (e.g., repeat monitoring at the same time of year over several years).

PROPOSED METRICS

The following indicator metric is proposed for monitoring macroalgae:

- Opportunistic Macroalgal Blooming Tool (OMBT), a multi-metric index for macroalgal abundance.

1.1 OPPORTUNISTIC MACROALGAL ABUNDANCE (OMBT-EQR)

Indicator type: Primary.

Metric: Opportunistic Macroalgal Blooming Tool (OMBT) Ecological Quality Rating (EQR) – an overall multi-metric index score between 0 (major disturbance) to 1 (minimally disturbed).

Unit of measurement: No unit.

Spatial scale: Estuary-wide or in very large estuaries distinct arms can be surveyed separately (e.g., Kaipara Harbour).

Applicability: Intertidally dominated estuaries (e.g., SIDE or SSRTREs with SIDE characteristics).

Rationale: Several international estuary monitoring programmes have selected macroalgae as a primary eutrophication indicator in estuaries (e.g., Sutula 2011; WFD-UKTAG 2014) because of its important ecosystem function (i.e., food-web support), acceptable signal to noise ratio and measurable response to catchment nutrient loads and other management controls (e.g., point sources, flushing time, habitat protection).

The Estuary Trophic Index (Robertson et al. 2016) recommended a multi-metric index to assess opportunistic macroalgal abundance in New Zealand estuaries based on the Opportunistic Macroalgal Blooming Tool (OMBT) established for the European Water Framework Directive (WFD-UKTAG 2014). The OMBT was chosen, in preference to individual macroalgae indicators, because it incorporates macroalgal percent cover, biomass, and level of entrainment (i.e., growth within the sediment matrix) into one index. The multi-metric is a more reliable measure of estuary degradation than percent cover alone (Scanlan et al. 2007; Sutula 2011; WFD-UKTAG 2014) because a thin layer (low biomass) of cover does not have the same negative effects on underlying sediments and biota as a thick layer (high biomass) of cover. Where opportunistic macroalgae is the primary eutrophication response rather than phytoplankton (e.g., in estuaries with large intertidal areas rather than subtidally dominated estuaries), New Zealand studies is estuaries without mangrove have demonstrated a link between the OMBT-EQR score and nutrient inputs, making it a useful indicator for management (e.g., Robertson et al. 2017; Plew et al. 2020; Stevens et al. 2022).

Method: Briefly, the National Estuary Monitoring Protocol (NEMP; Robertson et al. 2002) broad-scale mapping approach, including refinements by Salt Ecology (Stevens et al. 2023), is used to map the spatial extent of macroalgae. In each mapped macroalgae patch, percent cover, wet-weight biomass and entrainment (i.e., stable growth within the sediment matrix) are recorded (see Roberts et al. 2022b). The approach combines the use of aerial imagery, detailed field ground-truthing, point sampling (i.e., biomass and entrainment) and post-field digital mapping using Geographic Information System (GIS) technology. Quality Assurance/Quality Control (QA/QC) procedures include checking for duplicated or overlapping GIS polygons, identification of gaps or slivers (i.e., small polygons) and validation of field codes and field data (i.e., biomass, percent cover, entrainment).

Example of quadrat sampling for wet weight biomass (left and middle) and an example of an entrained mat of macroalgae (right).

The OMBT-EQR combines these different measures of opportunistic macroalgal proliferation (i.e., area, percent cover, biomass, entrainment) across the Available Intertidal Habitat (AIH*) into an integrated measure of ecological condition (WFD-UKTAG 2014; Stevens et al. 2022). While the original method was described in the Water Framework Directive (2014), small method improvements have been made to reflect its use in New Zealand (see Plew et al. 2020; Roberts et al. 2022b; Stevens et al. 2022). As described above, percent cover alone does not provide an accurate representation of the impacts of macroalgae on sediment condition and benthic macrofauna. However, w here biomass and entrainment are not measured, percent cover thresholds from the OMBT sub-metric can be used to provide a preliminary assessment of estuary condition.

Remote-sensing methods are being explored under an Envirolink Tools project, however, at present they remain under-development and are currently not suitable for calculating the OMBT-EQR.

**The AIH is the area of intertidal habitat that is suitable for macroalgal growth (see Scanlan et al. 2007). For New Zealand estuaries we recommend the AIH include the entire intertidal area excluding salt marsh and, where applicable, mangroves. While some areas (e.g., channel edges subject to scouring, rocky habitat) may not be suitable for growth, field data indicate that these features generally comprise only a small part of the intertidal area.*

Assessment baseline: The most ecologically relevant baseline is temporal change from natural or unimpacted state. However, in most cases it cannot be directly measured due to direct (e.g., reclamation) and/or indirect (e.g., nutrient inputs) estuary modification. In these instances, a baseline of natural or unimpacted state can be estimated from historic imagery. Alternatively, a contemporary baseline (i.e., a point in time that will be used to compare future monitoring data) can be established from the first set of reliable data. The risk with using a contemporary baseline is that significant degradation may have already occurred before its establishment, such that maintenance of current state may not be an appropriate management target.

Other approaches include comparison of calculated OMBT-EQRs to ratings for other natural or unimpacted estuaries (e.g., Freshwater Estuary, Stewart Island) or modelling approaches to predict OMBT-EQRs under different catchment nutrient inputs scenarios (e.g., natural land use; see Plew et al. 2020).

Measurement considerations: Detailed methods for monitoring macroalgae (i.e, OMBT-EQR) are outlined in the NEMP revision (MfE in prep 2024).

Macroalgal cover and biomass tend to increase in summer (e.g., Scanlan et al. 2007) so within-estuary measurements should be taken at the same time of each year (ideally around the peak of growth in summer, e.g., October - March) to limit the effect of seasonal changes on measurement results. Ideally, measurement frequency for each estuary, or representative estuaries in a region, would be determined by a risk assessment (i.e., higher frequency when there are known blooms). Where a problem is identified measurements should be repeated annually, or at least once every 3 years (WFD-UKTAG 2014). Where there are no obvious problems, measurements should be repeated every 5 years to track long-term trends.

Current New Zealand sampling methods, mapping accuracy, and classification criteria exhibit inconsistencies that the NEMP revision (MfE in prep 2024) will aim to address. Accuracy when applying a standard method is expected to be within 10% of the true value for repeat measurements conducted by the same provider. However, comparisons undertaken by Salt Ecology suggest results can be potentially highly variable between providers where there is a difference in provider experience, methods used, and ground-truthing effort undertaken.

Supporting field metadata requirements include date, time, tide height, GPS coordinates for point-based data (e.g., percent cover, biomass, entrainment), substrate type and substrate condition (i.e., aRPD). Other supporting indicators such as climate conditions, land use, nutrient inputs, and other broad- and fine-scale metrics are also useful.

Calculation of statistic: For an intertidal estuary, one-off mapping (undertaken late spring/summer) is sufficient to calculate the multi-metric OMBT-EQR for nuisance macroalgae. Repeat surveys are required to assess temporal and spatial changes.

The calculation of the OMBT-EQR is described in detail in WFD-UKTAG (2014) and Roberts et al. (2022b). Briefly, each metric in the OMBT has equal weighting and is combined to produce an Ecological Quality Rating (EQR). The measured metrics (Table A2-1 & A2-2) are normalised and re-scaled to an equidistant index score between 0 and 1 (see WFD-UKTAG 2014). The equidistant index score for each metric is averaged to establish the final OMBT-EQR score between 0 (major disturbance) to 1 (minimally disturbed). To adapt the WFD-UKTAG (2014) method to New Zealand estuaries, improvements have been made to biomass thresholds, as described in Plew et al. (2020; Table 2), and the method for calculating an EQR score when estuaries have ≤5% cover across the AIH (see Stevens et al. 2022).

Table A1-12. Description of the measured metrics and the individual calculation statistic (from WFD-UKTAG 2014). AIH = Available intertidal habitat and AA = Affected Area.

Measurement	Definition	Calculation statistic
% cover of AIH	The % cover is estimated as an average over the whole of the available intertidal habitat for the waterbody	% cover of macroalgae within AIH = Total area of algae (ha)/ AIH (ha) x 100 Where Total area of algae (ha) = Sum of [patch size (ha) x (average % cover for patch/100)]
Total affected area [AA] (hectares)*	The total extent of the bloom, measured in hectares and based on the external perimeter of the bloom	Affected Area, AA (ha) = Sum of all macroalgae patch areas.
AA/AIH (%)*	The affected area (ha) as a percentage of the total available intertidal habitat (ha)	AA/AIH (%) = AA (ha)/ AIH (ha) x 100
Biomass (g/m^{-2}) of Affected Area (AA)	This is the average biomass per square metre over the affected area only	Biomass of Affected Area (g/m^{-2}) = Total biomass (g/m^2) / [AA (ha)/10000^] Where Total biomass (g) = Sum of [{patch size (ha)/10000^} x average biomass for the patch (g/m^2)) ^*converted hectares to* m^2
Biomass (g/m^{-2}) of AIH	This is the average biomass per square metre over the whole of the available intertidal habitat	Biomass of AIH (g/m^{-2}) = Total biomass (g) / [AIH (ha)/10000^] Where Total biomass (g) = Sum of [{patch size (ha)/10000^} x average biomass for the patch (g/m^2)) ^*converted hectares to* m^2
Presence of entrained algae (%)	% patches where algae are growing in stable beds or with 'roots' deep (e.g., >30mm) within the sediments (i.e., more likely to regenerate a bloom)	Presence of Entrained Algae = (No. patches with entrained algae / total no. of patches) x 100

Table A1-13. Thresholds for measured OMBT metrics, including New Zealand revisions (Plew et al. 2020).

	High[1]	Good	Moderate	Poor	Bad
% cover on Available Intertidal Habitat (AIH)	0 - ≤5	>5 - ≤15	>15 -≤25	>25 - ≤75	>75 - 100
Affected Area (AA) [>5% macroalgae] (ha)[2]	≥0 - 10	≥10 - 50	≥50 - 100	≥100 - 250	≥250
AA/AIH (%)[2]	≥0 - 5	≥5 - 15	≥15 - 50	≥50 - 75	≥75 - 100
Average biomass (g/m^2) of AIH[3]	≥0 - 100	≥100 - 200	≥200 - 500	≥500 - 1450	≥1450
Average biomass (g/m^2) of AA[3]	≥0 - 100	≥100 - 200	≥200 - 500	≥500 - 1450	≥1450
% algae entrained	≥0 - 1	≥1 - 5	≥5 - 20	≥20 - 50	≥50 - 100

[1] Where ≤5% cover AIH, the EQR is calculated based on AA only as described in Stevens et al. (2022).
[2] Only the lower EQR of the 2 metrics, AA or AA/AIH, should be used in the final EQR calculation (WFD-UKTAG (2014).
[3] Updated biomass thresholds for New Zealand estuaries as described in Plew et al. (2020).

Potential bands and/or thresholds and rationale (including caveats):

<u>Thresholds:</u> The derivation of the thresholds presented in Table A1-2 are discussed in detail in Scanlan et al. (2007) and the updated biomass thresholds are discussed in Plew et al. (2020). Final OMBT-EQR thresholds presented in A1-3 are outlined in the WFD-UKTAG (2014).

Briefly, percent cover thresholds presented in the WFD-UKTAG (2014) were determined through expert opinion and supporting literature (see Scanlan et al. 2007 and references therein). Low levels of cover (i.e., <5% cover across the AIH) were considered 'reference' with progressively increasing cover scaled based on the impact within the affected areas. Plew et al. (2020) lowered the biomass thresholds for use in New Zealand estuaries based on unpublished data from >25 shallow well-flushed intertidal NZ estuaries (Robertson et al. 2016) and international studies (e.g., Green et al. 2014; McLaughlin et al. 2014; Sutula et al. 2014). These studies showed significant negative effects on biota at macroalgal biomass <1000g/m^2 (wet weight), which is below the original WFD-UKTAG (2014) thresholds.

The ETI condensed the WFD-UKTAG (2014) OMBT-EQR 5-band system to 4-bands by combining the poor/bad thresholds. We propose reinstating the 5-band system here (Table A2-3) as follows:

- 'Very good' represents no detectable change in macroalgal cover due to anthropogenic activities and is classified as an OMBT-EQR >0.8 in the WFD-UKTAG (2014).
- 'Good' represents a situation where macroalgae cover has slightly increased compared to natural conditions and there are no persistent growths or impacts to habitats (e.g., seagrass) or underlying sediments (WFD-UKTAG 2014).
- The mid-point ('fair') represents where an estuary is in a moderate state of health and there are increased cover and/or biomass of opportunistic species (WFD-UKTAG 2014). Some smothering of benthic habitats (e.g., seagrass) and degradation of underlying sediments are likely.
- The 'fair' to 'poor' threshold represents a threshold at with there is a high risk of reaching a tipping point and a permanent regime shift to a degraded state. This is supported by both the sub-metrics (A2-1) and monitoring data (e.g., Robertson et al. 2017; Roberts et al. 2022b; Roberts et al. 2023), where an OMBT-EQR below 0.4 represents a situation where there is persistent, high biomass blooms of nuisance macroalgae that are >20% entrained (WFD-UKTAG 2014).
- The 'poor' to 'very poor' threshold represents excessive algal growth and a likely regime shift to a persistent, degraded state in the affected area that is difficult to reverse.

Caveats: While it has been demonstrated that macroalgae abundance (measured as OMBT-EQR) is linked to catchment-derived nutrient inputs, management of such inputs may not lead to an immediate improvement in estuary condition, particularly where macroalgal blooms have become persistent and entrained. Studies have shown that macroalgae can utilise internal stores of nutrients to support growth (e.g., Dudley et al. 2022) and also source nutrients from enriched sediments (e.g., Robertson & Savage 2018), meaning growth can be sustained for extended periods in the absence of external inputs. We currently lack comprehensive understanding of legacy effects and hysteresis trajectories in estuaries. However, these should be considered, particularly when using indicators like macroalgae to track management outcomes.

Other factors influencing macroalgae expression include non-nutrient related changes e.g., macroalgal losses through flood scouring, channel flushing, wind-driven waves and temperature. Non-nutrient related macroalgal reductions can lead to a temporary improvement in the OMBT-EQR score, but where there is ongoing input of excess nutrients, nuisance macroalgal growth is likely to quickly re-establish to pre-disturbance levels. This has been observed in New River Estuary, Southland, where a large flood led to a temporary improvement in the OMBT-EQR score, with macroalgae re-establishing to pre-flood levels within 1-3 years (Roberts et al. 2022a).

A further situation that affects an OMBT-EQR score is when macroalgal die-back leads to extreme levels of enrichment, and sediment conditions become so poor that macroalgae can no longer survive (Stevens et al. 2022). In this situation a reduction in macroalgal cover and biomass improves the OMBT-EQR score, which incorrectly suggests an improvement in estuary condition. To our knowledge, this situation has only been observed on a large scale in two estuaries in New Zealand, Jacobs River Estuary and New River Estuary in Southland (Roberts et al. 2022a; Roberts et al. 2023). Despite these severe levels of degradation the OMBT-EQR score continued to detect 'poor' macroalgal state (see Fig. A1-1; Roberts et al. 2022a) Such situations highlight the importance of interpreting OMBT-EQR scores alongside other eutrophic indicators (e.g., sediment enrichment indicators such as TOC, TN, TS), field observations, and drivers of macroalgae response.

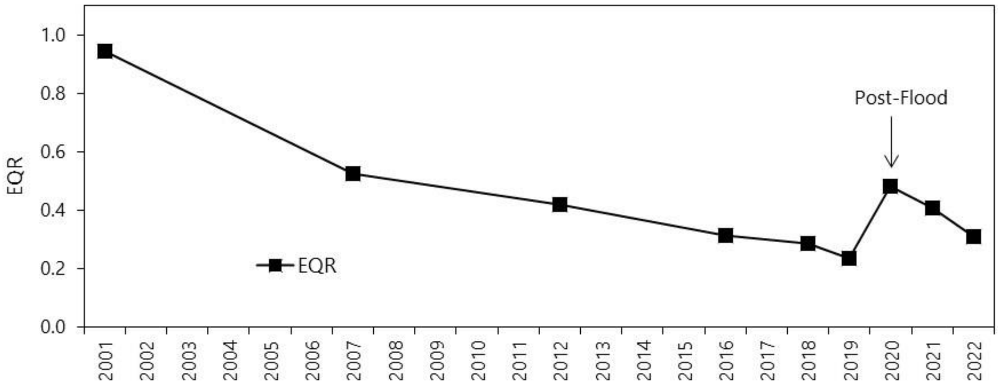

Fig. A1-1. OMBT-EQR in New River Estuary, Southland. Flood scouring led to a temporary improvement in OMBT-EQR score in 2020 and then macroalgae re-established in flood affected areas. Severe levels of degradation where macroalgae are no longer able to survive were first recorded in 2018.

Finally, it is important to recognise that the expression of nuisance macroalgae may be different in estuaries where there are other potential nutrient uptake pathways. For example, mangroves have high nutrient uptake rates that can potentially can mitigate coastal eutrophication (Gritcan 2018). Likewise salt marsh also provide nutrient buffering capacity (Nelson & Zavaleta 2012). Removal of these habitats may also lead to increased blooms where nutrients are released either through sediment disturbance or breakdown of vegetation (Lundquist et al. 2012; Augyte & Pickart 2014; Lundquist et al. 2014; 4Sight 2017). While such systems may have different nutrient thresholds at which nuisance macroalgae begin to establish (i.e., there may be a lag in macroalgal response), the multi-metric thresholds presented in A1-2 and Table A1-3 have been developed based on effects of macroalgae on biota and sediment quality (e.g., Hull 1987; Wither 2003; Scanlan et al. 2007; Green et al. 2014; Sutula et al. 2014; Plew et al. 2020), which are applicable to all intertidal estuaries.

Summary of proposed thresholds:

Table A1-14. Macroalgae (OMBT-EQR) thresholds for SIDEs and SSRTRE (with SIDE characteristics) estuary types.

OMBT-EQR	Ecological Quality Status				
	Very Good	Good	Fair	Poor*	Very Poor
	≥0.8 to 1.0	≥0.6 to <0.8	≥0.4 to <0.6	≥0.2 to <0.4	0.0 to <0.2
Narrative	Ecological communities are healthy and resilient.	Ecological communities are slightly impacted by additional macroalgae growth. No persistent impacts to habitats (e.g., seagrass) or underlying sediments.	Ecological communities are moderately impacted by macroalgae. Moderate cover across in the AIH and biomass (≥200-500g.m^{-2}), with areas of persistent growths becoming established. Some smothering of benthic habitats (e.g., seagrass) and degradation of underlying sediments likely.	Excessive algal growth. Ecological communities at high risk of undergoing a rapid regime shift to a persistent, degraded state. High cover, across the AIH, and high biomass (≥500-1450g.m^{-2}), with persistent growths established. Degradation to underlying sediments, strongly impacted ecological communities and loss of seagrass expected.	Excessive algal growth and a regime shift to a persistent, degraded state in the AA that is difficult to reverse. Very high cover, across in the AIH, and very high biomass (≥1450g.m^{-2}), with loss of benthic infauna, seagrass and degraded sediment condition (i.e., oxygen depletion).

*High risk of reaching a tipping point where wide-spread persistent, high cover, high biomass, entrained macroalgae are established and are likely difficult to reverse (i.e., regime shift).

Overall confidence in thresholds/ bands: High

Recommendation: Opportunistic macroalgal abundance (OMBT-EQR)

Adopt as preliminary numeric thresholds pending data analysis/review of New Zealand data.

Links to other indicators: Other indicators that serve as explanatory variables for changes in macroalgae (OMBT-EQR) include substrate type and quality (e.g., TOC, TN, TS, aRPD), water quality indicators (e.g., clarity, turbidity, salinity, nutrients) and climate variables (e.g., wind, temperature, etc.). Furthermore, complementary stressor indicators include nutrient and sediment loads, land use types and hydrodynamic characteristics such as flushing time, tidal exchange and dilution.

Alternative metrics considered: Taxonomic composition has been considered internationally and deemed inappropriate and impractical because presence alone does not imply deterioration, as nuisance species are a natural feature of estuaries, and estuaries are generally species poor due to fluctuating salinity, light availability and hydrodynamics (Scanlan et al. 2007; Wilkinson et al. 2007; WFD-UKTAG 2014).

Other macroalgae indicators that are more suitable at a site scale (note estuary-wide estimates can be obtained from multiple site-specific samples), include biochemical markers measured in macroalgae tissue such as chlorophyll, free amino acids, nitrogen (N), carbon (C), N:C and stable isotopes of $\delta^{15}N$ and $\delta^{13}C$ (Barr et al. 2020 and references therein). These biochemical markers can be used to assess eutrophication status, nitrogen storage capacity of macroalgae tissue and potential nitrogen sources. However, these indicators should be accompanied by other metrics that include spatial extent (e.g., OMBT-EQR), at a minimum.

Additional work recommended:

i. Collate standardised national data (and associated metadata) on macroalgae extent and OMBT-EQR scores. For example, Salt Ecology have OMBT-EQR data for ~50 estuaries (some over multiple years). It would be useful to combine this with other national datasets (e.g., Cawthron, NIWA and councils) in preparation for more comprehensive analyses.
ii. Additional data collection is required across a range of geographic regions (i.e., particularly estuaries with mangroves) alongside other supporting indicators and nutrient loads to improve local stressor-response relationships.
iii. Assess OMBT-EQR versus total nitrogen concentration (e.g., Plew et al. 2020) for estuaries containing mangroves to determine whether mangroves potentially buffer the effects of nutrients in estuaries and if simple predictive models can be used to predict estuary state (i.e., update Plew et al. 2018).
iv. Further research is required on the effects of macroalgal biomass on New Zealand macrofauna to validate current biomass thresholds. Further understanding is also needed on tipping points for macroalgae collapse at high levels of enrichment (i.e., where decomposition of high biomass blooms lead to severe eutrophic sediment conditions in which macroalgae are no longer able to survive) and responsiveness of the indicator to management interventions (i.e., whether legacy effects from persistent stable beds delay positive outcomes).
v. Explore whether remote sensing methods can be used to assist calculation of the OMBT-EQR, e.g., by improving percent cover estimates, remotely assessing biomass and entrainment, and reducing ground-truthing requirements.
vi. Analyse within and between provider accuracy in mapping of percent cover and measures/estimates of biomass and entrainment.
vii. Explore active management methods that include macroalgal removal, particularly in situations where issues are localised and not yet persistent and self-sustaining.

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APPENDIX 2. MANGROVES

Only one species of mangrove, the grey mangrove or manawa (*Avicennia marina* subsp. *australasica*) is found in New Zealand (Morrisey et al. 2010). Mangroves are present only on the North Island, with distributions extending southward to Kawhia Harbour on the west coast, and to Ōhiwa Harbour on the east coast. Mangroves are typically (but not exclusively) found in sheltered, intertidal areas in estuaries. Their lower elevation limit is roughly at mean sea level (MSL), such that mangroves can be submerged for no more than ~6h per tidal cycle. Mangrove forests provide a diversity of physical and ecological functions in New Zealand estuaries, contributing organic matter to food webs, providing habitat structure, mitigating against climate change through carbon sequestration, and serving as a natural defence against coastal hazards (Morrisey et al. 2010; Lundquist et al. 2017; Bulmer et al. 2020; Gijsman et al. 2021).

BACKGROUND

In general, mangrove forests have expanded in New Zealand's northern estuaries from the mid-1800s onwards. This has largely occurred due to habitat expansion and vertical accretion of (unvegetated) intertidal flat habitat, suitable for mangrove colonisation (Suyadi et al. 2019). This accelerated estuary infilling process is driven by catchment sediment loading associated with large-scale catchment deforestation and conversion to pastoral agriculture and more recent land-use intensification (e.g., production forestry) (Morrisey et al. 2010; Horstman et al. 2018; Swales et al. 2020). Their national distribution is controlled by low winter air temperatures, frost frequency, biogeography and oceanography that limit mangrove propagule dispersal, and the limited availability and relative remoteness of suitable estuarine environments south of their present range (de Lange & de Lange 1994; Osland et al. 2017).

Climate warming and associated sea level rise will likely drive further changes in mangrove extent, with landward/upslope migration to maintain their position above MSL being the primary mechanism for shoreward changes in distribution, accompanied by reductions in mangrove suitability at lower tidal heights (McBride et al. 2016; Suyadi et al. 2019). The relatively wide range of elevation (i.e., MSL to Highest Astronomical Tide [HAT]) that mangroves occupy, relatively large tidal ranges, as well as generally fine sediment-rich estuarine systems, mean that New Zealand mangrove forests are unlikely to be lost to inundation during this century (Lovelock et al. 2015). In some locations, mangrove habitat is expanding into salt marsh habitat; and mangroves are likely to displace salt marsh without management interventions, as has been observed elsewhere globally (Doughty et al. 2015; Whitt et al. 2020). For mangroves and salt marsh, interventions that remove built physical barriers to migration, as lowland areas are inundated, will mitigate the likelihood of mangrove and salt marsh habitat loss and generate opportunities for restoration of freshwater–estuarine wetland ecosystems (Stewart-Sinclair et al. 2024). Although climate warming may influence landward migration (facilitated by sea level rise), biogeographic limitations to southward range expansion remain (de Lange & de Lange 1994; Saintilan et al. 2014).

New Zealand mangroves are also deliberately removed, including to support infrastructure (i.e., access, transit, power lines etc.), and over larger extents (i.e., 10s of ha) to support local values of recreation, accessibility and viewscape. Large-scale removals have declined significantly since the 2010s when this was common (e.g., Tauranga, Whangamata, Tairua) and following a review documenting degraded state and limited recovery subsequent to removal (Lundquist et al. 2012; Lundquist et al. 2014; Bulmer et al. 2017; Lundquist et al. 2017).

Monitoring coarse scale changes in mangrove habitat (i.e., areal extent) is readily amenable to remote sensing (Suyadi et al. 2018a, 2019; Bulmer et al. 2024). Remote sensing may not, however, capture incremental changes, low density mangrove stands on forest fringes, nor adequately capture the recent movement of mangrove into saltmarsh habitat. Much of the information on mangrove habitat quality, associated with ecological and environmental characteristics (e.g., macrofaunal communities, sedimentation rates, sediment properties etc.) requires field surveys (Swales et al. 2011).

PROPOSED METRICS

The following indicator metrics are proposed for monitoring mangrove forest extent and quality.

1. Change in areal extent (ha) of mangrove forest from baseline.
2. Change in areal extent (ha) of mangrove forest covered by tall and dwarf mangroves from baseline.

2.1 MANGROVE FOREST EXTENT

Indicator type: Supporting.

Metric: Change in areal extent (ha) of mangrove forest from baseline.

Unit of measurement: Hectares (ha).

Spatial scale: National, regional and estuary-wide.

Applicability: Northern New Zealand estuarine and coastal waters where suitable habitats for mangrove forests exist, within the mangrove latitudinal range.

Rationale: Areal extent is an efficient, cost-effective and sensitive indicator of spatial and temporal changes in mangrove forest extent. Timeframes of 10-year intervals are likely suitable to quantify changes in broad-scale extent, with shorter timeframes for areas of active expansion. Aerial photographic surveys since the late 1930s indicate the rate of mangrove habitat expansion in New Zealand's northern estuaries has averaged over 3% yr^{-1} (range 0.2–20.2% yr^{-1}) (Morrisey et al. 2010; Suyadi et al. 2019). With the exception of deliberate removal, there are few observations of natural declines in mangrove forests in New Zealand, for example in one Auckland estuary due to changes in hydrodynamic conditions, e.g., Lundquist et al. (2014).

Method: Areal extent measurements can be acquired from broad scale maps using established NEMP methods (e.g., Robertson et al. 2002, Stevens et al., in prep), with up-to date aerial or satellite imagery used to record mangrove features. The ability to detect change increases with increasing measurement frequency, spatial resolution and accuracy.

Quantitative techniques for extracting hyperspectral signatures indicating presence of mangroves from satellite remote sensing have been developed for New Zealand mangroves based on current satellite (Sentinel-2 images) technology (Bulmer et al. 2024). Sentinel-2 satellite images are open source, multispectral (13 bands), have a resolution of 10m, with a 5-day revisit time, and have been providing images since 2015. In combination with regional 1m resolution LiDAR Digital Elevation Model (DEM) data and routine machine learning algorithms, Sentinel-2 images have high mapping accuracy (Kappa accuracy score >0.9) (Bulmer et al. 2024). Aerial photographs and multi-spectral imagery (i.e., Landsat images) have also been used to provide comparable historical analyses (Swales et al. 2009; Suyadi et al. 2018a).

Assessment baseline: The most ecologically relevant baseline is temporal change in areal extent of mangroves compared to the most recent sampling interval. Comparison to a natural reference state may also be useful to reflect long-term change. However, a natural reference state is seldom able to be directly measured due to historical estuary modification. A baseline state may also be defined from historic imagery or the first set of reliable contemporary measurements, noting that these may not reflect maximum potential physical extent due to historical changes, e.g., estuary reclamation. Where contemporary data are used to define the current state, it is important that they reflect representative conditions (e.g., does not reflect an episodic impact from a recent storm).

Measurement Considerations: The metric is well suited for broad-scale estimates of mangrove extent, using NEMP broad scale mapping methods or remote sensing to delineate boundaries of mangrove forests, noting that the latter is least accurate in areas of active expansion that may be associated with patchy, fragmented or sparse mangroves, or are dominated by seedlings with limited canopy width. Ground-truthing is also required for capturing fine-scale incremental changes, particularly for infilling of low density mangrove stands, or for shoreward or seaward expansion of forest fringes, or to capture expansion of mangroves into saltmarsh habitat (Suyadi et al. 2019). Field surveys can also capture recruitment events, noting that seedling mortality is highly variable, with rates of 30-50% at sheltered tidal creek locations (Lundquist et al. 2017), to >99% at highly exposed sites such as the Firth of Thames (Swales et al. 2015). Thus, presence of seedlings should be excluded from use in the extent metric. While no specific threshold to define tree density is recommended, typically mangrove boundaries should be delineated by presence of mature trees. As mature trees range in height, canopy width, and density, seedlings are typically defined as those below 1m, though for dwarf mangrove stands, a lower threshold of 0.5m may be used to ensure mature trees of smaller stature are included.

Field surveys are best carried out in late winter or early spring, after natural seedling mortality events due to frost or due to winter storms, to minimise influence of seedling recruitment on estimates of forest extent. Methods for fine-scale monitoring of boundaries and expansion/contraction of individual mangrove forests using hand-held GPS are described in Swales et al. (2011). Most mangrove forests exhibit variable patterns of expansion, ranging from gap filling in sparse mangrove stands, seaward, landward or upstream expansion, or transition from tall to dwarf mangrove forests (Suyadi et al. 2019). Some locations have exhibited episodic recruitment events (e.g., Firth of Thames - Balke et al. 2014), whereas others have shown stable patterns with no expansion over decades (Horstman et al. 2018).

Ideally, monitoring frequency for broad scale measurements of mangrove forest extent would be decadal, with ground-truthing of a smaller number of representative estuaries. Higher frequency measurements might be suitable (i.e., 2-yearly) for estuaries where rapid change in mangrove extent might be anticipated due to high levels of sediment erosion in neighbouring catchments (i.e., locations with extensive road works or urban development), or where mangrove expansion into saltmarsh habitats is occurring.

Calculation of statistic: Percent change in areal extent (ha) of mangrove forest from baseline:

$$\text{Percent change in areal extent of mangrove forest} = \frac{\text{(Current measured areal extent in ha – Baseline areal extent in ha)}}{\text{(Baseline areal extent in ha)}} \times 100$$

Metrics can be calculated at a range of scales, including change relative to national mangrove extent, regional mangrove extent, and individual estuary mangrove extent.

Potential bands and/or thresholds and rationale (including caveats): Increasing mangrove extent is usually indicative of sediment erosion in the neighbouring catchment, and thus estuarine habitat degradation. However, future climate change may result in decreases in mangrove extent, due to the presence of barriers to shoreward expansion. Furthermore, consented and illegal clearing of mangroves can result in decreases in mangrove extent. Significant increase or decrease in mangrove extent can result in adverse effects on ecological health, reflected in the proposed narrative thresholds in Table A2-1. There is a lack of information to inform the development of numeric thresholds.

Summary of proposed thresholds:

Table A2-1. Recommended narrative bands for rate of change in mangrove areal extent (at an individual estuary scale) in New Zealand estuaries.

Change in spatial extent from baseline*	Ecological Quality Status				
	Very Good	Good	Fair	Poor	Very Poor
Narrative	Mangrove forest extent reflects baseline state*, and is stable	Slight increase or decrease in mangrove extent due to anthropogenic pressures	Moderate increase or decrease in mangrove extent due to anthropogenic pressures	Significant increase or decrease in mangrove extent due to anthropogenic pressures	Very large increase or decrease in mangrove extent due to anthropogenic pressures

* A baseline state may be defined from historic imagery or the first set of reliable contemporary measurements, noting that these may not reflect maximum potential physical extent due to historical changes, e.g. estuary reclamation. Where contemporary data are used to define the baseline it is important that they reflect representative conditions.

Overall confidence in thresholds/ bands: Fair.

Recommendation: Potentially worthwhile for further investigation but substantial further development or data required.

Links to other indicators: Indicators that serve as explanatory variables for changes in mangrove extent include catchment sediment loads, sedimentation rate, and climate variables (e.g., wind, temperature etc.). Salt marsh extent is also linked, with high likelihood of mangrove expansion into shoreward salt marsh with sea level rise.

Alternative metrics considered: No alternative metrics are suggested.

Additional work recommended:

i. Standardise methodology for broad-scale mangrove habitat mapping using satellite imagery.
ii. Collate standardised national data (and associated metadata) on mangrove extent and explore relationships between changes in mangrove extent and ecological health to refine narrative thresholds, and to develop numeric thresholds.
iii. Provide guidance on how to consistently define reference/baseline conditions.
iv. Explore additional spatial metrics to assess condition via satellite remote sensing, for example global mangrove forest patch characteristics (Hai et al. 2022) that could be applied in Aotearoa.
v. Quantify correlation between sediment erosion rates and annual rates of mangrove expansion to inform the development of numeric thresholds.

2.2 CHANGE IN AREAL EXTENT OF MANGROVE FOREST COVERED BY TALL AND DWARF MANGROVES (AS INDICATOR OF MANGROVE QUALITY)

Indicator type: Supporting.

Metric: Change in areal extent (ha) of mangrove forest covered by tall and dwarf mangroves from baseline. Mature 'tall' mangroves range in height from 1m to >6m, where dwarf mangroves are stunted growth morphologies, and are typically <1 m in height at maturity.

Unit of measurement: Percent (%) change in areal extent from baseline.

Spatial scale: National, regional and estuary-wide.

Applicability: Northern New Zealand estuarine and coastal waters where suitable habitats for mangrove forests exist, within the mangrove latitudinal range.

Rationale: Mangrove height, as well as characteristics of patches (e.g., width), have a strong influence on ecosystem services provided, with taller mangroves associated with provision of more ecosystem services than dwarf mangroves (Horstman et al. 2014; Suyadi et al. 2018b). Mangrove height is typically associated with hydrodynamic characteristics, and sediment infilling and changes in hydrodynamics has in some cases resulted in replacement of tall mangroves by dwarf mangrove stands (Suyadi et al. 2019).

Aerial photographic surveys since the late 1930s indicate the rate of mangrove habitat expansion in New Zealand's northern estuaries has averaged over 3% yr^{-1} (range 0.2–20.2% yr^{-1}), with larger rates of increase in dwarf mangroves than of tall mangrove (Morrisey et al. 2010; Suyadi et al. 2019). Timeframes of 10-year intervals are likely suitable to quantify changes in broad-scale extent, with shorter timeframes for areas of active expansion.

Method: Areal extent measurements can be acquired from broad scale maps using established National Estuary Monitoring Protocol (NEMP) methods (e.g., Robertson et al. 2002, Stevens et al., in prep), with up-to date aerial or satellite imagery used to record mangrove features. The ability to detect change increases with increasing measurement frequency, spatial resolution and accuracy.

Quantitative techniques for extracting hyperspectral signatures indicating presence of mangroves from satellite remote sensing have been developed for New Zealand mangroves based on current satellite (Sentinel-2 images) technology (Bulmer et al. 2024). Sentinel-2 satellite images are open source, multispectral (13 bands), have a resolution of 10m, with a 5-day revisit time, and have been providing images since 2015. In combination with regional 1m resolution LiDAR Digital Elevation Model (DEM) data and routine machine learning algorithms, Sentinel-2 images have high mapping accuracy (Kappa accuracy score >0.9) (Bulmer et al. 2024). Aerial photographs and multi-spectral imagery (i.e., Landsat images) have also been used to provide comparable historical analyses (Swales et al. 2009; Suyadi et al. 2018a), including mangrove height for more recent photographs (Swales et al. 2009; Suyadi et al. 2018a). Ground-truthing may be required to confirm mangrove stature for current satellite technology.

Assessment baseline: The most ecologically relevant baseline is temporal change in the proportion of tall vs dwarf mangroves compared to the most recently sampled baseline. Historical temporal change from natural reference state is also relevant to confirm long-term historical change in mangrove stature and quality from reference states. Reference states are seldom able to be directly measured due to historical estuary modification, and due to difficulties in distinguishing mangrove stature from historical photographs prior to ~1990. A baseline state may also be defined from historic imagery or the first set of reliable contemporary measurements, noting that these may not reflect maximum potential physical extent due to historical changes, e.g. estuary reclamation. If contemporary data are used to define the baseline it is important that they reflect representative conditions (e.g., does not reflect an episodic impact from a recent storm).

Measurement Considerations: The metric is well suited for broad-scale estimates of changes in quality of mangrove forests as measured by reductions in the proportion of tall mangroves. NEMP broad scale mapping methods or remote sensing can be used to delineate boundaries of mangrove forests, noting that the latter is least accurate in areas of active expansion that may be associated with patchy, fragmented or sparse mangroves, or are dominated by seedlings with limited canopy width. Ground-truthing is often required to identify the size structure of mangrove forests (e.g., tall, dwarf forms), which is useful when converting density-estimates to ecosystem services provided by mangrove forests (i.e., carbon sequestration, coastal hazard mitigation).

Ideally, monitoring frequency for broad scale changes in stature of mangrove forest extent would be decadal, with ground-truthing of a smaller number of representative estuaries. Higher frequency measurements might be suitable (i.e., 2-yearly) for estuaries where rapid change in mangrove stature might be anticipated due to high levels of sediment erosion in neighbouring catchments (i.e., locations with extensive road works or urban development), or where mangrove expansion into saltmarsh habitats is occurring.

Calculation of statistic: Percent change in areal extent (ha) in tall (or dwarf) mangroves from baseline:

Percent change in proportion of total area extent consisting of tall mangrove stature = current proportion – baseline proportion.

$$\text{Current proportion of areal extent of tall mangrove forest} = \frac{(\text{Current tall mangrove measured areal extent in ha})}{(\text{Total mangrove areal extent in ha of tall and dwarf mangroves})} \times 100$$

Metrics can be calculated at a range of scales, including change in mangrove stature at national, regional, and individual scales.

Potential bands and/or thresholds and rationale (including caveats): There are no known thresholds or numeric bands for New Zealand mangrove forest quality, and reductions in the proportion of tall compared to dwarf mangrove stands are considered as a correlative indicator of mangrove quality. Mangrove density, height, and characteristics of patches (e.g., width) have a strong influence on ecosystem services provided (Horstman et al. 2014; Suyadi et al. 2018b). Percent cover is not suitable indicator of mangrove forest quality or thresholds in response to disturbance, as percent cover by mangrove habitat at an estuary extent varies based on estuary age and state of infilling, as well as drivers linked to climate change and catchment sediment erosion (Suyadi et al. 2019). Mangrove height also varies substantially across its New Zealand range. While typically taller mangroves are found at northern latitudes, dwarf morphologies are also found in Northland, and tall mangrove stands (e.g., >4m) are also common in the Auckland and Waikato regions (Bulmer et al. 2016; Suyadi et al. 2019). Substantial variability in morphology can be found within estuaries, and within small embayments based on distance to shore/tidal creek (Horstman et al. 2018; Suyadi et al. 2019).

Historical evidence to inform 'baseline' mangrove distributions varies based on availability of aerial images in the 1930s and beyond. Historical changes prior to the 1930s (both losses and gains) as well as changes in stature of New Zealand's mangrove forests have not been accurately quantified because substantial habitat change occurred prior to systematic aerial photographic surveys.

Summary of proposed thresholds: No thresholds have been proposed.

Overall confidence in thresholds/ bands: Undeveloped.

Recommendation: Not endorsed.

Links to other indicators: Other indicators that serves as explanatory variables for changes in mangrove quality include catchment sediment loads, sedimentation rate, and climate variables (e.g., wind, temperature etc.). Saltmarsh extent is also linked, with high likelihood of mangrove expansion into shoreward saltmarsh with sea level rise.

Alternative metrics considered: No alternative metrics are suggested.

Additional work recommended:

i. Standardise methodology for broad-scale mangrove habitat mapping, including using satellite imagery to delineate stature.
ii. Collate standardised national data (and associated metadata) on mangrove extent and stature.
iii. Provide guidance on how to consistently define reference/baseline conditions.

iv. Explore additional spatial metrics to assess condition via satellite remote sensing, for example global mangrove forest patch characteristics (Hai et al. 2022) that could be applied in Aotearoa.
v. Explore applicability of comprehensive indices of mangrove forest quality, such as those developed globally, including ecological and environmental characteristics (e.g., macrofaunal communities, turbidity), as well as social attributes (Ibrahim et al. 2019), reflecting mangrove use and economic value.

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APPENDIX 3. MUD-ELEVATED (25% MUD CONTENT) SEDIMENT

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The combination of accelerated sediment accretion rates and increased sediment mud content has long been recognised as a major stressor on estuaries and other coastal ecosystems, and which can significantly alter the hydrodynamic, geomorphology, and ecological characteristics of the receiving system impacting its ecological health. Of particular concern are the accumulation of silt and clay particle size fractions <63µm (collectively termed 'mud') which 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), and which can result in widespread mud deposition zones developing in upper estuary tidal flats (Robertson et al. 2016b). The following background text is primarily drawn from Zaiko et al. (2018) and MfE (2022).

BACKGROUND

The gradual infilling of estuaries with sediment eroded from land is a natural process, but sediment accumulation rates have increased by orders of magnitude since European settlement in many places (e.g., Handley et al. 2017, Hunt 2019a, Ministry for the Environment and Stats NZ 2019, Parliamentary Commissioner for the Environment 2020, Ministry for the Environment 2022). This was initially caused by widespread catchment deforestation, much of this during the mid-1800s to early-1900s, but some current human activities and land use practices can also increase rates of soil erosion and resulting sediment loads delivered to waterways and estuaries. Some activities within estuaries, such as aquaculture, channel dredging, and structures such as causeways can also affect sediment mud content (an important measure of sediment quality) and sediment accretion rates. Further, legacy effects of previous land use decisions mean sediment impacts may be borne for decades or centuries after management changes are made, with some effects nearly impossible to reverse (Townsend and Lohrer, 2015).

Excessive fine (silt and clay) sediment inputs can affect biodiversity by altering habitats, smothering benthic species, and suppressing important biological and biogeochemical processes such as feeding, respiration, photosynthesis, reproduction, recruitment, and denitrification (e.g., Kennish 2002, Thrush et al. 2004, Lohrer et al. 2003, O'Meara et al 2017, Thrush et al. 2021). Impacts also include the loss or degradation of shellfish beds (e.g., Thrush et al 2013), and altered microbial activities (which are critical for organic matter degradation and nutrient regeneration), diminished benthic primary productivity, and reduced oxygenation of surficial sediments by capping the seabed, clogging sediment pore spaces, and depriving micro- and macrophytes of light (Berkenbusch et al. 2002). Townsend and Lohrer (2015) report consequences from short-term deposition 'events' (primarily burial) that include lethal effects on benthic biota, changes in benthic species composition, loss of sensitive species, decline in diversity, and modification of animal behaviours (Hewitt et al. 2003, Thrush et al. 2004, Lohrer et al. 2004, Norkko et al. 2002a). Previous work on ecological breakpoints (e.g., Berthelsen et al. 2018, Robertson et al. 2015, and references therein), and subsequent analysis of Salt Ecology national data from 766 stations across 36 estuaries with paired macroinvertebrate and sediment mud data indicate the most diverse and abundant macrobenthic communities occur in sediments with mud concentrations of less than ~20-25%. Therefore, the revised NEMP (Stevens et al. in prep), defines sediments with >25% mud as 'mud-elevated' and indicative of likely ecological degradation.

NEMP monitoring data show many New Zealand estuaries have large areas of mud-elevated (>25% mud) sediments, and often elevated (relative to pre-catchment deforestation baseline) sedimentation rates. For these reasons, sediment mud content is considered a key attribute for management and a useful supporting indicator for assessing estuary trophic status.

PROPOSED METRICS

The following indicator metrics are proposed for monitoring mud-elevated (>25% mud) sediment extent:

1. Extent of AIH (excluding salt marsh) with mud-elevated (>25% mud) sediment.
2. Change in extent of intertidal mud-elevated (>25% mud) sediment from the first measured baseline.

These should be used as part of a multi-faceted approach that includes assessment of sedimentation rate, and predicted sediment loads to the estuary, and sediment mud content (see Appendices 7 and 10).

3.1 EXTENT OF INTERTIDAL AREA WITH MUD-ELEVATED (25% MUD CONTENT) SEDIMENT

Indicator type: Supporting.

Metric: Extent of AIH (excluding salt marsh, and mangroves where applicable) with mud-elevated (25% mud content) sediment.

Unit of measurement: Percent (%) of available intertidal habitat (AIH) excluding salt marsh (and mangroves where applicable).

Spatial scale: Estuary-wide within AIH excluding salt marsh (and mangroves where applicable).

Applicability: Intertidally dominated estuaries (e.g., SIDE or SSRTREs with SIDE characteristics).

Rationale: Areal extent is an efficient and cost-effective indicator of estuary condition and is sensitive enough to detect broad spatial and temporal changes in surface substrate type when measurements are repeated over time, and standard measures of spatial distribution of 'mud-elevated' sediment have been established under the NEMP. However, although there is a strong relationship between increasing sediment mud content and persistent ecological degradation (e.g., to macrofauna), the relationship between the spatial extent of 'mud-elevated' sediment and overall biological impacts is still being established for New Zealand estuaries (Robertson et al. 2016). Notwithstanding, because of the adverse effects associated with fine sediment described above, and historically elevated inputs of fine sediment to many estuaries in New Zealand, a reasonable bottom-line management target is that mud-elevated substrate should not increase substantially from its current extent.

Method: Briefly, the NEMP (Robertson et al. 2002) broad-scale mapping approach, including refinements by Salt Ecology (Stevens et al. 2023), is used to map the spatial extent of defined substrate features. The Salt Ecology revisions designate five fine unconsolidated substrate classes based on sediment mud content (S=Sand: 0-10%; MS=Muddy Sand (moderate mud): ≥10-25%; MS=Muddy Sand (high mud): ≥25-50%; SM=Sandy Mud: ≥50-90%; M=Mud: ≥90%), the latter three classes being used to define 'mud-elevated' (>25% mud) sediments. These classes reflect categories that can be subjectively assessed in the field by experienced practitioners at a coarse level, and validated by the laboratory analysis of particle grain size samples (wet sieving) collected from representative sites. Extensive mapping experience has shown that transitional boundaries between unconsolidated substrate classes can be mapped to within ±20-50m where they have been thoroughly ground-truthed and validated with particle grain size samples.

Areal extent measurements are typically acquired from broad scale maps using established NEMP methods, with up-to date aerial or satellite imagery used to record substrate features. The ability to detect change increases with increasing measurement frequency, spatial resolution and accuracy. Ground-truthing maps are ideally <50cm/per pixel resolution at a scale of between 1:2000 and 1:5000, as at a coarser pixel resolution and scale it becomes difficult to reliably characterise features. Annotated field maps of validated features, combined with field notes and georeferenced photographs, are digitised into shapefiles to produce maps of substrate extent. Machine learning with automated digitisation based on the spectral analysis of imagery remains in development in New Zealand (e.g., Ha et al. 2020), but is not expected to be suitable for determining 'mud-elevated' (25% mud content) sediment extent.

Depending on the extent of ground-truthing and the QA/QC methods (if any) used in initial NEMP surveys, it may be necessary to update data following data QA/QC checks, e.g., to remove any overlapping or duplicated polygons, or exclude terrestrial features. Reclassification of initial NEMP substrate features into defined mud content classes may also be necessary.

Assessment baseline: The first reliable contemporary measurement of mud-elevated sediment extent in the AIH, excluding salt marsh habitat which is well known as an effective trap of estuary sediment. A baseline would ideally be measured over ~3 consecutive years to quantify likely natural variability in extent.

Measurement considerations: Estuary-wide substrate classification using NEMP visual assessment methods is considered a coarse initial screening approach for determining the potential scale of contemporary sediment issues. Where improved accuracy in the definition of substrate boundaries is required, additional sampling approaches are recommended such as the use of targeted grain size analyses along fixed transects, or through stratified, random or grid sampling approaches, often applied at a sub-estuary scale. While it is possible to extend sampling into sub-tidal areas, e.g., through wading, SCUBA surveys or remote grab sampling, the NEMP methods are designed primarily for use in intertidal areas.

Contemporary mud-elevated extent generally overlies and obscures historical extent. Historical extent can be assessed by analysis of subsurface features, ranging from simple approaches such as digging holes to reveal underlying sediment layers, to more complex sediment coring methods involving analysis of grain size accompanied by carbon, radioisotope or pollen analyses to date sediments and determine deposition rates (see Appendix 7 for further detail). Such approaches are very useful for improving the understanding of contemporary state.

There are inconsistences in current New Zealand sampling methods, mapping accuracy, and classification criteria. Method accuracy is expected to be ~±10% of the true value for repeat measurements conducted by the same provider, but potentially highly variable between providers.

Statistic calculated from mud-elevated areal extent: Extent (ha) of intertidal mud-elevated (>25% mud content) sediment, excluding salt marsh habitat. Reported as percent of intertidal area (excluding salt marsh):

$$\text{Intertidal mud-elevated extent} = \frac{\text{Current measured mud-elevated extent (ha)}}{\text{Intertidal area excluding salt marsh (ha)}} \times 100$$

Potential bands and/or thresholds and rationale (including caveats): The New Zealand Estuary Trophic Index (ETI) Toolbox project (Robertson et al. 2016) proposed preliminary thresholds on the basis that impacts from elevated mud contents are well described (see above and also Appendix 11), and that the larger the spatial extent of mud-elevated sediment, the larger the likely ecological damage. Expert opinion, and a cursory review of New Zealand NEMP data, was used to defined four bands of potential impact for the ETI (Bands A-D, Table A3-1), noting further work was required in order to determine an overall estuary rating for mud-elevated sediment. Because intertidal areas in the mid and low tide range near the estuary entrance are commonly subjected to higher tidal flushing and wave action than upper tidal ranges, accumulation of mud-elevated sediment is often most apparent among salt marsh and on upper intertidal flats away from the entrance. On the basis that the upper, mid and low tidal zones each comprise ~1/3rd of the intertidal area, mud-elevated sediments over 50% of the upper zone (~15% of the intertidal area) are considered to likely reflect 'Poor' ecological state. Salt marsh habitat is excluded from this metric due to its effective trapping of fine sediment.

There appears to have been little or no further consideration of the proposed thresholds since development of the ETI, although it is noted that substantial additional NEMP data on the extent of mud-elevated sediment relative to intertidal extent are now available. Consequently, it is proposed that the ETI thresholds be retained for guidance only pending review of New Zealand data to determine their appropriateness. Expert judgement has been used to propose ≥25% as a 'Very poor' threshold on the basis that if 25% of the intertidal zone outside of salt marsh comprised mud-elevated sediments, it would reflect widespread degradation of ecological state.

It is emphasised that some estuaries will have naturally elevated sediment inputs that result in large naturally occurring areas of mud-elevated sediment. Although these will be rated as degraded by the proposed thresholds, the ratings should be used in a precautionary manner to prompt further investigation to determine the likely change from either natural or historical state when degraded conditions are identified. Where uncertainty exists about the historical extent of mud-elevated sediments within an estuary, coring to reveal underlying sediment type is recommended to validate assumptions.

Summary of proposed thresholds:

Table A3-1: Recommended thresholds for the percent of intertidal area (excluding salt marsh, and mangroves where applicable) with mud-elevated (>25% mud content) sediment.

Percent of AIH with mud-elevated sediment*	Ecological Quality Status				
	Very Good (A)	Good (B)	Fair (C)	Poor (D)	Very Poor
	<1%	1 to <5%	≥5 to <15%	≥15% to <25%	≥25%
Narrative	Localised impacts likely reflective of natural state in many estuaries. Negligible impacts on estuary ecological function.	Increasing extent, likely in discrete parts of upper estuary areas. Relatively small impacts on total estuary ecological function.	Increasing extent, likely over connected parts of upper estuary areas. Moderate impacts on total estuary ecological function.	Widespread extent likely across >50% of upper estuary areas. Large impacts on total estuary ecological function.	Widespread extent likely across much of the upper estuary and extending into other areas. Large impacts on total estuary ecological function.

*Assessment needs to consider inputs from natural processes, e.g. shoreline erosion, noting anthropogenic climate change may increase impacts.

Overall confidence in thresholds/ bands: Fair. Thresholds are based on expert judgement, but require refinement based on an assessment of existing New Zealand data.

Recommendation: Percent of AIH with mud-elevated (25% mud content) sediment

Adopt Table A3-1 as guidance only thresholds pending analysis/review of New Zealand data.

Links to other indicators: Other indicators that serve as explanatory variables for changes in mud-elevated substrate extent include catchment sediment loads, sedimentation rate, mud content, water quality indicators (e.g., clarity, turbidity) and climate variables (e.g., wind, temperature, rainfall etc.). The impact of anthropogenic stressors on ecosystems may be highly context specific (i.e., place and history are very important), compounded by high variability in pressures such as physical damage, changes in sea level, severe storm frequency and intensity, brought about by climate change.

Alternative metrics considered: Other than change from a recent baseline (see following) no alternative metrics considered for spatial extent.

Additional work recommended:

i. Standardise sampling methods and reporting metrics, based on the current NEMP revision (Stevens et al. in prep).
ii. Collate standardised national data (and associated metadata) on measured mud-elevated extent.
iii. Test the classification accuracy of subjective assessments of substrate mud content using existing validation data.
iv. Analyse within and between provider mapping accuracy and assess confidence intervals on the assessment of temporal and spatial change.
v. Revise interim thresholds based on iii. and iv. to refine percent loss breakpoints.
vi. Undertake further studies to determine the potential historical mud-elevated extent of New Zealand estuaries.
vii. Investigate development of supporting thresholds based on change measured in hectares.

3.2 CHANGE IN INTERTIDAL MUD-ELEVATED (>25% MUD) SEDIMENT EXTENT FROM THE FIRST ACCURATE BASELINE

Indicator type: Primary.

Metric: Change in intertidal mud-elevated (25% mud content) sediment extent relative to the first accurate baseline.

Unit of measurement: Percent (%) change in areal extent from baseline.

Spatial scale: Estuary-wide.

Applicability: Intertidally dominated estuaries (e.g., SIDE or SSRTREs with SIDE characteristics).

Rationale: Areal extent is an efficient and cost-effective indicator of estuary condition and is sensitive enough to detect broad spatial and temporal changes in surface substrate type when measurements are repeated over time, and standard measures of spatial distribution of mud-elevated sediment have been established under the NEMP. This metric can also be applied at a sub-estuary scale in larger systems. However, although there is a strong relationship between increasing sediment mud content and persistent ecological degradation (e.g., to macrofauna), the relationship between the spatial extent of mud-elevated sediment and overall biological impacts is still being established for New Zealand estuaries (Robertson et al. 2016). Notwithstanding, because of the adverse effects associated with fine sediment described above, and historically elevated inputs of fine sediment to many estuaries in New Zealand, a bottom-line management target is that mud-elevated substrate should not increase substantially from its current extent.

Method: Briefly, the NEMP (Robertson et al. 2002) broad-scale mapping approach, including refinements by Salt Ecology (Stevens et al. 2023), is used to map the spatial extent of defined substrate features. The Salt Ecology revisions designate five fine unconsolidated substrate classes based on sediment mud content (S=Sand: 0-10%; MS=Muddy Sand (moderate mud): ≥10-25%; MS=Muddy Sand (high mud): ≥25-50%; SM=Sandy Mud: ≥50-90%; M=Mud: ≥90%), the latter three classes being used to define 'mud-elevated' (>25% mud) sediments. These classes reflect categories that can be subjectively assessed in the field by experienced practitioners at a coarse level, and validated by the laboratory analysis of particle grain size samples (wet sieving) collected from representative sites. Extensive mapping experience has shown that transitional boundaries between unconsolidated substrate classes can be mapped to within ±20-50m where they have been thoroughly ground-truthed and validated with particle grain size samples.

Areal extent measurements are typically acquired from broad scale maps using established NEMP methods, with up-to date aerial or satellite imagery used to record substrate features. The ability to detect change increases with increasing measurement frequency, spatial resolution and accuracy. Ground-truthing maps are ideally <50cm/per pixel resolution at a scale of between 1:2000 and 1:5000, as at a coarser pixel resolution and scale it becomes difficult to reliably characterise features. Annotated field maps of validated features, combined with field notes and georeferenced photographs, are digitised into shapefiles to produce maps of substrate extent. Machine learning with automated digitisation based on the spectral analysis of imagery remains in development in New Zealand (e.g., Ha et al. 2020), but is not expected to be suitable for determining mud-elevated (25% mud content) sediment extent.

Assessment baseline: The first accurately measured measurement of mud-elevated sediment extent in the AIH, excluding salt marsh habitat which is well known as an effective trap of estuary sediment.

Measurement considerations: Estuary-wide substrate classification using NEMP visual assessment methods is considered a screening approach for determining the potential scale of contemporary sediment issues. Where improved accuracy in the definition of substrate boundaries is required, additional sampling approaches are recommended such as the use of targeted grain size analyses along fixed transects, or through stratified, random or grid sampling approaches, often applied at a sub-estuary scale. While it is possible to extend sampling into sub-tidal areas, e.g., through wading, SCUBA surveys or remote grab sampling, the NEMP methods are designed primarily for use in intertidal areas.

Historical mud-elevated extent is generally overlain by subsequent estuary infilling making determination difficult, although it can be assessed by analysis of subsurface features, ranging from simple approaches such as digging holes to reveal underlying sediment layers, to more complex sediment coring methods involving analysis of grain size accompanied by carbon, radioisotope or pollen analyses to date sediments and determine deposition rates (see Appendix 7 for further detail). Such approaches are very useful for improving the understanding of contemporary state.

There are inconsistences in current New Zealand sampling methods, mapping accuracy, and classification criteria. Method accuracy is expected to be ~±10% of the true value for repeat measurements conducted by the same provider, but potentially highly variable between providers.

Statistic calculated from time series of mud-elevated areal extent: Percent change in areal extent (ha) of intertidal mud-elevated (>25% mud content) sediment, excluding salt marsh habitat, from the first accurate baseline:

$$\text{Percent change in intertidal mud-elevated (>25\% mud content) extent} = \frac{\text{(Baseline extent - Current extent)}}{\text{(Baseline extent)}} \times 100$$

Potential bands and/or thresholds and rationale (including caveats): Due to the high level of uncertainty with spatial thresholds proposed in Table A3-1 above, and known impacts from increasing mud, there is logic in suggesting that guidance to trigger further investigation of natural versus anthropogenic change would be 'any measurable increase in the areal coverage of intertidal mud-elevated substrate from its current extent'. It is proposed that this guidance be supported by tentative thresholds for the scale of change (Table A3-2) and should be targeted toward early identification of potential issues. Natural sediment infilling needs to be factored into any assessment of change.

Allowing for potential variability in natural extent and mapping accuracy, it is recommended that increases of ≥5 to <10% are used as a precautionary threshold to trigger decisions regarding the need for further evaluation and active management (Table A4-2). Increases of ≥10% and ≥20% are proposed breakpoints for 'Poor' and 'Very Poor' bands indicative of potentially significant degradation, and increases of <5%, or decreases in extent, reflect the 'Good' and 'Very Good' bands. The relatively high level of stringency of the proposed thresholds reflects that contemporary measures will be made over relatively short time frames, e.g., 3-5 years, and should be targeted toward early identification of potential issues. Analysis of recently measured changes in New Zealand estuaries is recommended to refine threshold breakpoints.

When interpreting this metric, it is important to consider what the scale of percent change means in absolute terms, i.e., change in ha. For example, a small percent increase in an estuary with a large mud-elevated extent may represent a significant area of habitat loss that is not immediately captured by the proposed percent change thresholds. Conversely, in an estuary where the starting extent is small, a large percent increase may only represent a small increase in spatial extent and be of less relative importance. It is beyond current scope to consider this matter in greater detail, but it is important that councils and providers keep it in mind, and further work is required if thresholds are to be scaled based on estuary size or type.

Waikawa Estuary, Southland.

Summary of proposed thresholds:

Table A3-2. Recommended thresholds for the percent increase of intertidal mud-elevated (>25% mud content) sediment in the AIH, excluding salt marsh habitat, from the first accurate baseline.

Percent increase in mud-elevated sediment from first accurate baseline*	Ecological Quality Status				
	Very Good	Good	Fair	Poor	Very Poor
	≤0	>0 to <5%	≥5% to <10%	≥10% to <20%	≥20%
Narrative	Mud-elevated substrate reducing. No change or potential improvement in ecological function.	Mud-elevated substrate stable or within expected natural variation or mapping accuracy. Negligible impact in ecological function.	Measurable increase in mud-elevated substrate attributable to anthropogenic impacts. Potentially moderate impact on ecological function.	Large increase in mud-elevated substrate attributable to anthropogenic impacts. Potentially large impact on ecological function.	Very large increase in mud-elevated substrate attributable to anthropogenic impacts. Large impact on ecological function.

*Assessment needs to consider change due to natural processes, e.g. shoreline erosion, noting anthropogenic climate change may increase impacts.

Overall confidence in thresholds/ bands: High. Thresholds are based on expert judgement, and suited to the early detection of change within limits of method accuracy, but require refinement based on an assessment of existing New Zealand data. However, increases in mud-elevated substrate from its current extent should trigger an early warning to address potential causes and significance of any change.

Recommendation: Change in intertidal mud-elevated (>25% mud) sediment extent from the first accurate baseline

Adopt Table A3-2 as guidance only thresholds pending analysis/review of New Zealand data.

Links to other indicators: Other indicators that serves as explanatory variables for changes in mud-elevated substrate extent include catchment sediment loads, sedimentation rate, mud content, water quality indicators (e.g., clarity, turbidity) and climate variables (e.g., wind, temperature, rainfall etc.). The impact of anthropogenic stressors on ecosystems may be highly context specific (i.e., place and history are very important), compounded by high variability in pressures such as physical damage, changes in sea level, severe storm frequency and intensity, brought about by climate change.

Alternative metrics considered: No alternative metrics considered for change in spatial extent.

Additional work recommended:

i. Collate standardised national data (and associated metadata) on measured mud-elevated extent.
ii. Analyse within and between provider mapping accuracy and assess confidence intervals on the assessment of temporal and spatial change.
iii. Revise interim thresholds based on i. and ii. to refine percent loss breakpoints.
iv. Undertake further studies to determine the potential historical mud-elevated extent of New Zealand estuaries.
v. Investigate development of supporting thresholds based on change measured in hectares.

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Flood deposition of mud on cockle beds, Delaware Inlet, Nelson.

Deposits of terrestrial mud overlying sand/shell-dominated sediment, Moutere Inlet, Tasman.

APPENDIX 4. SALT MARSH

Author: Leigh Stevens (Salt Ecology)

Salt marshes are found throughout New Zealand in the low energy upper tidal margins of estuaries and coastlines that are periodically inundated by sea water, and where terrestrial plants are unable to survive. Most estuarine salt marsh grows in the upper tidal zone between mean high water neap (MHWN) and mean high water spring (MHWS) tide height where vegetation stabilises and traps fine sediment transported by tidal flows (e.g., Balke et al 2016).

BACKGROUND

Salt marshes are important biogenic habitats dominated by salt-tolerant plant species present at the land-sea interface. Salt marsh zonation is commonly evident, resulting from the combined influence of factors including salinity, inundation period, elevation, wave exposure, and sediment type. Salt marshes are naturally species poor compared to terrestrial systems, but have high biodiversity and provide habitat for a variety of plant, bird, fish and insect life, are amongst the most productive habitats on earth, offer a wide range of ecosystem services (e.g., sediment stabilisation, erosion mitigation), and have strong aesthetic appeal.

Historical losses have been substantial in New Zealand, with large tracts of salt marsh cleared, drained and reclaimed for activities including farming, urban development and roading. Remaining salt marsh is sensitive to a wide range of ongoing pressures including grazing, weed invasion, reclamation, and altered flow regimes, e.g., culverting, flap-gates and causeways. Climate change has the potential to add to these pressures through increased storm frequency and particularly sea level rise contributing to coastal squeeze where displaced salt marsh cannot migrate inland due to anthropogenic or natural barriers. The actual effects will depend on how widespread and intense the pressures are, whether there are single or multiple pressures, and the effectiveness of management and mitigation approaches. Nature-based approaches utilising salt marsh to mitigate against erosion and to restore biodiversity are becoming relatively common in New Zealand estuaries, albeit at a local scale.

Salt marsh responds to natural and human disturbances primarily through changes in spatial extent. Rates of natural recovery are generally slow (up to 40 years), and displaced or degraded salt marsh may not fully recover without additional interventions (e.g., Kelleway 2006, Martin 2008). Coarse abundance measurements (e.g., areal extent) allow for assessment of salt marsh distribution and can be used to detect large-scale temporal gains or losses of habitat. Detailed surveys repeated over a period of time are generally required to assess local or region-specific expansion or retraction at a meaningful management scale. Both can be collected using well established rapid and non-destructive monitoring approaches. A key limitation in assessing changes in salt marsh extent is the often-limited availability of data on historical extent, although this can be estimated using historical imagery or maps, combined with elevation data, e.g. LiDAR, to indicate areas likely to have been inundated prior to human modification, e.g., prior to the creation of tidal flap-gates, bunds.

Measurements of other salt marsh parameters and assessments of salt marsh quality (e.g., plant inventories, vegetative biomass, biodiversity, carbon sequestration, sediment accrual) are not routinely collected as part of SOE monitoring because they are generally labour intensive, are often site- or patch-specific, and lack standardised national protocols leading to limited data and variable baselines for comparison (Lohrer et al. in prep.).

PROPOSED METRICS

The following indicator metrics are proposed for assessing salt marsh.

1. Salt marsh extent as a percentage of the intertidal area *suitable* for salt marsh.
2. Change in salt marsh extent relative to the first accurate baseline.
3. Change in salt marsh extent relative to estimated historical extent.

It is recommended that these three metrics are used together in the assessment of salt marsh. Salt marsh quality is not recommended for threshold development at this point in time due to limited data, absence of agreed indicators, and a lack of standardised national protocols.

4.1 SALT MARSH EXTENT AS A PERCENTAGE OF THE INTERTIDAL AREA SUITABLE FOR SALT MARSH

Indicator type: Supporting.

Metric: Salt marsh extent as a percentage of the intertidal area suitable for salt marsh.

Unit of measurement: Percent (%) of 'Available Salt marsh Habitat' (ASH). Note: We propose use the term ASH to refer to the intertidal area suitable for salt marsh, and define it as 100% of the estuary area between mean high-water neap (MHWN) and mean high water spring (MHWS) tide height.

Spatial scale: Estuary-wide between MHWN and MHWS.

Applicability: All New Zealand estuarine and coastal waters without mangroves, where a suitable habitat for salt marsh exists, specifically intertidal areas between MHWN and MHWS tide height. Salt marsh species growing in terrestrial zones not tidally inundated with saline water should be excluded from the metric.

Rationale: Areal extent is an efficient and cost-effective indicator of salt marsh and is sensitive enough to detect broad spatial and temporal changes when measurements are repeated over time. Temporal changes in the position and/or size of salt marsh can be assessed by repeat measurements and related to the effects of anthropogenic activities such as vegetation clearance, reclamation, or physical disturbance, and to a lesser extent impacts related to fine sediment or nutrients. The metric can be consistently applied across different scales of data resolution, e.g., national scale remote sensing to detailed estuary-wide surveys. From a management perspective, most instances of salt marsh loss can be directly related to individual stressors.

Method: Areal extent measurements are typically acquired from broad scale maps using established National Estuary Monitoring Protocol (NEMP) methods (e.g., Robertson et al. 2002, Stevens et al., in prep), with up-to date aerial or satellite imagery used to record salt marsh features. Data can be classified from a single sampling event, with the ability to detect change improved by increasing spatial resolution and accuracy. Machine learning with automated digitisation based on the spectral analysis of imagery remains in development in New Zealand (e.g., Ha et al. 2020), with few councils currently using it as a primary method.

Field ground-truthing surveys are typically carried out on foot by experienced practitioners to validate features visible on imagery, and to characterise dominant salt marsh species composition and extent. Ground-truthing maps are ideally <50cm/per pixel resolution at a scale of between 1:2000 and 1:5000, as at a coarser pixel resolution and scale it becomes difficult to reliably characterise salt marsh features. Annotated field maps of validated salt marsh features, combined with field notes and georeferenced photographs, are digitised into shapefiles to produce maps of salt marsh extent.

As defined above, the metric is expressed as salt marsh extent as a percentage of the ASH. The area suitable for salt marsh growth will vary based on individual estuary size, type and local climate, substratum and hydrodynamic regime. New Zealand estuaries include large intertidal areas unsuited for salt marsh growth (i.e., areas outside of the MHWN and MHWS range of the ASH).

Assessment baseline: The first quantitative areal extent measurements obtained by a combination of broad scale mapping and extensive ground-truthing.

Measurement considerations: Within-estuary measurements of salt marsh extent should be taken at the same time of each year (ideally around the peak of growth in summer, e.g., October - March) to limit the effect of seasonal changes on measurement results. This metric is applied only to the contemporary areas suitable for salt marsh and does not incorporate areas cut-off by flapgates or bunds, etc. These areas are addressed under the metric for 'change in salt marsh extent relative to estimated historical extent' (see Section 4.3).

There are inconsistences in current New Zealand sampling methods, mapping accuracy, and classification criteria for determining salt marsh extent. Method accuracy is estimated to be ~±5% of the true value, and reasonably precise for repeat measurements conducted by the same provider, but both could vary between providers. Development of new technologies, e.g., remote sensing and automated mapping, will also require future consideration.

Statistic calculated from the measurement of salt marsh extent: Salt marsh extent (ha) as a percentage of the Available Salt marsh Habitat (ASH):

$$\text{Salt marsh extent} = \frac{\text{Current measured salt marsh extent (ha)}}{\text{Available Salt marsh Habitat (ha)}} \times 100$$

Potential bands and/or thresholds and rationale (including caveats): It has been suggested internationally that a fully functioning salt marsh should cover between 25-50% of the intertidal area suitable for salt marsh growth (De Jong 2004, Dijkema et al. 2004), equivalent to our definition of the ASH above. Thresholds for the European Union were proposed for 5 bands with breakpoints at 5, 10, 25 and 50% of salt marsh as a percentage of the suitable intertidal area (WFD-UKTAG 2014).

The WFD thresholds are potentially applicable to New Zealand estuaries and could be validated relatively easily using available data on salt marsh extent and intertidal estuary bathymetry derived from existing LiDAR data or estuary models. Exceptions to the thresholds are expected be for estuaries containing mangroves (*Avicennia marina* subsp. *australasica*) or the introduced cord grass (*Spartina* spp.), the latter a targeted pest plant now relatively limited in extent. Both species can grow above MHWN tide height as well as below this zone and can compete with salt marsh for space.

Previously in New Zealand, Salt Ecology have applied tentative screening criteria to assist Councils in assessing salt marsh extent, with threshold boundaries for 4 bands set at 5, 10, and 20% of the intertidal area. The subjectivity of proposed band thresholds has been acknowledged (e.g., Stevens and Forrest 2020, Stevens et al. 2020, Stevens et al. 2023), with use limited to broadly assessing ecological status in conjunction with estimated change from historical extent.

The international WFD-UKTAG (2014) criteria are considered suitable for adoption as numerical criteria until such time as the extent of salt marsh within the ASH in New Zealand estuaries is assessed, and New Zealand-specific thresholds can be developed. In light of the limited analysis of salt marsh extent in estuaries with mangroves, it is recommended that these thresholds be applied as guidance-only in estuaries with mangroves, until such time as additional assessment is undertaken.

A narrative of **ecological quality status** enables changes in areal extent to be related to ecological condition. Ecological quality status attempts to characterise the degree of change on a continuum from natural state to highly degraded to provide ecological context to current state, and measured changes from it.

Summary of proposed thresholds:

Table A4-1: Recommended thresholds for salt marsh extent (ha) as a percentage of the Available Salt marsh Habitat (ha) in New Zealand estuaries without mangroves.

	Ecological Quality Status				
Percent of ASH*	Very Good	Good	Fair	Poor	Very Poor
	≥50%	≥25 to <50%	≥10 to <25%	≥5 to <10%	≥0 to <5%
Narrative	Salt marsh is at the upper end of expected natural extent, and retains full ecological function.	Salt marsh within expected natural extent, and maintains full ecological function.	Moderate reduction in salt marsh extent from expected natural extent attributable to anthropogenic impacts. Moderate impact on ecological function.	Substantial reduction in salt marsh extent attributable to anthropogenic impacts. Potentially large impact on ecological function.	Very large reduction in salt marsh extent attributable to anthropogenic impacts. Large impact on ecological function.

*ASH = Available Salt marsh Habitat defined as 100% of the estuary area between MHWN and MHWS.

Overall confidence in thresholds/ bands: Fair. Thresholds are based on international guidance, but require refinement based on an assessment of existing New Zealand data.

Recommendation: Salt marsh extent as a percentage of the ASH

Adopt Table A4-1 as preliminary numeric thresholds pending analysis/review of New Zealand data. Substantial data on the broad scale extent of salt marsh (including estimates of historical extent), have been collected as part of council SOE monitoring. To date, these data have not been compiled into a national data set or analysed.

Links to other indicators: Salt marsh extent should also be considered in the context of likely losses compared to both historical and contemporary baselines (see following sections). Indicators that serve as explanatory variables for changes in salt marsh extent include land drainage, flow regulation (e.g., flap gates), direct physical damage (vegetation clearance, grazing, reclamation), displacement by mangroves or *Spartina*, and changes in sea level, severe storm frequency and intensity, and temperature brought about by climate change. The impact of anthropogenic stressors on ecosystems may be highly context specific (i.e., place and history are very important).

Alternative metrics considered: No alternative metrics considered for spatial extent.

Additional work recommended:

i. Collate standardised national data (and associated metadata) on salt marsh extent.
ii. Assess measured salt marsh extent relative to Available Salt marsh Habitat (ASH).
iii. Revise interim thresholds based on ASH.
iv. Investigate development of a salt marsh multi-metric for New Zealand, similar to that developed by the WFD, which combines measures of current extent, loss from historical extent, and loss from a contemporary baseline.
v. Assess the degree of potential salt marsh displacement by mangroves or *Spartina*.
vi. Assess the need to develop separate guidance for estuaries containing mangroves or *Spartina*.

Freshwater Estuary, Rakiura, Southland.

4.2 CHANGE IN SALT MARSH EXTENT FROM THE FIRST ACCURATE BASELINE

Indicator type: Primary.

Metric: Change in salt marsh extent from the first accurate baseline.

Unit of measurement: Percent (%) change in areal extent from baseline.

Spatial scale: Estuary-wide between MHWN and MHWS.

Applicability: All New Zealand estuarine and coastal waters where a suitable habitat for salt marsh exists (ASH as defined in 4.1 above), specifically intertidal areas between mean high water neap (MHWN) and mean high water spring (MHWS) tide height. This metric of change can be applied to estuaries with mangroves.

Rationale: Areal extent is an efficient and cost-effective indicator of salt marsh and is sensitive enough to detect broad spatial and temporal changes when measurements are repeated over time. Reliable quantification of temporal change in extent requires an accurate baseline measurement of salt marsh extent. Temporal changes in the position and/or size of salt marsh can be assessed by repeat measurements and related to the effects of anthropogenic activities such as vegetation clearance, reclamation, or physical disturbance, and to a lesser extent impacts related to fine sediment or nutrients, or salt marsh displacement by mangroves or *Spartina*. A decrease in salt marsh extent likely indicates a loss of ecological value and function. A change in extent can be consistently applied across different scales of data resolution, e.g., national scale remote sensing to detailed estuary-wide surveys. From a management perspective, most instances of salt marsh loss can be directly related to individual stressors.

Method: An accurate salt marsh baseline can be obtained by a combination of broad scale mapping and extensive ground-truthing. Hence, as described for 4.1, broad-scale maps for baseline conditions should have been developed using established National Estuary Monitoring Protocol (NEMP) methods (e.g., Robertson et al. 2002, Stevens et al., in prep) or similar systematic methods, with up-to date aerial or satellite imagery used to record salt marsh features. Depending on the extent of ground-truthing and the QA/QC methods (if any) used in initial surveys, it may be necessary to update data following data QA/QC checks, e.g., to remove any overlapping or duplicated polygons, or exclude terrestrial features.

Assessment baseline: The first quantitative areal extent measurements obtained by a combination of broad scale mapping and extensive ground-truthing.

Measurement considerations: As noted under 4.1, the baseline measurements of salt marsh should ideally have been taken at the same time of each year (ideally around the peak of growth in summer, e.g., October - March) to limit the effect of seasonal changes on assessment of temporal change.

If resources are limited, measurement frequency for each estuary, or representative estuaries in a region, would ideally be determined by a risk assessment (i.e., higher frequency measurement for estuaries with higher pressures or greater risk of loss). Where a problem is identified (e.g., salt marsh health appears compromised, there is evidence of significant physical damage, or salt marsh is suspected to be undergoing rapid decline) measurements should be repeated annually, or at least once every 3 years. Where there is no obvious problem, measurements can be repeated every 5 years, or longer intervals may be recommended based on site data and expert judgement.

There are inconsistences in current New Zealand sampling methods, mapping accuracy, and classification criteria. Method accuracy is estimated to be ~±5% of the true value and reasonably precise for repeat measurements conducted by the same provider, but both could vary between providers. Development of new technologies, e.g., remote sensing and automated mapping, will also require future consideration.

Statistic calculated from time series of salt marsh extent: Percent change in areal extent (ha) of salt marsh from the first accurate baseline:

$$\text{Percent change in salt marsh extent from first accurate baseline} = \frac{(\text{Baseline extent} - \text{Current extent})}{(\text{Baseline extent})} \times 100$$

To understand the rate of change and enable comparison among estuaries, the percent change in salt marsh extent could be expressed on an annualised basis by dividing by the number of years since the baseline was established. Further consideration of this approach was beyond what could be included in the current project scope.

Potential bands and/or thresholds and rationale (including caveats): The most comprehensive thresholds of salt marsh change relative to a baseline appear to be the European WFD (Davey 2013, WFD-UKTAG 2014). The natural condition is no loss (0%), or an increase of salt marsh extent. The upper WFD breakpoint is set at a precautionary 10% loss on the basis that there may be relatively high natural variability in salt marsh extent (estimated at ~±20%). While natural variability in New Zealand salt marsh extent is not well documented the ±20% WFD rate of natural variability is substantially above the ±5% we estimate for New Zealand salt marsh based on field observations over decadal scales and our expert opinion.

WFD thresholds are set as bands of 0-10, 10-25, 25-50, 50-75, and >75% loss (rated High/Good/Moderate/Poor/Bad) (WFD-UKTAG 2014 and references therein). The rationale for the thresholds is unclear and appears arbitrary. Given that salt marsh is of high ecological value, and further anthropogenic losses are difficult to justify or reverse, it seems undesirable to rate losses of up to 25% as 'Good'. Instead, for New Zealand, it is recommended that the 'Very Good' and 'Good' ecological status bands should reflect <5% further loss attributable to anthropogenic change. Allowing for potential variability in natural extent and mapping accuracy, a threshold band of ≥10% loss is proposed as a precautionary threshold to trigger decisions regarding the need for further evaluation and active management (Table A4-2). Losses of ≥20% are considered likely indicative of potentially significant degradation. The higher level of stringency compared to the WFD criteria reflects that contemporary measures will be made over relatively short time frames, e.g., 3-5 years, and should be targeted toward early identification of potential issues. Analysis of recently measured changes in New Zealand estuaries is recommended to refine threshold breakpoints.

When interpreting this metric, it is important to consider what the scale of percent change means in absolute terms, i.e., change in ha. For example, a small percent loss in an estuary with a large salt marsh extent may represent a significant area of habitat loss that is not immediately captured by the proposed percent change thresholds. Conversely, in an estuary where the starting extent is small, a large percent loss may only represent a small decrease in spatial extent and be of less relative importance. It is beyond current scope to consider this matter in greater detail, but it is important that councils and providers keep it in mind, and further work is required if thresholds are to be scaled based on estuary size or type.

A narrative of ecological quality status enables changes in areal extent to be related to ecological condition. Ecological quality status attempts to characterise the degree of change on a continuum from natural state to highly degraded to provide ecological context to current state, and measured changes from it.

Mapping salt marsh, Whangarae Estuary, Marlborough.

Summary of proposed thresholds:

Table A4-2. Tentative ecological quality status boundaries for the percent loss of salt marsh extent from the first accurate baseline in New Zealand estuaries.

Percent salt marsh loss from first accurate baseline*	Ecological Quality Status				
	Very Good	Good	Fair	Poor	Very Poor
	≤0	>0 to <5%	≥5% to <10%	≥10% to <20%	≥20%
Narrative	Salt marsh extent is stable or expanding, and maintains natural ecological function.	Salt marsh extent within expected natural variation or mapping accuracy, and maintains natural ecological function.	Moderate decline in salt marsh extent attributable to anthropogenic impacts. Moderate impact on ecological function.	Large decline in salt marsh extent attributable to anthropogenic impacts. Potentially large impact on ecological function.	Very large decline in salt marsh extent attributable to anthropogenic impacts. Large impact on ecological function.

*Assessment needs to consider losses from natural processes, e.g. shoreline erosion, noting anthropogenic climate change may increase impacts.

Overall confidence in thresholds/ bands: High. Thresholds are based on expert judgement, taking into account international guidance, but require refinement based on an assessment of existing New Zealand data. Consideration should be given to developing specific thresholds applicable to short-term annual change and longer-term change, e.g., 10-year intervals. To understand the rate of change and enable comparison among estuaries, the percent change in salt marsh extent could be expressed on an annualised basis by dividing by the number of years since the baseline was established. Further consideration of this approach was beyond what could be included in the current project scope.

Recommendation: Change in salt marsh extent relative to the first accurate baseline:

Adopt Table A4-2 as preliminary numeric thresholds pending analysis/review of New Zealand data.

Links to other indicators: Salt marsh losses should be considered in the context of both historical and contemporary extent (see related sections). Indicators that serve as explanatory variables for changes in salt marsh extent include land drainage, flow regulation (e.g., flap gates), direct physical damage (vegetation clearance, grazing, reclamation), displacement by mangroves or *Spartina*, and changes in sea level, severe storm frequency and intensity, and temperature brought about by climate change. The impact of anthropogenic stressors on ecosystems may be highly context specific (i.e., place and history are very important).

Alternative metrics considered: No alternative metrics considered for salt marsh loss.

Additional work recommended:

i. Collate standardised national data (and associated metadata) on measured salt marsh losses.
ii. Assess measured salt marsh losses attributable to natural processes or variation.
iii. Revise interim thresholds based on ii. to refine percent loss breakpoints.
iv. Investigate development of a salt marsh multi-metric index for New Zealand, similar to that developed by the WFD, which combines measures of current extent, loss from historical extent, and loss from a contemporary baseline.
v. Consider the merit of expressing percent change in salt marsh extent on an annualised basis (dividing by the number of years since the baseline was established) to enable standardised comparison among estuaries.

4.3 CHANGE IN SALT MARSH EXTENT FROM ESTIMATED HISTORICAL EXTENT

Indicator type: Supporting.

Metric: Change in salt marsh extent from estimated historical extent.

Unit of measurement: Reported as percent (%) change in areal extent from historical estimate.

Spatial scale: Estuary-wide between MHWN and MHWS.

Applicability: All New Zealand estuarine and coastal waters where a suitable habitat for salt marsh exists (ASH as defined in 4.1 above), specifically intertidal areas between mean high water neap (MHWN) and mean high water spring (MHWS) tide height.

Rationale: Due to the historical modification of many New Zealand estuaries (primarily from reclamation and drainage), the previous two metrics (current salt marsh extent or loss from the first accurate baseline) may not appropriately characterise ecological vulnerability relative to historical extent. The metric of percent change from historical extent therefore places the measurement of contemporary state (and changes from it) into a wider context. Areal extent is an efficient and cost-effective indicator of salt marsh and is sensitive enough to detect broad spatial and temporal changes when measurements are repeated over time. Temporal changes in the position and/or size of salt marsh can be assessed by repeat measurements and related to the effects of anthropogenic activities such as vegetation clearance, reclamation, or physical disturbance, and to a lesser extent impacts related to fine sediment or nutrients. A decrease in salt marsh extent likely indicates a loss of ecological value and function. A change in extent can be consistently applied across different scales of data resolution, e.g., national scale remote sensing to detailed estuary-wide surveys. Maps of historical extent also highlight areas potentially suitable for salt marsh restoration in response to predicted inundation from sea level rise.

Method: Substantial modification from natural state has generally occurred prior to the earliest available historical imagery (ca. 1940's in New Zealand). Historical extent therefore needs to be predicted, in most instances, from a combination of elevation data (e.g., LiDAR, topographical maps), historical maps, early aerials, paintings and photographs, written descriptions, oral histories and local knowledge. Contemporary ground-truthed maps can also help in identifying historical features visible on older images. Historical extent may also be estimated using modelling approaches that factor in key requirements for salt marsh growth, specifically the MHWN to MHWS tidal range. Expert judgement is required to interpret data, and outputs will generally be less accurate than contemporary mapping practices which are based on ground-truthing. Data can be classified from a single mapping event, with the ability to detect change improved by increasing spatial resolution and the accuracy or detail of available historical information.

Assessment baseline: The estimated historical areal extent of salt marsh prior to human modification.

Measurement considerations: If deriving maps from historical imagery in the absence of ground truthing, spatial accuracy and resolution of historical imagery is generally lower than for more recent imagery, and some salt marsh features, e.g., herbfield can be difficult to distinguish in black and white images. Further, where imagery is not collected at low tide, some features may be obscured. Species composition is also very difficult to accurately determine in the absence of ground-truthing or local knowledge.

Statistic calculated from time series of salt marsh extent: Percent change in areal extent (ha) of salt marsh from the estimated historical extent:

$$\text{Percent change in salt marsh extent from historical baseline} = \frac{(\text{Estimated historical extent (ha)} - \text{Current extent (ha)})}{\text{Estimated historical extent (ha)}} \times 100$$

Potential bands and/or thresholds and rationale (including caveats): Thresholds of the percent of historical salt marsh remaining are included in the European WFD (Davey 2013, WFD-UKTAG 2014) with breakpoints set evenly at 20, 40, 60 and 80% (applied in the WFD as the proportion of historic extent remaining as opposed to loss from historical extent). These thresholds have previously been applied in New Zealand as interim guidance, (e.g., Stevens and Forrest 2020, Stevens et al. 2020). While such thresholds make intuitive sense (the greater the loss from historical extent, the greater the likely decline in ecological biodiversity and resilience), the rationale for the specific thresholds is unclear. Given their seemingly arbitrary selection, evaluation of changes in New Zealand estuaries from predicted historical state is recommended to refine threshold breakpoints.

When interpreting this metric, it is important to consider what the scale of percent change means in absolute terms, i.e., change in ha. For example, a small percent loss in an estuary with a large salt marsh extent may represent a significant area of habitat loss that is not immediately captured by the proposed percent change thresholds. Conversely, in an estuary where the starting extent is small, a large percent loss may only represent a small decrease in spatial extent and be of less relative importance. It is beyond current scope to consider this matter in greater detail, but it is important that councils and providers keep it in mind, and further work is required if thresholds are to be scaled based on estuary size or type.

A narrative of ecological quality status enables changes in areal extent to be related to ecological condition. Ecological quality status attempts to characterise the degree of change on a continuum from natural state to highly degraded to provide ecological context to current state, and measured changes from it.

Summary of proposed thresholds:

Table A4-3: Tentative ecological quality status boundaries for the percent loss of salt marsh areal extent from estimated historical extent in New Zealand estuaries.

Percent salt marsh loss from estimated historical extent*	Ecological Quality Status				
	Very Good	Good	Fair	Poor	Very Poor
	<0% to <20%	≥20% to <40%	≥40% to <60%	≥60 % to <80%	≥80% loss
Narrative	Salt marsh within expected natural variation of natural state, and is stable or expanding.	Salt marsh extent reduced beyond expected natural variation, and maintains natural state ecological function.	Measurable decline in salt marsh extent attributable to anthropogenic impacts. Moderate impact on ecological function.	Substantial decline in salt marsh extent attributable to anthropogenic impacts. Large impact on ecological function.	Very large decline in salt marsh extent attributable to anthropogenic impacts. Large impact on ecological function.

*Assessment needs to consider losses from natural processes, e.g., shoreline erosion, noting anthropogenic climate change may increase impacts.

Overall confidence in thresholds/ bands: Fair. Thresholds are based on expert judgement, taking into account international guidance, but require refinement based on an assessment of existing New Zealand data. In particular, the scale of losses proposed (adopted from WFD criteria) appear high, e.g., the classification of 20-40% losses as 'Good' likely underestimates ecological degradation.

Recommendation: Change in salt marsh extent from historical extent.

Adopt Table A4-3 as preliminary numeric thresholds pending analysis/review of New Zealand data.

Links to other indicators: Historical salt marsh extent should also be considered in the context of both contemporary extent and recent change (see related sections). Indicators that serve as explanatory variables for changes in salt marsh extent include land drainage, flow regulation (e.g., flap gates), direct physical damage (vegetation clearance, grazing, reclamation), and changes in sea level, severe storm frequency and intensity, and temperature brought about by climate change. The impact of anthropogenic stressors on ecosystems may be highly context specific (i.e., place and history are very important).

Alternative metrics considered: No alternative metrics considered.

Additional work recommended:

i. Collate standardised national data (and associated metadata) on historical salt marsh extent.

ii. Revise interim thresholds based on ii. to refine percent loss breakpoints.
iii. Investigate development of a salt marsh multi-metric index for New Zealand, similar to that developed by the WFD, which combines measures of current extent, loss from historical extent, and loss from a contemporary baseline.
iv. Consider the merit of expressing percent change in salt marsh extent on an annualised basis (dividing by the number of years since the baseline was established) to enable standardised comparison among estuaries.

Whanganui/Westhaven Inlet, Tasman.

4.4 SALT MARSH QUALITY

Indicator type: Supporting.

Metric: Various. Examples include measures of vegetative biomass, species composition, biodiversity, carbon sequestration, sediment accrual, fragmentation, disturbance recovery rates, introduced species presence.

Unit of measurement: Various.

Spatial scale: Site-specific within MHWN and MHWS.

Applicability: All New Zealand estuarine and coastal waters where a suitable habitat for salt marsh exists (ASH as defined in 4.1 above), specifically intertidal areas between mean high water neap (MHWN) and mean high water spring (MHWS) tide height.

Rationale: Salt marsh quality is closely tied to ecological integrity. However, salt marsh quality measurements are not routinely collected in New Zealand because they are generally labour intensive, are often site- or patch-specific and lack standardised national method protocols, contributing to a lack of consistent data and variable baselines for comparison (Lohrer et al. in prep.). Further, New Zealand-specific data that quantifies stressor impacts on 'quality' and associated ecosystem services are limited, and data on tipping points are lacking (Lohrer et al. in prep.). Consequently, development of thresholds for salt marsh quality are not recommended. However, individual quality metrics may be useful for determining the specific impact of identified management interventions, e.g. recovery after stock exclusion.

Methods: Various – not described.

Measurement considerations: Due to the relatively high cost of salt marsh quality assessments, and the likelihood that stressors will determine priorities at a site-specific scale, expert appraisal of monitoring requirements with regard to management outcomes is recommended.

Statistic: None proposed.

Potential bands and/or thresholds and rationale (including caveats): None proposed. High level narrative thresholds have potential to be developed as preliminary screening criteria to help determine if more detailed investigation is warranted. Substantial further work required to refine narrative thresholds.

Summary of proposed thresholds: None proposed.

Overall confidence in thresholds/ bands: Undeveloped

Recommendation: Salt marsh quality

No further consideration of numeric thresholds is recommended. Development of narrative thresholds for classifying different states of visually observable degradation related to physical impacts, e.g. grazing, presence of introduced species, could be considered.

Links to other indicators: Salt marsh extent, and losses compared to historical and contemporary baselines (see previous sections) are expected to provide an effective way to determine if salt marsh quality is potentially compromised. Indicators that serve as explanatory variables for changes in salt marsh extent, and likely degradation of salt marsh quality, include land drainage, flow regulation (e.g., flap gates), direct physical damage (vegetation clearance, grazing, reclamation), and changes in sea level, severe storm frequency and intensity, and temperature brought about by climate change. The impact of anthropogenic stressors on ecosystems may be highly context specific (i.e., place and history are very important) and should be taken into account when determining monitoring priorities.

Additional work recommended:

i. Review sampling methods and reporting metrics to determine whether standardised national data can be compiled in future.
ii. Consider developing visual guides for classifying different states of visually observable degradation related to physical impacts, e.g. grazing, presence of introduced species, to facilitate consistency in reporting.

iii. Consider whether a rapid-screening metric for salt marsh quality could be developed from the above to derive potential narrative thresholds of salt marsh quality.

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APPENDIX 5. SEAGRASS

Author: Leigh Stevens (Salt Ecology)

Seagrass (also known as eelgrass) comprises one marine species in New Zealand - *Zostera muelleri*. It is found throughout New Zealand from Northland to Stewart Island (Anderson et al. 2019) in sheltered, intertidal and shallow (<5m depth) subtidal estuarine and coastal waters.

BACKGROUND

Seagrass is recognised as an important marine biogenic habitat type that provides shelter and food for other species, is an important nursery habitat for juvenile fish, stabilises the seabed and influences biogeochemical processes such as nutrient cycling (e.g., Turner and Schwarz 2006, Matheson et al. 2009, Anderson et al. 2019, Morrison et al. 2009).

In general, seagrass has declined in extent in New Zealand over the last 60 years (Matheson et al. 2009, 2011, Anderson et al. 2019). **Natural losses** can occur through severe storms which can uproot seagrass plants, wave or channel scouring, or by a slime mould wasting disease which can affect growth. However, human-induced pressures are likely responsible for most of the observed seagrass loss in New Zealand (e.g., Turner and Schwarz 2006, Matheson et al. 2009). **Pressures** include runoff of sediment, nutrients, and other contaminants from land (Zabarte-Maeztu et al. 2021); physical damage caused by dredging, reclamation, coastal development, mooring; overgrazing by introduced species (i.e., black swans); competition from invasive species (e.g., *Spartina* spp. and *Caulerpa* spp.); or displacement by species such as mangroves. **Climate change** has the potential to add to these pressures, via sea level rise, increasing temperatures and increased storm frequency. The actual effects will depend on how widespread and intense the pressures are, whether there are single or multiple pressures, and the effectiveness of management approaches.

Seagrass responds to **natural and human disturbances** through changes in spatial extent, percent cover, density (number of plants), biomass and/or morphology (e.g., leaf length or width) (Zabarte-Maeztu et al. 2021). Coarse abundance measurements (e.g., areal extent) allow for assessment of seagrass distribution and can be used to detect large-scale temporal gains or losses of seagrass habitat. Percent cover observations repeated over a period of time are generally required to assess local or region-specific seagrass expansion or retraction at a meaningful management scale. Both abundance and percent cover can be collected using rapid and non-destructive monitoring approaches sensitive enough to reflect changes in water or sediment quality, thus are frequently used in monitoring programmes (e.g., Neckles et al. 2012).

Measurements of other seagrass parameters (e.g., shoot density, biomass and/or morphology (leaf length or width)) over time are able to detect changes in seagrass condition/quality without a change in areal extent or cover, but are less widely collected because they are generally destructive, labour intensive and often site- or patch-specific. Shanahan et al. (2023) describe seagrass health metrics to enable early detection of environmental deterioration and to increase the power and likelihood of predicting causative changes in seagrass health and condition in New Zealand. However, limited data on natural variation in, for example, leaf morphology or biomass, and a poor signal to noise ratio, mean numerical thresholds for assessing changes in seagrass health for management purposes do not appear sufficiently advanced to enable their application.

PROPOSED METRICS

The following indicator metrics are proposed for monitoring seagrass.

1. Change in extent of dominant (>50% cover) intertidal seagrass relative to the first accurate baseline.
2. Change in area weighted average percent cover (density) of intertidal seagrass with >10% cover.
3. Seagrass quality (visually observable measures).

5.1 CHANGE IN EXTENT OF DOMINANT (>50% COVER) INTERTIDAL SEAGRASS FROM FROM THE FIRST ACCURATE BASELINE

Indicator type: Primary.

Metric: Change in areal extent (ha) of dominant (>50% cover) intertidal seagrass from the first accurate baseline.

Unit of measurement: Percent (%) change in areal extent from baseline.

Spatial scale: Estuary-wide in the AIH (available intertidal habitat excluding salt marsh, and mangroves where applicable).

Applicability: All New Zealand estuarine and coastal waters where a suitable habitat type for seagrass exists, notably a sandy to muddy substratum.

Rationale: Areal extent is an efficient and cost-effective indicator of seagrass condition and is sensitive enough to detect broad spatial and temporal changes in seagrass abundance when measurements are repeated over time. It is routinely collected because it is relatively inexpensive, robust, and highly reproducible (e.g., Fourqurean et al. 2001; Krause-Jensen et al. 2004; Neckles et al. 2012). The expected seagrass extent (either ha or as a percentage of the intertidal estuary area) under different levels of degradation in New Zealand estuaries is uncertain due to natural variability and limited data (particularly from unimpacted estuaries). Therefore, substantial additional work would be required, and it may not be possible, to define thresholds based solely on seagrass extent. Consequently, thresholds based on seagrass extent are not recommended. However, temporal changes in the position and/or size of seagrass beds within an estuary can be assessed by repeat measurements and related to the effects of anthropogenic inputs of sediment or nutrients, or activities such as vegetation clearance, reclamation, or physical disturbance. A decrease in seagrass extent likely indicates a loss of ecological value and function. A change in extent can be consistently applied across different scales of data resolution, e.g., national scale remote sensing to detailed estuary-wide surveys.

Method: An accurate seagrass baseline can be obtained by a combination of broad scale mapping and extensive ground-truthing. Areal extent measurements are typically acquired from broad scale maps using established National Estuary Monitoring Protocol (NEMP) methods (e.g., Robertson et al. 2002, Stevens et al. in prep), with up-to date aerial or satellite imagery used to record seagrass features. The ability to detect change increases with increasing measurement frequency, spatial resolution and accuracy. Machine learning with automated digitisation based on the spectral analysis of imagery remains in development in New Zealand (e.g., Ha et al. 2020), with few councils currently using it as a primary method. Depending on the extent of ground-truthing and the QA/QC methods (if any) used in initial surveys, it may be necessary to update data following data QA/QC checks, e.g., to remove any overlapping or duplicated polygons, and to standardise percent cover classifications.

Field ground-truthing surveys are typically carried out on foot by experienced practitioners around the period of peak vegetative growth (e.g., October - March) to validate features visible on aerial imagery, and to derive visual estimates of seagrass, in particular percent cover. Ground-truthing maps are ideally <50cm / pixel resolution at a scale of between 1:2000 and 1:5000, as at a coarser pixel resolution and scale it becomes difficult to reliably characterise seagrass features. Annotated field maps of validated seagrass features, combined with field notes and georeferenced photographs, are digitised into shapefiles to produce maps of seagrass extent and corresponding cover.

Natural state is seldom able to be directly measured due to historical estuary modification. Historical baseline state/s may be derived from historic imagery, noting that historical changes, e.g. estuary reclamation may have occurred prior to the earliest imagery.

Assessment baseline: The first quantitative areal extent measurements obtained by a combination of broad scale mapping and extensive ground-truthing. A baseline would ideally be measured over ~3 consecutive years to quantify likely natural variability in seagrass extent.

Measurement considerations: Because seagrass beds are not uniform and will vary in density within an estuary, it is important to have a standard measure of what constitutes a 'bed' so that there is consistency in reporting. We propose this metric for seagrass extent be defined as dominant (>50% cover) intertidal seagrass in the Available Intertidal Habitat (AIH), i.e., all areas where seagrass is the dominant intertidal surface feature outside of salt marsh habitat. Percent cover (see following metric) is commonly collected when assessing seagrass. Limiting the metric to areas with >50% cover will exclude sparse beds which commonly have high variability associated with recording or when digitising features from imagery that have not been validated via ground-truthing surveys. It will enable high level classifications commonly reported in early NEMP surveys (e.g., seagrass present as a dominant surface feature) to be utilised, and is potentially well-suited to the use of remote sensing methods for data collection, where variability associated with recording sparse beds is currently uncertain but expected to be relatively high.

Seagrass cover and biomass tend to increase in summer and decrease in winter (e.g., Ramage and Schiel 1999, Turner 2007, Duncan 2017) so within-estuary measurements should be taken at the same time of each year (ideally around the peak of growth in summer, e.g., October - March) to limit the effect of seasonal changes on measurement results. Seagrass cover also has the potential to vary with natural inter-annual climatic cycles that alter wind intensity and direction, as wind-waves and currents are an important influence on seagrass patch dynamics (e.g., Turner and Schwarz 2006 and references therein). These climatic pressures may be influenced by climate change effects on wind patterns or increases in storm intensity and frequency. Relevant data on local climatic conditions (especially wind speeds and direction) will therefore be important in interpreting any changes noted in seagrass cover.

If resources are limited, measurement frequency for each estuary, or representative estuaries in a region, would ideally be determined by a risk assessment (i.e., higher frequency measurement for estuaries with higher pressures or greater risk of loss). Where a problem is identified (e.g., seagrass health appears compromised, there is evidence of significant physical damage, or seagrass is suspected to be undergoing rapid decline) measurements should be repeated annually, or at least once every 3 years. Where there is no obvious problem, measurements can be repeated every 5 years.

There are inconsistences in current New Zealand sampling methods, mapping accuracy, and classification criteria. Method accuracy could be expected to be ~±5% of the true value and reasonably precise for repeat measurements conducted by the same provider, but both could vary between providers. Development of new technologies will also require future consideration in terms of accuracy and precision.

From a management perspective, there will be difficulties separating the response of seagrass to multiple stressors, compounded by a likely non-linearity of response, and limited data on the extent of natural seagrass variability.

Statistic calculated from time series of seagrass areal extent: Percent change in areal extent (ha) of dominant (>50% cover) intertidal seagrass from the first accurate baseline:

$$\text{Percent change in areal extent of seagrass} = \frac{(\text{Baseline extent (ha)} - \text{Current extent (ha)})}{\text{Baseline extent (ha)}} \times 100$$

Potential bands and/or thresholds and rationale (including caveats): The European WFD (Foden 2007, WFD-UKTAG 2014) propose thresholds of change relative to natural state (maximum potential physical extent) which is set using expert judgement and historical data. Where no historic data or expert guidance exist, the baseline state is set based on the first set of reliable contemporary data that reflects baseline conditions.

The WFD ecological quality thresholds were defined based on a model by Krause-Jensen et al. (2003) that analysed the importance of light, wave exposure and salinity on the biomass, cover and shoot density in a large dataset crossing different geographic regions at different depth intervals (from high tide to shallow subtidal habitat) to determine changes attributable to natural variability and to anthropogenic activity.

The High (Very Good)/Good class boundary was set at ≤10% loss of seagrass extent from measured or predicted natural state conditions (maximum potential physical extent). The Good/Moderate boundary value was set at 30% loss from natural state conditions to allow for natural variability but be sensitive enough to highlight variability caused by anthropogenic activity. A loss of 70% was considered an appropriate Poor/Bad (Very Poor) boundary to reflect vulnerability of the remaining bed to possible changes in hydrodynamics or altered sediment regime. The remaining Moderate/Poor class boundary was chosen arbitrarily as the mid-point between 30% and 70%, i.e., at 50% (WFD-UKTAG 2014). The WFD thresholds are similar to those proposed for New South Wales (e.g., Roper et al. 2011), the Gulf of Mexico (e.g., Goodin et al. 2018), and for wetland loss in New Zealand (e.g., Clarkson et al., 2003). A significant limitation with the above thresholds is the need to estimate natural state. Given that seagrass is of high ecological value, and further anthropogenic losses are difficult to justify or reverse, it also seems undesirable to rate losses of up to 30% as 'Good'. Further, estimates of natural variability upon which the WFD criteria were set appear to overestimate natural variability of dominant (>50% cover) seagrass beds observed in New Zealand (Leigh Stevens, Salt Ecology, pers. obs.).

In New Zealand, more stringent thresholds for changes in contemporary seagrass extent have previously been proposed and applied as an early indicator of degradation (e.g., Robertson et al., 2016, Stevens et al., 2023). These thresholds of ecological impairment ranged from Very Low (<5% loss) to High (≥20% loss). It is considered more appropriate to adopt thresholds based on these bands in New Zealand than those of the WFD. However, it is emphasised that assessments of natural variability, measurement accuracy, and measured temporal change, should be undertaken to refine the thresholds.

Allowing for potential variability in natural extent and mapping accuracy, ≥10% loss is proposed as a precautionary threshold to trigger decisions regarding the need for further evaluation and active management (Table A5-1). Losses of ≥20% are considered likely indicative of potentially significant degradation. The higher level of stringency compared to the WFD criteria reflects that contemporary measures will likely be made over relatively short time frames, e.g., 3-5 years, and should be targeted toward early identification of potential issues. Analysis of recently measured changes in New Zealand estuaries is recommended to refine threshold breakpoints.

When interpreting this metric, it is important to consider what the scale of percent change means in absolute terms, i.e., change in ha. For example, a small percent loss in an estuary with a large seagrass extent may represent a significant area of habitat loss that is not immediately captured by the proposed percent change thresholds. Conversely, in an estuary where the starting extent is small, a large percent loss may only represent a small decrease in spatial extent and be of less relative importance. It is beyond current scope to consider this matter in greater detail, but it is important that councils and providers keep it in mind, and further work is required if thresholds are to be scaled based on estuary size or type.

A narrative of ecological quality status enables changes in areal extent to be related to ecological condition. Ecological quality status attempts to characterise the degree of change on a continuum from natural state to highly degraded to provide ecological context to current state, and measured changes from it.

Summary of proposed thresholds:

Table A5-1: Recommended thresholds for the percent loss of dominant (>50% cover) intertidal seagrass from the first accurate baseline in New Zealand estuaries.

% loss of dominant (>50% cover) seagrass from first accurate baseline	Ecological Quality Status				
	Very Good	Good	Fair	Poor	Very Poor
	≤0	>0 to <5%	≥5% to <10%	≥10% to <20%	≥20%
Narrative	Seagrass extent is stable or expanding, and maintains natural ecological function.	Seagrass extent within expected natural variation or mapping accuracy, and maintains natural ecological function.	Moderate decline in seagrass extent attributable to anthropogenic impacts. Moderate impact on ecological function.	Large decline in seagrass extent attributable to anthropogenic impacts. Potentially large impact on ecological function.	Very large decline in seagrass extent attributable to anthropogenic impacts. Large impact on ecological function.

Overall confidence in thresholds/ bands: High. Thresholds are based on expert judgement, taking into account international guidance, but require refinement based on an assessment of existing New Zealand data. Consideration should be given to developing specific thresholds applicable to short-term annual change and longer-term change, e.g., 10-year intervals. To understand the rate of change and enable comparison among estuaries, the percent change in seagrass extent could be expressed on an annualised basis by dividing by the number of years since the baseline was established. Further consideration of this approach was beyond what could be included in the current project scope.

Recommendation: Percent loss of dominant (>50% cover) Intertidal seagrass from first accurate baseline:

Adopt Table A5-1 as preliminary numeric thresholds pending analysis/review of New Zealand data.

Links to other indicators: Indicators that serve as explanatory variables for changes in seagrass extent include catchment sediment and nutrient loads, sedimentation rate, substrate, opportunistic macroalgae, water quality indicators (e.g., clarity, turbidity) and climate variables (e.g., wind, temperature etc.). The impact of anthropogenic stressors on ecosystems may be highly context specific (i.e., place and history are very important), compounded by high variability in pressures such as physical damage, changes in sea level, severe storm frequency and intensity, and temperature brought about by climate change.

Alternative metrics considered: Percent cover of intertidal seagrass (as a proxy for density), and seagrass quality, are complementary metrics which are addressed below. Inclusion of seagrass with a percent cover of <50% was considered but not adopted due to high levels of expected variance in data capture. Subtidal seagrass was not included because of high cost associated with data collection.

Additional work recommended:

i. Evaluate the consistency and accuracy of remote sensing methods for recording seagrass across a range of percent cover to determine minimum consistent data capture and reporting thresholds.
ii. Analyse within- and between-provider mapping accuracy.
iii. Collate standardised national data (and associated metadata) on seagrass extent.
iv. Analyse relationships between seagrass extent and other indicators (e.g., sediment accretion rates, nutrient concentrations, catchment land-use change) to explore links between potential drivers of change and seagrass extent. Refine thresholds as appropriate.
v. Develop standard methods to consistently define baseline conditions.
vi. Assess natural temporal variation and variation attributable to anthropogenic stressors that can be managed.

Waikawa Estuary, Southland.

5.2 CHANGE IN AREA WEIGHTED AVERAGE PERCENT COVER (DENSITY) OF INTERTIDAL SEAGRASS WITH >10% COVER FROM THE FIRST ACCURATE BASELINE

Indicator type: Supporting.

Metric: Change in area weighted average percent cover (density) of intertidal seagrass with >10% cover from the first accurate baseline.

Unit of measurement: Percent (%) change from baseline.

Spatial scale: Estuary-wide, measured within seagrass extent.

Applicability: All New Zealand estuarine and coastal waters where a suitable habitat type for seagrass exists, notably a sandy to muddy substratum.

Rationale: *Seagrass extent* (see previous metric) reflects the measured footprint of dominant (>50%) seagrass within an estuary. *Seagrass percent cover* is a measure that characterises variability in surface cover of the visually observable seagrass within this footprint, and in seagrass patches with <50% cover. *Seagrass density* is a measurement of the number of seagrass shoots in a defined area. Seagrass percent cover is commonly used as a proxy measure of seagrass density because density measurements are labour intensive to collect and cannot be reliably assessed from aerial imagery or by remote sensing methods. Density is most commonly used as a measure of seagrass health at a local scale.

Seagrass percent cover (the metric proposed here as a proxy for density) is routinely collected because it is relatively inexpensive, robust, highly reproducible (e.g., Fourqurean et al., 2001; Krause-Jensen et al. 2004; Neckles et al., 2012), and is sensitive enough to indicate broad spatial and temporal changes in seagrass density when measurements are repeated over time. Areas with high percent cover (dense beds) generally represent more stable, resilient and ecologically important areas than areas with a low percent cover (sparse beds). While the seagrass extent metric presented previously indicates changes in total area, percent cover measurements indicate changes in seagrass health before any change in bed extent occurs. It therefore offers an early warning of potential degradation.

Temporal changes in the percent cover of seagrass can be assessed by repeat measurements and related to the effects of anthropogenic inputs of sediment or nutrients, or activities such as vegetation clearance, reclamation, or physical disturbance. The metric can be consistently applied across different scales of data resolution, e.g., national scale remote sensing to detailed estuary-wide surveys.

Method: Percent cover measurements are typically classified from visual assessments collected during field surveys (see table adjacent), aided by the use of visual guides or stratified random quadrat sampling using gridded quadrats to assess cover. Post-field measurements of photo-quadrats using, for example, Coral Point Count (CPC) software (100-point grid overlay) can also be used to quantify seagrass cover. Data are commonly acquired from broad scale surveys undertaken using established National Estuary Monitoring Protocol (NEMP) methods (e.g., Robertson et al. 2002, Stevens et al., in prep), with up-to date aerial or satellite imagery used to record seagrass features. The ability to detect change increases with increasing measurement frequency, spatial resolution and accuracy. Machine learning with automated digitisation based on the spectral analysis of imagery remains in development in New Zealand (e.g., Ha et al. 2020), with few councils currently using it as a primary method.

Recommended percent cover classification classes.

Class	Coarse category	Fine category
Absent or trace	<1%	<1%
Very sparse	1 to <10%	1 to <10%
Sparse	10 to <30%	10 to <20%
		20 to <30%
Low-Moderate	30 to <50%	30 to <40%
		40 to <50%
High-Moderate	50 to <70%	50 to <60%
		60 to <70%
Dense	70 to <90%	70 to <80%
		80 to <90%
Complete (≥90%)	≥90%)	≥90%

Field ground-truthing surveys are typically carried out on foot by experienced practitioners around the period of peak vegetative growth (e.g., October - March) to validate features visible on aerial imagery, and to derive visual estimates of seagrass. Ground-truthing maps are ideally <50cm / pixel resolution at a scale of between 1:2000 and 1:5000, as at a coarser pixel resolution and scale it becomes difficult to reliably characterise seagrass features. Annotated field maps of validated seagrass features, combined with field notes and georeferenced photographs, are digitised into shapefiles to produce maps of seagrass extent and corresponding cover.

Assessment baseline: The first quantitative percent cover measurements obtained by a combination of broad scale mapping and extensive ground-truthing. A baseline would ideally be measured over ~3 consecutive years to quantify likely natural variability in seagrass percent cover.

Measurement considerations: The management objective is for seagrass cover to increase or remain at the maximum potential for the site, with the expectation that percent cover will decrease if there is ecological deterioration in the water body (WFD-UKTAG 2014). The metric is best suited to seagrass in the Available Intertidal Habitat (AIH). Field observations indicate low percent cover areas are often associated with high temporal variability, particularly when growing in mobile substrate. Further, there is expected to be higher variability associated with recording sparse beds using remote sensing methods, or when digitising features from imagery that has not been validated via ground-truthing surveys, than compared to higher density beds (pers obs. Leigh Stevens, Salt Ecology). It is therefore recommended that changes in seagrass percent cover are calculated from areas where cover is ≥10%.

Seagrass cover and biomass tends to increase in summer and decrease in winter (e.g., Ramage and Schiel 1999, Turner 2007, Duncan 2017) so within-estuary measurements should be taken at the same time of each year (ideally around the peak of growth in summer, e.g., October - March) to limit the effect of seasonal changes on measurement results. Seagrass cover also has the potential to vary with natural inter-annual climatic cycles that alter wind intensity and direction, as wind-waves and currents are an important influence on seagrass patch dynamics (e.g., Turner and Schwarz 2006 and references therein). These pressures may be influenced by climate change effects on wind patterns or increases in storm intensity and frequency. Relevant data on local climatic conditions (especially wind speeds and direction) will therefore be important in interpreting any changes noted in seagrass cover.

If resources are limited, measurement frequency for each estuary, or representative estuaries in a region, would ideally be determined by a risk assessment (i.e., higher frequency measurement for estuaries with higher pressures or greater risk of loss). Where a problem is identified (e.g., seagrass health appears compromised, there is evidence of significant physical damage, or seagrass is suspected to be undergoing rapid decline) measurements should be repeated annually, or at least once every 3 years. Where there is no obvious problem, measurements can be repeated every 5 years.

There are inconsistences in current New Zealand sampling methods, mapping accuracy, and classification criteria. Method accuracy could be expected to be ~±10% of the true value and reasonably precise for repeat measurements conducted by the same provider, but both could vary between providers. Development of new technologies will also require further consideration in terms of accuracy and precision.

From a management perspective, there will be difficulties separating the response of seagrass to multiple stressors, compounded by a likely non-linearity of response, and limited data on the extent of natural seagrass variability.

Natural state is seldom able to be directly measured due to historical estuary modification. Historical baseline state/s may be derived from historic imagery, noting that historical changes, e.g. estuary reclamation may have occurred prior to the earliest imagery, and determining percent cover from imagery in the absence of ground-truthing is imprecise.

Statistic calculated from time series of weighted average seagrass percent cover: Percent change from a baseline in the area weighted average cover of intertidal seagrass with >10% cover:

To calculate the weighted average of seagrass cover, for each time series (i.e., field survey) multiply each percent cover classification (x) by its area (w) to derive its product, then sum all the products and divide by total area.

$$\bar{x} = \frac{\sum w_i x_i}{\sum w_i}$$

To compare surveys (with respect to a baseline) determine percent change in the weighted average of >10% intertidal seagrass cover from a defined baseline as follows:

$$\text{Percent change in mean cover of seagrass} = \frac{(\text{Baseline mean percent cover - Current measured mean percent cover})}{(\text{Baseline mean percent cover})} \times 100$$

The following table presents a worked example of calculations comparing a baseline survey to a repeat survey (the same approach can be used to compare subsequent repeat surveys with each other).

Baseline survey					Repeat survey				
%		ha		Product	%		ha		Product
10	x	0	=	0	10	x	44	=	440
20	x	0	=	0	20	x	42	=	840
30	x	0	=	0	30	x	0	=	0
40	x	0	=	0	40	x	45	=	1800
50	x	0	=	0	50	x	0	=	0
60	x	80	=	4800	60	x	17	=	1014
70	x	0	=	0	70	x	0	=	0
80	x	0	=	0	80	x	15	=	1168
90	x	822	=	73980	90	x	0	=	0
100	x	0	=	0	100	x	1.5	=	150
		902		78780			164		5412
Weighted mean % cover					Weighted mean % cover				
78780/902=87					5412/164=33				

% change between surveys: ((33-87)/87)*100=62

Potential bands and/or thresholds and rationale (including caveats): The most comprehensive thresholds of change relative to a baseline appear to be those of the WFD (Foden 2007, WFD-UKTAG 2014). The baseline is set as either estimated natural state (maximum potential cover) or the first set of reliable contemporary data that reflects representative conditions, and the greatest cover recorded in the first cycle of monitoring.

Ecological quality thresholds for percent cover were defined based on a model by Krause-Jensen et al. (2003) that analysed the importance of light, wave exposure and salinity on the biomass, cover and shoot density of a large dataset crossing different geographic regions at different depth intervals (from high tide to shallow subtidal) to determine changes attributable to natural variability and to anthropogenic activity.

The High (Very Good)/Good class boundary was set at ≤10% loss of seagrass percent cover. The Good/Moderate boundary value was set at 30% loss to allow for natural variability but be sensitive enough to highlight variability caused by anthropogenic activity. A loss of 70% was considered an appropriate Poor/Bad (Very Poor) boundary to reflect vulnerability of the remaining bed to possible changes in hydrodynamics or altered sediment regime. The remaining Moderate/Poor class boundary was chosen arbitrarily as the mid-point between 30% and 70%, i.e. at 50% (WFD-UKTAG 2014).

When sequential yearly percent cover survey data exist, calculation of a 5-year rolling mean considerably reduces noise in this metric, and underlying trends should become more apparent (allowing more stringent ecological quality thresholds to be applied). The % cover metric rolling mean is an average of that year and the previous four years' measurements (WFD-UKTAG 2014).

Similar to comments in Section 5.1 for changes in spatial extent, given that seagrass is of high ecological value, and further anthropogenic losses are difficult to justify or reverse, it seems undesirable to rate reductions in seagrass density (measured as percent cover) of up to 30% as 'Good'. Similarly, losses of >50% (Poor) or >70% (Very poor) seem very permissive. Further, estimates of natural variability upon which the WFD criteria were set appear to overestimate natural variability of seagrass beds observed in New Zealand (Leigh Stevens, Salt Ecology, pers. obs.). Therefore, while the WFD thresholds are potentially appropriate for assessing change relative to historical extent, more stringent criteria appear appropriate for comparison with recently measured baselines. These thresholds are proposed as an initial starting point but require refinement using New Zealand data.

Like for the areal extent metric, defining ecological quality status for percent cover enables better characterisation of anthropogenic impacts. Ecological quality status attempts to characterise the degree of change on a continuum from natural state to highly degraded to provide ecological context to current state, as well as any measured changes from it.

Summary of proposed thresholds:

Table A5-2. Recommended interim thresholds for change in the area weighted average percent cover of intertidal seagrass with >10% cover compared to the first accurate baseline.

% reduction in area weighted average percent cover from first accurate baseline	Ecological Quality Status				
	Very Good	Good	Fair	Poor	Very Poor
Mean annual cover	>0 to ≤10%	>10% to ≤30%	>30% to ≤50%	>50 % to ≤70%	>70% loss
5 year rolling mean	>0 to ≤5%	>5% to ≤15%	>15% to ≤25%	>25 % to ≤35%	>35% loss
Narrative	Seagrass beds reflect natural state, and are stable or expanding.	Seagrass beds are within expected natural variation from natural state.	Impacts from anthropogenic activity cause measurable decline in seagrass cover.	Impacts from anthropogenic activity cause significant decline in seagrass cover.	Impacts from anthropogenic activity significantly compromise seagrass integrity.

Overall confidence in thresholds/ bands: Fair. These thresholds require refinement using New Zealand data. In particular, thresholds based on international criteria appear permissive, and while potentially appropriate for assessing change relative to historical extent, more stringent criteria appear appropriate for comparison with recently measured baselines.

Recommendation: Area weighted average percent cover (density) of intertidal seagrass with >10% cover:

Adopt Table A5-2 as guidance only thresholds pending analysis/review of New Zealand data.

Links to other indicators: Indicators that serve as explanatory variables for changes in seagrass cover include catchment sediment and nutrient loads, sedimentation rate, substrate, opportunistic macroalgae, water quality indicators (e.g., clarity, turbidity) and climate variables (e.g., wind, temperature etc.). Seagrass health (see following section and also Zabarte-Maeztu 2021), is also important. The impact of anthropogenic stressors on ecosystems may be highly context specific (i.e., place and history are very important), compounded by high variability in pressures such as physical damage, changes in sea level, severe storm frequency and intensity, and temperature brought about by climate change.

Alternative metrics considered: Seagrass density (shoot numbers within a defined area) measures are potentially a more accurate measure of seagrass cover, but have not been recommended for broad scale monitoring due to the high level of sampling effort required to collect data. Seagrass density is suitable for inclusion in targeted studies of seagrass quality, which are commonly undertaken at a site-specific, rather than an estuary-wide, scale. Areal extent of intertidal seagrass and seagrass quality are complementary metrics for seagrass. Seagrass quality is addressed below.

Additional work recommended:

i. Collate standardised national data (and associated metadata) on seagrass cover.
ii. Analyse relationships between seagrass cover and other indicators (e.g., sediment accretion rates, nutrient concentrations, catchment land-use change) to explore links between potential drivers of change and seagrass cover. Refine thresholds as appropriate.
iii. Develop standard methods to consistently define baseline conditions.
iv. Assess likely temporal variation attributable to anthropogenic and natural stressors.
v. Analyse within and between provider mapping accuracy.

5.3 SEAGRASS QUALITY

Visually observable impacts related to nutrient enrichment (e.g., nuisance epiphyte or macroalgal cover, fine sediment smothering, presence of fungal wasting disease).

Indicator type: Supporting.

Metric: Mean percent cover of total seagrass extent impacted by nuisance epiphyte or macroalgal cover, or fine sediment smothering.

Unit of measurement: Percent cover.

Spatial scale: Estuary-wide or site-specific.

Applicability: All New Zealand estuarine and coastal waters where a suitable habitat type for seagrass exists, notably a sandy to muddy substratum.

Rationale: Shanahan et al. (2023) summarise a range of seagrass health indicators for use in environmental management in New Zealand, and which provide early warning indicators of seagrass stress from a range of environmental and anthropogenic pressures. Indicators include direct seagrass measures, e.g., shoot density, biomass, morphology (leaf length or width), flowering, leaf nitrogen and carbon content; indirect indicators of stress, e.g., presence of fungal wasting disease, epiphyte and macroalgal cover; and indirect measures of conditions that may affect seagrass growth, e.g., light environment, water temperature, sediment characteristics. Most of these indicators respond to a variety of stressors and hence the signal-to-noise ratio can be poor or difficult to elucidate. Biomass data are difficult to interpret with limited power to detect change without very large sample size (Sutula 2011). Further, many methods are generally destructive, labour intensive and often site- or patch-specific, have limited data on natural variation, particularly in New Zealand (for example, leaf morphology or biomass), with numerical thresholds uncommon or uncertain regarding their possible application for use in New Zealand. Any attempt to develop numerical thresholds relevant to estuary management in New Zealand would require substantial effort and may ultimately prove unsuccessful.

Of the various quality indicators listed above, nuisance epiphyte or macroalgal cover, fine sediment smothering (see Hale et al. 2024), or presence of fungal wasting disease have potential to be used by experts in a rapid broad-scale visual field assessment of seagrass condition related primarily to nutrient-driven eutrophication or fine sediment smothering. It is possible that narrative thresholds could be developed as an initial screen for the potential presence of seagrass stressors to guide decisions on the merit of further evaluation, although this has not yet been developed.

Methods: See Shanahan et al. (2023) for a description of general methods for specific indicators relating to the presence of fungal wasting disease, epiphyte cover, macroalgal cover, and the NEMP for sediment classification (estimated sediment mud content).

Measurement considerations: Field appraisal should be carried out by expert practitioners experienced in assessing seagrass quality and would ideally require the development of visual guides for classifying different states of visually observable degradation.

Statistic: None proposed.

Potential bands and/or thresholds and rationale (including caveats): Numerical thresholds are not proposed for seagrass quality indicators due to limited data availability and expected poor signal-to-noise ratio.

High level narrative thresholds have potential for development as preliminary screening criteria to help determine if more detailed investigation is warranted. Potential narrative thresholds are included below but require additional research to validate them.

Summary of proposed thresholds:

Table A5-3. Potential narrative criteria for the rapid visual assessment of seagrass condition in the available intertidal habitat (AIH) of New Zealand estuaries.

Seagrass quality	Ecological Quality Status				
	Very Good	Good	Fair	Poor	Very Poor
Narrative	Seagrass beds reflect natural state. Seagrass wasting disease, nuisance epiphytic growth, and nuisance macroalgae absent or negligible.	Seagrass beds are within expected natural variation from natural state. <10% cover of seagrass wasting disease, nuisance epiphytic growth, and nuisance macroalgae.	≥10-25% cover of seagrass wasting disease, nuisance epiphytic growth, and nuisance macroalgae. <25% cover of fine sediment on seagrass leaves.	≥25-50% cover of seagrass wasting disease, nuisance epiphytic growth, and/or nuisance macroalgae. ≥25-50% cover of fine sediment on seagrass leaves.	≥50% cover of seagrass wasting disease, nuisance epiphytic growth, and/or nuisance macroalgae. ≥50% cover of fine sediment on seagrass leaves.

Overall confidence in thresholds/ bands: Low. While there is ample literature available describing impacts of fine sediment and macroalgal/epiphytic growth on seagrass (including in NZ: e.g., Zabarte-Maeztu 2021), measuring these indicators in the field will require expert judgement, are difficult to make quantitatively, have not been validated, and are likely to be variable between observers. However, they offer initial high-level guidance on the condition of seagrass as a possible early warning of changes in seagrass extent or cover and possible drivers of change.

Recommendation: Seagrass quality

Further consideration of numeric thresholds not recommended. Substantial further investigative work required to refine narrative thresholds.

Links to other indicators: Other indicators that serve as explanatory variables for changes in seagrass extent include catchment sediment and nutrient loads, sedimentation rate, substrate, opportunistic macroalgae, water quality indicators (e.g., clarity, turbidity) and climate variables (e.g., wind, temperature etc.). The impact of anthropogenic stressors on ecosystems may be highly context specific (i.e., place and history are very important), compounded by high variability in pressures such as physical damage, changes in sea level, severe storm frequency and intensity, and temperature brought about by climate change.

Additional work recommended:

i. Collate standardised national data (and associated metadata) on seagrass quality measures to enable potential patterns in seagrass quality response to be identified.
ii. Develop visual guides for classifying different states of visually observable degradation related to nuisance epiphyte or macroalgal cover and fine sediment smothering.
iii. Refine potential narrative thresholds based on field data to determine whether a general screening metric can be developed.

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APPENDIX 6. SHELLFISH

Authors: Drew Lohrer (NIWA, Hamilton), Carolyn Lundquist (NIWA, Hamilton) – Shellfish bed extent

Authors: Barrie Forrest (Salt Ecology) – Shellfish quality

Although there are hundreds of species of bivalve molluscs in New Zealand, the term 'shellfish beds' is generally applied to 8-10 species of large, common, and well-known species including: cockles, dog cockles, pipi, wedge shells, oysters, green-lipped mussels, horse mussels, and scallops. Some bed-forming shellfish live on the sediment surface (e.g., green-lipped mussels, oysters, scallops), whilst others live deeper in the sediment (e.g., pipi, cockles, wedge shells). Most of the bed-formers are recognised as kaimoana or as ecologically important 'key' species.

BACKGROUND

Shellfish are a key indicator of ecological integrity in intertidal and shallow subtidal coastal and estuarine systems (Norkko et al. 2006; Thrush et al. 2006; Lohrer et al. 2010; Lohrer et al. 2013; Thrush et al. 2013). The denser and more extensive the shellfish beds are (bed extent), and the healthier the shellfish are within them (bed quality), the greater the ecosystem's ecological integrity. Different shellfish perform different ecological roles, therefore, having a diversity of shellfish types (e.g., cockle and wedge shell beds on intertidal flats; pipi beds in estuarine tidal channels; green-lipped mussels, horse mussels, and dog cockles in deeper areas) is also integral to ecological integrity.

One of the most recognisable indicators of degradation of marine ecosystems has been the collapse of natural shellfish populations throughout New Zealand. Extensive green-lipped mussel beds (*Perna canaliculus*), covering an estimated 500km^2 of seafloor habitat in the Hauraki Gulf, were decimated by a bottom-contact dredge fishery (1910-1960), and an additional ~100km^2 were lost from the Marlborough Sounds (Greenway 1969; Anderson et al. 2019; Hauraki Gulf Forum 2020; Urlich & Handley 2020; Toone et al. 2021; Toone et al. 2023). High density horse mussel beds (*Atrina zelandica*) have almost completely disappeared from places where they were once common (Hauraki Gulf, Coromandel Peninsula, Marlborough Sounds), with just relict beds remaining (Norkko et al. 2006; Lohrer et al. 2010; Lohrer et al. 2013). Lucrative scallop fisheries have crashed nationwide, and populations have not rebounded despite harvesting bans including both rāhui and national-scale MPI fisheries closures (Hauraki Gulf Forum 2020). Pipi beds (*Paphies australis*) at the mouth of Whangārei Harbour covered 0.5% of the area in 2017 that they covered in 2005 (Williams et al. 2017; Patuharakeke Te Iwi Trust Board 2020), a ~10,000 tonne collapse in a little more than a decade. Hundreds of hectares of former shellfish habitat in Southland estuaries are now smothered under nuisance macroalgal mats (Plew et al. 2020; Stevens et al. 2022).

Shellfish kaimoana on tidal flats adjacent to large cities are exposed to landfill leachate and sewage effluent, a potential threat to people collecting and eating them. Even in rural areas, leaky septic systems and poor water/sediment quality (e.g., from upstream agriculture) can affect the fitness of shellfish for human consumption. Habitat quality of shellfish beds has also been impacted by bottom-contact fishing and terrigenous sediment inputs, which have resulted in muddy seafloor sediments with insufficient biogenic structure (Hale et al. 2024).

Cultural practices surrounding the collection of shellfish kaimoana have been handed down through generations and declines in shellfish bed extent and quality are well known and deeply concerning to Māori whose identity and wellbeing have relied upon connections to shellfish and mahinga kai for generations. Declines in shellfish bed extent and quality also affect recreational and commercial fishers, and any who appreciate the roles shellfish play in coastal ecosystems. Shellfish provide jobs and business opportunities for many New Zealanders including Māori (e.g., mussel and oyster aquaculture; scallop fisheries).

Without management interventions (e.g., restricting bottom contact fishing, reducing catchment sediment input, improving water quality), the prospects for shellfish recovery are poor. Climate change and increased frequency/intensity of storms over the next 10-30 years is predicted to increase sediment loading and sediment resuspension in estuarine and coastal areas (Herzig et al. 2024), potentially limiting recovery prospects further.

PROPOSED METRICS

The following indicator metrics are proposed for monitoring shellfish bed extent and quality.

1. Shellfish bed extent - section authors Drew Lohrer and Carolyn Lundquist
2. Shellfish bed quality - section author Barrie Forrest

6.1 SHELLFISH BED EXTENT

Indicator type: Supporting.

Metric: Change (ha, %) in areal extent of shellfish beds.

Unit of measurement: Hectares (ha).

Spatial scale: Whole of estuary scale.

Applicability: Any estuary nationwide that historically had shellfish beds.

Rationale: As evidenced by broad-scale historical declines, shellfish bed extent is sensitive to a diversity of impacts, including sedimentation, nutrients, pollutants, and overharvesting. Shellfish declines have been observed to occur rapidly, such that timeframes of 5-year intervals may be suitable to quantify changes in broad-scale extent, but shorter timeframes (1-2 yearly) should be explored for areas with rapid increases in local stressors.

Although this is a potentially important metric, defining the areal extent of a shellfish bed is difficult and often subjective. Conceptually, a shellfish bed is an area where shellfish are abundant/dense enough to be the defining feature of the habitat. For some species, quantifying bed extent can be relatively simple (e.g., intertidal oyster reefs, which form visual clusters that can be observed and mapped using aerial photos or drone imagery). For other species (i.e., those that are infaunal and whose densities are not revealed without digging through the sediment, or subtidal species that are generally surveyed using transect lines) it can be very challenging. Moreover, the density that is 'enough' to qualify as a shellfish bed will vary by species and is generally operationally defined (e.g., it can be applied to adults of harvestable size, to total individuals, or other). Although there are exceptions, very dense beds of large shellfish are generally considered to be 'healthy', whilst dwindling numbers, or the presence of small size classes only, are considered signs of poorer health.

Intertidal shellfish beds are primarily infaunal (with the exception of oysters) and are difficult to detect in aerial photographic surveys without ground-truthing (sediment excavation). Subtidal shellfish beds are typically epifaunal (e.g., green-lipped mussels, scallops), but aerial remote sensing does not penetrate through water, thus *in situ* subtidal surveys are generally required (e.g., using divers, towed cameras, or remotely operated cameras). It is extremely difficult to get broad enough coverage with underwater survey techniques to define the boundaries of subtidal shellfish beds - thus beds are often defined by abundance/density at sites or along transects.

Method: We propose a method based on the following steps:

(1) Start with readily identifiable intertidal/shallow shellfish kaimoana species only, e.g., cockles, wedge shells, pipi, mussels, oysters;

(2) Consult widely to come to an agreed operational threshold for what constitutes a high-density bed of each species;

(3) Use 'rapid habitat mapping' (sensu Lam-Gordillo et al. 2023, 2024; Needham et al. 2013) and other complementary sampling (coring, quadrats) and data (sediment type; sediment elevation above/below chart datum; tidal current flow speeds) where available to define GIS polygons of high-density shellfish habitats ("HD cockles", "HD pipi", etc.);

(4) Repeat the 'rapid habitat mapping' sampling periodically to determine expansion/contraction/change in the areal extent of these habitats; and

(5) Reconstruct historical shellfish bed extent using oral histories from mana whenua and any other available methodologies, such as long coring for analyses of shellhash.

An example to be followed is that presented in Lam-Gordillo et al. (2024 - Not in the public domain as at the time this report was completed, contact WRC for details). The method involves systematic coverage of estuarine intertidal area on foot with regular spot checking to define the spatial extent of habitats and assign them to pre-defined categories (e.g., 'High Density pipi habitat' = areas with >10 pipi sized >40 mm shell length in a 15 x 15 cm square quadrat; Needham et al. 2013; Lam-Gordillo et al. 2023). In the Waikato Region, the same fourteen estuaries were mapped 10 years apart, and changes in shellfish bed extent in the intervening period were able to be assessed in each (Lam-Gordillo et al. 2024). It is also possible to supplement and integrate the rapid habitat mapping with complementary sampling/surveying. For example, infaunal bivalves including cockles, wedge shells, and pipi are monitored at sentinel monitoring sites by many councils using standard sized cores (Hailes and 2009, Drylie 2021). This produces highly standardised data on bivalve abundance and size structure (often in classes, e.g., 0-5 mm, 5-10, 10-15, 15-20, 20-30, 30-40, >40mm). An Estuarine Toolkit published by NIWA (in English and te reo Māori) provides guidance on standard shellfish monitoring methods for intertidal shellfish (cockle, wedge shells, juvenile pipi (Swales et al. 2011). Most councils have started reporting estuarine monitoring data on the Land, Air, Water Aotearoa (LAWA) website (https://www.lawa.org.nz/explore-data/estuaries). MPI has funded surveys of cockles and pipi in many harbours and estuaries, which are generally designed to characterise both abundance and distribution of shellfish across the seascape (Williams et al. 2007; Berkenbusch et al. 2022). Some iwi groups have mapped cockle, pipi and green-lipped mussel beds using quantitative (usually quadrat-based) techniques (Paul-Burke et al. 2018).

Cockles, pipi, and mussels are monitored by local kaitiaki in many parts of New Zealand. This includes the monitoring of cockles, pipi, and mussels by Patuharakeke Te Iwi Trust on intertidal banks in outer Whangārei Harbour (Snake Bank, Mair/Marsden Bank) (Williams et al. 2017), the monitoring of cockles by Ngāti Whakehemo in intertidal soft-sediment habitats of Waihī Estuary, and the monitoring of subtidal mussel populations and beds by Ngāti Awa and the Te Ūpokorehe Resource Management Team in Ōhiwa harbour (Paul-Burke et al. 2018). Ngāti Awa has also collected information on scallop, horse mussel, pipi and cockle populations in Ōhiwa Harbour.

For subtidal species like green-lipped mussels and horse mussels, scuba transects, and underwater towed video transects may be used to quantify abundance. Auckland Council diver surveys of horse mussel abundance/size using transects and quadrats in Mahurangi Harbour were abandoned after densities dropped to the point where this type of survey technique was no longer affordable/practical. Diver and towed video surveys generally do not quantify shellfish bed extent (i.e., they only quantify shellfish density and size at specific sites). Observations of shellfish (e.g., size, degree of fouling or sediment smothering) and the number of live vs dead, may provide information on 'bed quality'.

Scallop beds have been surveyed for many years by MPI using standard benthic trawling techniques (Williams et al. 2019). Because of the destructiveness of the technique, methods are being developed to transition towards underwater towed camera surveys. Transitioning to camera-based surveys may also increase the availability of useful ancillary information on the appearance/condition of the habitat.

Measurement Considerations: The metric is best suited for intertidal flat and estuary-scale estimates of shellfish bed extent.

Calculation of statistic: Change (ha, %) in areal extent of shellfish beds.

Change (in hectares and as a percentage) can be calculated in at least two ways:

(1) Since last surveyed (or over time, given multiple survey dates). This provides information on whether shellfish bed extent is increasing, decreasing, or unchanged.
(2) Relative to estimated historical extent. This provides information on the degree of recovery relative to a healthier standard.

Metrics should be determined on a per estuary basis, noting that not all shellfish species may have been naturally present in all estuaries.

Potential bands and/or thresholds and rationale (including caveats): Preliminary thresholds have been proposed in Table A6-1 for change from estimated historical extent based on expert judgement, with the caveat that there is little justification or rationale for placing the band thresholds where they are. Although an estuary with >90% of its historical shellfish bed extent remaining is likely functioning better than one with 10% of its historical shellfish, there are no data describing whether an estuary with 60% of its historical shellfish is functioning at a 'Good', 'Moderate' or 'Poor' level. The bands are essentially arbitrary. Percent value (used as a continuous variable) is likely to be as informative as any banding scheme.

It is unclear how to merge or integrate information on the extent of multiple different shellfish species, each of which contributes to estuarine health and functioning differently. The loss of one key shellfish bed type may be enough to fundamentally alter an estuary, even if several of the other bed-forming shellfish species are present at near historical baseline levels.

Other considerations: Some shellfish species may exist in relatively distinct and definable beds (e.g., pipi and cockle shellbanks in outer Whangārei Harbour), however, the beds of other types of species (e.g., scallops) may be much harder to delineate and define. Measuring change in shellfish bed extent would be much easier for the former than for the latter.

Historical information on 'baseline' shellfish bed extent is typically based on anecdotal or qualitative reflections on historical shellfish beds or kaimoana collection sites. Reference states would be estuary and species-specific. For some commercial fisheries species (e.g., green-lipped mussels, scallops, pipis, cockles), stock assessments and regular stock surveys can provide recent historical backgrounds of trends in extent and density. Although maps showing the purported extent of green-lipped mussel coverage in outer Tamaki Strait/ Hauraki Gulf from the early 1900s are available, information on the natural reference state of cryptic non-harvested species like dog cockles is almost entirely lacking. It is likely, however, that natural reference states of all bed-forming shellfish species were likely better than today's degraded state.

Summary of proposed thresholds:

Table A6-1: Recommended thresholds for current shellfish bed extent (%) relative to estimated historical extent.

Percent reduction from estimated historical shellfish bed extent	Ecological Quality Status				
	Very Good	Good	Fair	Poor	Very Poor
	<0 to <10%	≥10% to <20%	≥20% to <50%	≥50 % to <75%	≥75% loss

Overall confidence in thresholds/ bands: Low.

Recommendation: Work on achieving Methods steps 1-3 as soon as possible for as many estuaries as possible. This will generate valuable information on shellfish bed extent, whether or not bands are ultimately adopted.

Links to other indicators: Other indicators that are linked to changes in shellfish bed extent include those reflecting catchment sediment loads and sedimentation rate (sediment accretion rate) and indicators of nutrient impacts (organic matter, nutrients in coastal waters, RPD). Bed-forming shellfish are also likely to be positively correlated with phytoplankton as this is a food source for sessile benthic filter-feeding bivalves. Macroinvertebrate community composition is also linked, and may provide information on the densities and sizes of some bed-forming shellfish species (e.g., cockles, wedge shells, pips) but it will not necessarily correlate with Shellfish Bed Extent and Quality.

Alternative metrics considered: No alternative metrics are suggested.

Additional work recommended:

i. Come to agreement and disseminate agreed operational definitions of 'high-density bed' for a set of readily identifiable estuarine shellfish species.
ii. Utilise existing published methods to rapidly map shellfish bed extent in estuaries.
iii. Define estuary-specific historical baselines for shellfish bed extent.
iv. Develop thresholds for percent change in shellfish bed extent from a recently measured baseline.
v. Consider the merit of expressing percent change in shellfish bed extent on an annualised basis (dividing by the number of years since the baseline was established) to enable standardised comparison among estuaries.

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6.2 SHELLFISH HEALTH

Author: Barrie Forrest (Salt Ecology)

BACKGROUND

Shellfish health in the current context refers to the ecological condition of shellfish, rather than attributes relevant to human consumption (e.g., Shumway & Rodrick 2009). Shellfish health was flagged at project contracting stage as an indicator that would likely require further development from a band-setting perspective.

Adequately addressing shellfish health indicator thresholds at the individual level is a significant undertaking. Potential indicators of shellfish health are numerous and derive from many different disciplines, with each requiring specialist expertise to evaluate and understand in the context of threshold development. At a high-level, examples of potential individual-based indicators are as follows:

- Field and lab-based measures of shellfish health, which can include: subjective visual grading of gonads; lab-based gonad histology; lab analysis of glycogen and other indicators; and morphometric, weight or volume-based measures of shells or flesh (Hickman & Illingworth 1980; Buchanan 2001; Williams & Babcock 2004; Fletcher et al. 2013; O'Connell-Milne et al. 2016).
- Functional approaches to assess stress in bivalves, which include valve opening (gape) or closure tests, and tests of byssus production and substratum attachment (e.g., Forrest & Blakemore 2006; Webb & Heasman 2006; Kelleghan et al. 2023).
- Laboratory analyses for sub-lethal markers of stress (e.g., caused by contaminants), for example, based on enzymes, oxidative stress markers, endocrine, physiological, immunological, and energetic approaches (Chahouri et al. 2023, and references therein).
- Analyses based on pathology, histology, molecular and other methods to establish the occurrence and effects (in individual shellfish) of disease agents (i.e., pathogens and parasites) or biotoxins from harmful phytoplankton (Elston & Ford 2011; Rhodes et al. 2013; Lane et al. 2016; Castinel et al. 2019; Webb and Duncan 2019; Rolton et al. 2022).

Except for causal associations between the occurrence of harmful phytoplankton or disease-agents, and shellfish health at the individual or population level, the other potential groups of indicators listed above have responses that are mainly too generic to be of use for routine monitoring. For example, shellfish health in estuaries will respond to a wide range of factors (e.g., geographic location, season, tidal elevation, water temperature, salinity, food availability, parasite load, reproductive stage), meaning that ascribing changes detected by SOE monitoring to anthropogenic influences would be challenging (Marsden & Pilkington 1995; O'Connell-Milne et al. 2016).

It is also noted that indicators relating to diseases, harmful phytoplankton and contaminant accumulation have non-ecological implications (mainly human health considerations) and would be addressed by the Ministry for Primary Industries. For example, the first course of action on finding visual evidence of dead or dying shellfish would likely be a response by MPI to consider whether disease was a primary cause.

Because of the range of factors above, further consideration of health-based indicators and thresholds at the individual shellfish level is not recommended.

PROPOSED METRICS

No specific metrics are proposed for shellfish quality.

Indicator type: Supporting.

Metric: Not determined. Depends on indicator.

Unit of measurement: Not determined. Depends on indicator.

Spatial scale: Specific to individual shellfish.

Applicability: Any estuary with important shellfish populations. However, the applicability and relevance of shellfish health indicators is likely to vary within, and among estuaries, and regionally, depending (among other things) on the shellfish species present, their population characteristics, and the extent to which their environment is subject to stressors. As well as stressors such as muddy sediment inputs and other environment factors that stress individual shellfish, in the broadest sense stressors also encompass biological agents including harmful phytoplankton, pathogens and parasites.

Rationale: No specific metrics are proposed for shellfish quality. A more comprehensive consideration of potential indicators would be needed to understand whether any had merit for further development. However, we suggest that any further development is best progressed as part of long-term and well-funded MBIE-type research.

Method: None proposed as this stage.

Measurement considerations: No measures proposed as this stage.

Calculation of statistic: Not applicable.

Potential bands and/or thresholds and rationale (including caveats): Not applicable.

Summary of proposed thresholds: No thresholds are being proposed.

Overall confidence in thresholds/ bands: Low. There are some studies/data, but large spatial and temporal variability make banding into thresholds inappropriate.

Recommendation: Shellfish Quality (Health)

No further consideration of numeric and/or narrative thresholds recommended.

Links to other indicators: Not applicable.

Alternative metrics considered: Shellfish population extent and standard population monitoring metrics (i.e., biomass, size-frequency distribution, recruitment) are alternative metrics that could be considered as primary indicators for shellfish health. However, development of population-based thresholds is not envisaged at this stage, with the only proposed indicator (in preceding Section 6.1) being the percent reduction from estimated historical shellfish bed extent.

Additional work recommended: No additional work is recommended at this stage.

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APPENDIX 7. SEDIMENTATION RATE

Authors: Steph Mangan (NIWA, Christchurch), Orlando Lam-Gordillo (NIWA, Hamilton) and Drew Lohrer (NIWA, Hamilton)

Sediment accretion rate refers to the change in level of the surface of the bed sediment relative to a fixed datum over a characteristic timescale of months to years, as a consequence of sedimentation. While sedimentation is a natural process, extensive land-use changes have increased the loading of fine sediments which can significantly alter the hydrodynamic, geomorphology, and ecological characteristics of the receiving system impacting its ecological health.

BACKGROUND

Eroded soil (terrigenous sediment) is considered to be a major contaminant affecting New Zealand's estuaries, with an estimated 192 million tonnes of soil being lost from the land each year (Ministry for the Environment & Stats NZ 2018). Sedimentation after heavy rainfall can adversely affect estuarine ecosystems by altering microbial activity, diminishing benthic primary productivity, and reducing oxygenation of surface sediment (Berkenbusch et al. 2002; Lohrer et al. 2004). As sediment accumulates over time, it can result in changes to the cohesiveness of the surface sediment, inhibit diffusive and advective transport of solutes resulting in changes to porewater geochemistry, alter food quality through nutritional differences of terrestrial and marine sediments, block filter-feeding appendages, and deter larval settlement (Ellis et al. 2002; Marinelli & Woodin 2002; Cummings et al. 2003), all of which have significant ecosystem implications.

Soil erosion estimates within New Zealand are higher than for other parts of the world due to steep terrain, weathered and friable rock, high rainfall and the frequent occurrence of high-intensity rainstorms (Basher 2013). Historic catchment deforestation, large-scale conversions from native forest to pastoral agriculture, land-use intensification and catchment disturbance have all contributed to current sediment accretion rates (SAR) being 10 to 100 times higher than during pre-human settlement (Swales et al. 2002; Thrush et al. 2004; Wilmshurst et al. 2008). Studies originating from Te Ika-a-Māui/North Island suggest that prior to Polynesian settlement (i.e., before 1300 A.D.) annual-average SAR was 0.1–0.5mm/yr (Swales & Hume 1995; Swales et al. 2012; Hunt 2019a and references therein). However, changes to land-use over the last century have resulted in annual-average SAR of 2–5mm/yr and up to 10 – 30mm/yr in some tidal creeks, mangrove forests, and estuaries at the base of large catchments (Hume & McGlone 1986; Sheffield et al. 1995; Swales et al. 2002; Hunt 2019a and references therein). While annual-average SAR can be a useful indicator of sediment stress, it should be noted that not all sediment stress is represented by sedimentation and thus will not be detectable from this measure alone.

PROPOSED METRICS

The following indicator metric is proposed for sedimentation.

1. Sediment accretion rate

7.1 SEDIMENT ACCRETION RATE (SAR)

Indicator type: Supporting.

Metric: Change in average annual sediment level at site-specific estuary location.

Unit of measurement: millimetres per year (mm/yr).

Spatial scale: Site-specific. It is not recommended to average SAR across a number of sites for whole estuary statistics.

Applicability: Nationally.

Rationale: SAR describes the vertical change to a substrate surface over time, giving a quantitative estimate of sedimentation at the measurement site. Site-specific sampling can be easily carried out with targeted, random, or stratified-random sampling approaches used to select sites.

Method: Historical sedimentation: Coring and dating methods down a vertical sediment profile can be used to generate long-term averages to gain an understanding of historical or natural sedimentation rates. Some examples of dating isotopes include caesium (^{137}Cs), lead 210 (^{210}Pb), radiocarbon (^{14}C) and pollen. Accuracy of measurements can be influenced by rate of bioturbation, deposition, and the degree of sediment compaction. These methods are not suitable for recent deposits because of bioturbation reworking the upper 5-10cm of sediment. Multiple measurement methods have the potential of delivering convergent lines of evidence, which can increase confidence in historic SAR estimates.

Short-term/modern sedimentation: Identifying changes in bed height from a given reference point (e.g., sediment rods, traps, plates) can provide sedimentation estimates on a daily to yearly scale. These methods are generally only be applied to intertidal areas for practical reasons. Methodology includes sediment plates which are widely used by regional authorities across New Zealand, and are large (often 20–30cm^2), flat plates which are buried a known distance (e.g., 20cm) beneath the sediment surface (Hunt 2019b, a). Plastic mesh plates have the advantage of being permeable to water and therefore less likely to influence surficial surface sediment compaction (Swales et al. 2002; Hunt 2019b). However, mesh plates are less appropriate if large bioturbators are present unless buried deeper below the surface sediment and they can be more difficult to accurately measure than solid plates. There are variations in the methods used, but broadly, on installation, the plates are levelled and the initial depth of sediment above the plates is measured. Future measurements are made by inserting a probe into the sediment until the plate is reached to determine accretion or erosion rates. For both sediment plate and rod measurements, the effect of localised surface irregularities above the plate and scouring around the rod need to be accounted for when taking measurements, e.g., by use of a straight edge to average surface irregularities, or by the collection of multiple measurements to obtain a representative site average.

Measurement Considerations: As sediment does not accumulate evenly, locations to be measured need to be carefully assessed. It is also important to understand the longer-term stability of the identified site, as the erosion and deposition of sediment has high temporal and spatial variations. According to Hunt (2023), three main considerations should be included when assessing SAR: (1) Temporal scales - Sedimentation operates over a range of complex temporal scales with short-term changes between each sample larger than the net long-term average rate. (2) Spatial distribution - The variability in sedimentation between the plates suggests that single plate measurements are unlikely to provide representative measures of estuary wide SAR. (3) Operational issues - In areas of continued erosion the plates can become uncovered, and scour can cause those plates to tilt. Poor choice of plate locations has led to infrequent sampling due to restrictions around access, weather and tides.

Calculation of statistic: To calculate SAR at each site over the entire monitoring period, replicate samples are averaged for each plate and for each sampling occasion, and then a linear trend fitted to the averaged sediment-level data. The average rate of sediment accumulation is calculated in mm/day from the slope of the linear trend line and then converted to mm/yr. The 95% confidence interval for the SAR is also calculated for the linear trend at each plate.

Potential bands and/or thresholds and rationale (including caveats): There are a few SAR guidelines that have been proposed for use nationally across New Zealand. For example, the Australian and New Zealand Environment and Conservation Council (ANZECC) & Agriculture and Resource Management Council of Australia and New Zealand (ARMCANZ) recommend guideline values should be set at a point below which the risk to the environment is likely to be low. An exceedance of the guideline value should trigger a management response (i.e., the initiation of further investigations) because of an increased likelihood of significant environmental damage (ANZECC & ARMCANZ 2000). Townsend and Lohrer (2015) proposed an ANZECC and ARMCANZ "Default Guideline Value of 2mm of sediment accumulation per year above the natural annual sedimentation rate for the estuary, or part of estuary, at hand". If the natural sedimentation rate (i.e., that under a native forested catchment) is not known, then the value of natural sedimentation is assumed to be 0mm/yr resulting in the default guideline value being used as the threshold.

Waikato Regional Council uses a Guideline Value of 2.2mm/yr of sediment accumulation above the natural annual sedimentation rate for the estuary (Hunt 2019b, 2023). Salt Ecology has used the above default guideline value to propose preliminary SAR thresholds (e.g., Stevens 2018). Here they use a value ≥2mm/yr above natural SAR to indicate an increased likelihood of significant environmental damage and is therefore rated as being reflective of 'Poor' conditions. Guidance on bands of 'Very good' (<0.5mm/yr) to 'Fair' (<1mm/yr) have been derived from international literature on the short-term impacts of sediment deposition (Lohrer et al. 2004) and the pre-Polynesian sedimentation rates stated within Townsend and Lohrer (2015).

The Estuay Trophic Index (ETI) assesses SAR based on the ratio between current sediment accumulation rate and the natural sediment accumulation rate (NSR), rather than an absolute rate, and uses modelled sediment deposition rates. The ETI suggested the use of four bands ranging from no stress (SAR = 1 to 1.1 x NSR) and increasing to highly stressed (1.1 to 2 x NSR, 2 to 5 x NSR and >5x NSR) (Robertson et al. 2016).

Although there is general agreement that average annual SAR above ~2mm/yr is likely to have adverse effects on estuarine benthic organisms (Townsend and Lohrer 2015; Stevens 2018; Hunt 2019a, 2023), other thresholds that have been proposed to divide SAR into categories of 'Good', 'Fair', and 'Poor' are not well underpinnned by ecological data collected in New Zealand or overseas. Most notably, it is difficult to derive clear relationships between decade-scale average annual SAR and ecological condition due to confounding effects of acute sediment deposition events (e.g., following major storms) and the fact that SAR is only one part of the stressor profile in estuaries.

Despite the described limitations, and noting that short-term trends of sedimentation are not directly comparable to long-term historical trends from cores, there is a high likelihood that ecological health will be (possibly linearly) degraded as levels of average annual SAR increase. Although speculative, a threshold/banding approach may be useful for environmental managers seeking guidance on SAR as a means of understanding and improving estuarine ecological health.

The thresholds proposed here are based on those suggested by Salt Ecology and considering the studies of Townsend and Lohrer (2015) and Hunt (2023).

Summary of proposed thresholds: Proposed SAR thresholds are presented in Table A7-1. The proposed thresholds are consistent with ANZECC and WRC default guideline value recommendations (Townsend and Lohrer 2015, Hunt 2023). Default guideline values (DGV) are meant to signal an increased likelihood of significant environmental damage, therefore, we used this to define the threshold point between 'Fair' and 'Poor' [DGVs in Townsend and Lohrer (2015) and Hunt (2023) are 2 and 2.2mm/yr above natural SAR, respectively]. A value of >2mm/yr was also used by Stevens 2018 to define 'Poor' conditions.

We do not recommend trying to differentiate SAR bands based on increments of less than 1mm/yr; this is impractical based on the amount of variation usually observed in linear least squares regression fits to SAR time-series data, the width of confidence intervals around the fits, and the influence of high/low values at either end of the time-series being analysed.

Table A7-1: Recommended SAR thresholds for New Zealand estuaries.

Average annual SAR (mm/yr)	Ecological Quality Status			
	Good	Fair	Poor	Very Poor
If assumed natural SAR ≤1mm/yr	0 to 1	≥1 to 3	≥3 to 10	≥10
mm/yr above natural SAR	0	>0 to 2	≥2 to 10	≥10
Narrative	No to minor stress on sensitive organisms.	Moderate stress on some species and a risk of sensitive macroinvertebrate species being lost.	Significant, persistent stress on a wide range of aquatic organisms.	A likelihood of local extinctions of keystone species and loss of ecological integrity.

Note that the threshold value between 'Poor' and 'Very Poor' (10mm/yr) is the most uncertain. It could be anywhere between 5mm/yr or 10mm/yr. Knowing exactly where this threshold lies is somewhat unimportant; we should aspire to keep our estuaries above 'Poor'. Moreover, the value of SAR (as a continuous variable that is inversely related to ecology health) is more important that knowing the health 'band' or category.

Overall confidence in thresholds/ bands: Fair. There are some studies, but they report high variability in SAR results which are event-dependent (i.e., erosion, storm, daily deposition, etc). Furthermore, information on the influence of average SAR on ecosystem health is lacking, with only one previous report on SAR, and the ANZECC calculations were based on short-term terrigenous sediment deposition experiments to infer annual SAR thresholds. However, SAR when used in conjunction with other indicators could potentially be used to distinguish between 'Good/Fair' and 'Poor/Very poor' ecological health.

Recommendation: SAR should be used in conjunction with other sediment indicators, e.g., as recommended in Townsend and Lohrer (2015).

Links to other indicators: Suspended sediment concentration (not included in the current project by MfE), bed sediment particle size distribution (sediment mud content), the areal extent of muddy sediment in an estuary and seagrass health. These factors, in addition to SAR, have been previously recommended to be considered together when understanding sedimentation impacts (Townsend & Lohrer 2015).

Alternative metrics considered: SAR can be used as one aspect of sedimentation impact assessments on estuaries. Other key factors to consider are suspended sediment concentration, change of within-site muddiness, areal extent of muddy substrate and the impacts of storm events (i.e., large sedimentation events), water turbidity, and seagrass extent (Hale et al. 2024).

Additional work recommended: Greater understanding of the link between SAR and ecological health.

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APPENDIX 8. MACROFAUNA

Authors: Orlando Lam-Gordillo (NIWA, Hamilton) and Barrie Forrest (Salt Ecology)

Macroinvertebrates are organisms >0.5mm body size, globally distributed, and often their living habit is closely associated with the seafloor, for example polychaetes, crustaceans, and molluscs (Snelgrove et al. 2014; Thrush et al. 2021). Macroinvertebrate communities are widely used as bioindicators to assist assessments of ecosystem health due to their sensitivity to natural and anthropogenic disturbances (Borja et al. 2000a; Thrush et al. 2008; Clark et al. 2020; Drylie 2021; Lam-Gordillo et al. 2022a Lam-Gordillo et al. 2024).

BACKGROUND

Macroinvertebrate communities are major providers of ecosystem functions and services in marine habitats. For example, they actively disperse and modify soft sediment habitats by bioturbating the sediment, which promotes sediment oxygenation and enhances sediment mineralisation and nutrient cycling , they transfer energy and matter from lower to higher trophic levels as food sources for fish and birds, and modify soft-sediment habitats through biological processes such as ingestion, digestion, excretion, and bioturbation, which facilitates microbial recycling of nutrients, detoxification of pollutants, and organic matter remineralization (Lohrer et al. 2004a; Kristensen et al. 2012; Thrush et al. 2017; Douglas et al. 2019). Macroinvertebrates are also important secondary and tertiary producers, constituting the link between benthic and pelagic ecosystems (Pearson & Rosenberg 1978; Villnäs & Norkko 2011; Kristensen et al. 2014; Thrush et al. 2021),

Past and ongoing anthropogenic pressures such as coastal development, conversion of natural habitats to land for agriculture and forestry, fishing and resource extraction, industrialisation, and increasing nutrient and sediment inputs, in combination with overarching climate change, are degrading the health of macroinvertebrate communities (Lohrer et al. 2010; de Juan et al. 2014; Hewitt et al. 2016; Ellis et al. 2017; Douglas et al. 2019; Hicks et al. 2019; Jones et al. 2022; Douglas et al. 2023; Lam-Gordillo et al. 2024). These anthropogenic pressures can alter the composition and structure of macroinvertebrate communities, potentially limiting the provision of key ecosystem services. For example, rapid and spatially widespread shifts were observed in macroinvertebrate community composition in New River and Jacob's River estuaries (Southland) following dairy intensification (Robertson et al. 2017; Lohrer et al. 2020; Forrest et al. 2022d; Roberts et al. 2022).

There are many New Zealand studies describing relationships between macroinvertebrate community composition and stressors such as nutrients and mud content in sediments (e.g., Lohrer et al. 2004b; Thrush et al. 2013; Douglas et al. 2017; Thrush et al. 2017; Forrest et al. 2021, 2022a; Forrest et al. 2023a; Forrest et al. 2023b; O'Connell-Milne & Forrest 2023). However, assessments identifying multiple stressors affecting these communities are limited, information on tipping points is scarce, and the further consequences to ecosystem functioning and provision of ecological services is lacking. With the existence of standardised national protocols for monitoring, collection, identification of macroinvertebrate communities, and macroinvertebrate-based metrics (Robertson et al. 2002; Hewitt et al. 2014; Clark et al. 2020; Greenfield et al. 2023), there is baseline information on macroinvertebrate communities for many of the main intertidally-dominated estuaries in New Zealand, as well as for some of the tidal river systems. This baseline will facilitate New Zealand-wide comparisons and the implementation of national guidelines, although more research on how to generalise and expand some of the macroinvertebrate-based metrics to the national level will be required.

PROPOSED METRICS

There are already several metrics and numeric bands (i.e., indicator thresholds) being used in New Zealand to describe the status of macroinvertebrate communities. We describe three metrics previously proposed for use in a New Zealand Estuary Trophic Index (ETI) (Robertson et al. 2016a) that aim to distil multivariate benthic community data into a single number, which are as follows:

1. AZTI's Marine Biotic Index (AMBI) – section author Barrie Forrest
2. National Benthic Health Models (BHM)– section authors Orlando Lam-Gordillo and Barrie Forrest
3. Traits Based Index (TBI) – section author Orlando Lam-Gordillo

8.1 AZTI'S MARINE BIOTIC INDEX

Indicator type: Supporting.

Metric: Numeric score on continuous 0-7 scale, with low scores indicating a low impact on ecological (macroinvertebrate) health and high scores a high impact.

Unit of measurement: AMBI score.

Spatial scale: Site, Estuary.

Applicability: National.

Rationale: Estuarine ecosystems are currently threatened by several anthropogenic pressures, affecting their ecological integrity, including changes in the composition and resilience of benthic macroinvertebrate communities. To understand the responses of ecological communities to anthropogenic disturbance and to manage and mitigate effects, indices for assessing the ecological integrity of estuarine and coastal waters have been created worldwide.

The AZTI Marine Biotic Index (AMBI) was initially developed in Europe, with a focus on the effects of organic enrichment on marine benthos (Borja et al. 2000b). Subsequent studies have demonstrated that AMBI can successfully be applied to evaluate broad sources of anthropogenic and natural disturbance in estuarine and coastal environments (Muxika et al. 2005; Borja et al. 2019). AMBI scores reflect the proportion of species abundances in each of five eco-groups (EG I to V; Roman numerals are used to designate eco-group number) that reflect sensitivity to disturbance, ranging from relatively sensitive (EG-I) to relatively resilient (EG-V). With the wide adoption of this index internationally, the AMBI database (last updated in June 2022) specifies EG's for more than 11,000 species or higher taxa.

In New Zealand, the AMBI was recommended as the primary macrofaunal index for assessing estuary health as part of the Estuary Trophic Index (ETI) Toolbox project that was completed in 2016 (Robertson et al. 2016a). The AMBI was not modified but was referred to in the ETI as the "NZ hybrid AMBI" on the premise that it utilises NZ-specific EG's (developed for sediment mud sensitivity) supported by international EGs as necessary (Robertson et al. 2016a). The AMBI has been evaluated in various estuary-specific and national studies in New Zealand (e.g., Rodil et al. 2013; Robertson et al. 2016b; Berthelsen et al. 2018; Forrest et al. 2022a; Forrest et al. 2022b; Forrest et al. 2023a), along with extensions to the original AMBI including: (i) multivariate AMBI (M-AMBI), which incorporates species richness and Shannon diversity (Muxika et al. 2007; Borja et al. 2012); and (ii) richness integrated AMBI (RI-AMBI), which accounts for proportional representation of EG's by richness as well as abundance (Robertson et al. 2016b; Berthelsen et al. 2019). Here we focus on the AMBI.

Method:

AMBI calculation involves the following steps:

1. Remove juveniles from the data when the particular species are not identified, and remove non-soft sediment taxa and epifauna (Borja & Muxika 2005).
2. Assign EGs to each taxon present, using NZ-specific EGs where available. NZ-specific EGs have been used in at least two national studies (Robertson et al. 2015; Berthelsen et al. 2018), however: (i) there is no 'agreed' comprehensive EG list, and (ii) there is uncertainty regarding the consistency of taxonomic resolution and aggregation in some of the taxa for which EGs have been developed (see next section). As such, we recommend that:
    a) AMBI scores are calculated using NZ-specific EGs for named species or higher taxa, but not placeholder names (e.g., Amphipoda sp. 1) for which taxonomic consistency is uncertain.
    b) NZ-specific EGs should be supplemented as necessary with international EG classifications (http://ambi.azti.es), until definitive EG classifications for New Zealand are developed.
    c) For New Zealand species without EGs, we recommend using EGs for similar taxa (e.g., other species within the same genus) following methods such as described by (Forde et al. 2013).

3. Based on criteria provided by Borja and Muxika (2005), we recommend that AMBI is calculated using data for individual replicates, and used with caution if samples have a very low number of taxa (≤3) and/or individuals (<3 per replicate), or the percentage of taxa without EGs is >20% (when this percentage is >50%, AMBI should not be used).
4. Notwithstanding the previous point, we recommend that, where replicates within a site do not meet AMBI criteria, replicate data are pooled and site-AMBI is calculated (provided the criteria are met after pooling).

Measurement considerations: The simplicity of the AMBI method and calculation (see next section) is intuitively appealing. However, whether AMBI scores reliably reflect the ecological quality status of a location depends on whether assigned eco-groups are a true reflection of each species' sensitivity. Initial work to better define AMBI eco-groups for New Zealand (Robertson et al. 2015) included taxa that were given generic placeholder names (e.g., Amphipod sp. 1, sp. 2). Subsequent taxonomic QA/QC work by NIWA and Coastal Marine Ecology Consultants (Mills et al. 2021) revealed that these placeholder names were not always used consistently among the estuaries included in New Zealand EG development (Robertson et al. 2015), with some of the placeholder names containing different species with potentially different sensitivities.

Calculation of statistic: AMBI scores are calculated based on the % contribution to abundance of EG-I to EG-V (Roman numerals are used to refer to eco-group), as follows:

AMBI = [(0 * %EG-I) + (1.5 * %EG-II) + (3 * %EG-III) + (4.5 * %EG-IV) + (6 * %EG-V)]) / 100

Potential bands and/or thresholds and rationale (including caveats): The parent index (Borja et al. 2000b) was based on a scale of 0-7, whereby 0 would represent a community solely comprising EG-I (theoretical but implausible) and 7 would represent an azoic state devoid of macroinvertebrates. As an output from the ETI, four bands (A-D) were proposed (Robertson et al. 2016a). The relationship between these two sets of thresholds is shown in Table A8-1, with the upper ETI Band D (>4.3-7) based on an analysis of national estuary data by Robertson et al. (2016b).

Table A8-1. AMBI bands and original descriptors (Borja et al. 2000b), and relationship to bands proposed for New Zealand (Robertson et al. 2016a). Descriptors are taken verbatim from source material.

<table>
<tr><th>Index</th><th>AMBI range</th><th>Original AMBI ecological quality descriptor</th><th>ETI band class and ecological quality descriptor</th><th>ETI AMBI range</th></tr>
<tr><td>0</td><td>0 to 0.2</td><td>Unpolluted/ normal</td><td rowspan="2">Band A. None to minor stress on benthic fauna. Community intolerant of organically enriched conditions and elevated muds.</td><td rowspan="2">0 to 1.2</td></tr>
<tr><td>1</td><td>>0.2 to 1.2</td><td>Unpolluted/ impoverished</td></tr>
<tr><td>2</td><td>>1.2 to 3.3</td><td>Slightly polluted/ Unbalanced</td><td>Band B. Minor to moderate stress on benthic fauna. Community tolerant of slight organic enrichment and moderate muds.</td><td>>1.2 to 3.3</td></tr>
<tr><td>3</td><td>>3.3 to 4.3</td><td>Meanly polluted/ transitional to pollution</td><td>Band C. Moderate to high stress on benthic fauna. Community tolerant of moderate organic enrichment and elevated muds.</td><td>>3.3 to 4.3</td></tr>
<tr><td>4</td><td>>4.3 to 5.0</td><td>Meanly polluted/ polluted</td><td rowspan="4">Band D. Persistent, high stress on benthic fauna. Community tolerant of high and very high enrichment and elevated muds or community is devoid of life.</td><td rowspan="4">>4.3 to 7</td></tr>
<tr><td>5</td><td>>5.0 to 5.5</td><td>Heavily polluted/ transitional to heavy pollution</td></tr>
<tr><td>6</td><td>>5.5 to 6.0</td><td>Heavily polluted</td></tr>
<tr><td>7</td><td>>6.0 to 7</td><td>Azoic</td></tr>
</table>

Summary of proposed thresholds: We recommend modifying the ETI four-band scheme in Table A8-1 above, to a five-band scheme, as follows and summarised in Table A8-2:

- Retain the same thresholds as ETI Bands A to C.

- Disaggregate Band D into two separate bands: comprising AMBI >4.3 to 5.0, and AMBI >5.0 to 7. Although New Zealand data indicate most AMBI scores are <5 (Robertson et al. 2016b; Salt Ecology unpubl. data), enabling scores >5 caters for situations more severe degradation (e.g., due to extreme eutrophication) occurs.

Table A8-2. Recommended thresholds for macroinvertebrate AMBI scores.

AMBI score	Ecological Quality Status				
	Very Good	Good	Fair	Poor	Very Poor
	0 to 1.2	>1.2 to 3.3	>3.3 to 4.3	>4.3 to 5.0	>5.0 to 7
Narrative	No to minor stress on benthic fauna. Community intolerant of organically enriched conditions and elevated muds.	Minor to moderate stress on benthic fauna. Community tolerant of slight organic enrichment and moderately elevated muds.	Moderate to high stress on benthic fauna. Community tolerant of moderate organic enrichment and elevated muds.	High stress on benthic fauna. Community tolerant of high enrichment and elevated muds.	Persistent, high stress on benthic fauna. Community tolerant of very high enrichment and elevated muds or community is devoid of life.

Overall confidence in threshold/ bands: High . There is widespread international acceptance and use of the index, and the proposed bands are consistent with the ETI (with the exception that Band D is partitioned into two sub-bands). Key uncertainties or issues to address or at least recognise are:

- The need to confirm or develop reliable and agreed EGs for New Zealand species or higher taxa.
- Recognition that macroinvertebrate communities, hence AMBI scores, can reach levels consistent with the proposed 'Poor' band due to natural stressors.

Recommendation: AZTI's Marine Biotic Index

Adopt Table A8-2 as preliminary numeric thresholds pending data analysis/review.

Links to other indicators: Other indicators linked to AMBI are sediment mud content, nutrients, and enrichment indicators (especially total organic carbon, total sulfur & RPD).

Alternative metrics considered: Alternative metrics considered here (discussed below as stand-alone indicators) are the TBI and National BHM. With respect to AMBI specifically, based on the conclusions of a national study by Berthelsen et al. (2018), we also recommend the RI-AMBI as an index worth further evaluation. The calculation method for the RI-AMBI described by Robertson et al. (2016b) is straightforward, and scores can be compared to the same thresholds described here.

Additional work recommended: Since the initial efforts to develop EGs for NZ-AMBI calculation, there now exist more comprehensive macrofaunal data sets across many New Zealand estuaries, which have been through a QA/QC process, and for which associated data on sediment quality are available (i.e., sediment grain size, trophic state indicators, trace metals). For the full potential of AMBI to be realised, we therefore recommend the following:

i. Develop NZ-specific EGs using QA/QC'ed macrofauna datasets, for which associated sediment quality data are available.

ii. Road test the EGs with a group of experts to achieve consensus, and make the agreed EGs available nationally (e.g., include EGs in the C-SIG Coastal Species Resource Tool; see: https://specieskey.atlasmd.com/).

iii. Further evaluate the limitations of AMBI in different estuary types and consider its utility as a tool for assessment of temporal change in stressor affects at discrete sites. Simultaneously, evaluate the RI-AMBI.

iv. Develop a guidance and tools (e.g., R code, desktop app) specific to the NZ-AMBI to enable easy calculation of AMBI and RI-AMBI by councils and science providers.

8.2 NATIONAL BENTHIC HEALTH MODELS (BHM)

Indicator type: Supporting

Metric: Numeric score on continuous 1-5 scale, with low scores indicating low impact and high scores indicating high impact, relative to other New Zealand estuaries used to develop the National BHM.

Unit of measurement: BHM score.

Spatial scale: Site, Estuary.

Applicability: National.

Rationale: Estuarine ecosystems are currently threatened by several anthropogenic pressures, affecting their ecological integrity, including changes in the composition and resilience of benthic macroinvertebrate communities. To understand the responses of ecological communities to anthropogenic disturbance and to manage and mitigate effects, indices for assessing the ecological integrity of estuarine and coastal waters have been created worldwide. The New Zealand National Benthic Health Models (BHMs) use information about organisms living in soft sediments to assign a score that indicates the relative health of an estuarine site in response to sediment mud content (Mud BHM) and heavy metal contamination (Metals BHM) (Clark et al. 2020; Clark 2022a). The BHM index helps with prioritisation of mitigation measures because each of the models are linked to a specific stressor (sedimentation, or heavy metal contamination based on Cu, Pb and Zn). The BHMs use macroinvertebrate data commonly collected for estuary fine-scale monitoring by councils, which avoids the need for additional sampling effort (Clark et al. 2020; Clark 2022a). Since original publication in 2020, the BHMs have been applied to most New Zealand estuaries for which councils have routine SOE monitoring data.

Method: Methods for the calculation and use of BHMs are fully described by (Clark et al. 2020) and (Clark 2022a). Briefly, the approach is as follows:

(1) Collect macroinvertebrates using standard protocols (e.g., Robertson et al. 2002),

(2) Identify macroinvertebrates to the lowest practical taxonomic level (aiming to match the taxa lists presented in Clark et al. (2020) and Clark (2022a)),

(3) Compile and use the model data for mud and metals,

(4) Fill the Excel file template, and

(5) Run the multivariate canonical analysis of principal coordinates in PRIMER software.

Measurement considerations: The National BHMs have been shown to perform well (R^2 values between CAP scores and stressor levels, see Clark et al. 2020 for details) in two main estuary types (i.e., tidal lagoons and shallow river valleys) across five to six aggregated regions of New Zealand (Mud BHM: Abel, Banks, Chalmers, Portland, Raglan and Northeastern; Metals BHM: Abel, Southeastern, Portland, Raglan and Northeastern) (Clark et al. 2020). In some instances where the BHM has been applied to assess temporal changes at specific estuary sites, the Mud BHM has been relatively unresponsive to large changes in sediment mud content (Forrest et al. 2022c), suggesting that more work needs to be done to understand the BHM an indicator for monitoring and management purposes. Councils appear supportive (e.g., Auckland and Waikato Regional Council) of the use of the National BHM models, with further testing and refinement urged as more data become available.

Calculation of statistic: BHMs are calculated based on multivariate canonical analysis of principal coordinates (see Clark et al. (2020) for details). To date, BHM scores for most councils have been calculated by Cawthron Institute or NIWA. Although familiarity with calculation steps and access to PRIMER software is required, the methodologies are openly available to council staff who wish to calculate scores on their own. In some cases, Cawthron or NIWA could provide a guiding or verification role for councils who are calculating scores on their own.

Potential bands and/or thresholds and rationale (including caveats): The thresholds/bands used for the BHMs are scores from 1-5. These are a measure of impact relative to other New Zealand estuaries on which the BHM approach was developed, rather than an absolute measure. The scores are shown in Table A8-3 and range from 1 - <2 representing a 'very low' relative impact, and ≥5 representing a 'very high' relative impact. The thresholds are based on even increments in BHM scores and have not been tailored to consider where the strongest shifts in macrofauna occur. Note that subsequent to the publication of the BHM, 'absolute' thresholds based on three categories ('good', fair', 'poor') have been proposed for Metals BHM scores (Clark 2022b), which are benchmarked against highly conservative sediment quality guidelines derived primarily from field-based species sensitivity distributions (e.g., Hewitt et al. 2009). For present purposes, we focus on the relative impact thresholds.

Summary of proposed thresholds:

Table A8-3. Summary of the threshold/band values proposed by the BHMs as a relative measure of impact rather than an absolute measure of health.

BHM Group	Level of impact relative to other estuarine sites in New Zealand	BHM score
1	Very low	1 to <2
2	Low	2 to <3
3	Moderate	3 to <4
4	High	4 to <5
5	Very High	≥5

Overall confidence in thresholds/ bands: High in terms of rating estuaries against each other in a relative sense; but Fair as a monitoring tool, reflecting that some studies have not found a compelling relationship between mud content and Mud BHM when assessing temporal change at discrete sites. That said, temporal trends in BHM scores can nonetheless be evaluated to determine whether there is a directional change (e.g., degradation) in estuary state.

Recommendation: National Benthic Health Models (BHM)

Adopt as preliminary numeric thresholds/bands but further testing and refinement urged as more data become available for wider estuaries in New Zealand. In particular, the BHM needs to be tested using time series data from discrete sites within estuaries where marked changes in mud or metals levels have occurred, to evaluate its efficacy for council SOE monitoring.

Links to other indicators: Other main indicators linked to the BHMs are mud content and metal concentrations (Cu, Pb, Zn) in sediment. Also, catchment drivers such as sediment and contaminant mass loads are important related considerations.

Alternative metrics considered: Alternative metrics considered here (discussed as stand-alone indicators) are the TBI and AMBI.

Additional work recommended:

i. Continue to trial the BHMs in a wider range of other estuary types across New Zealand, and evaluate the efficacy of the method for tracking temporal change in the effects of sediment mud and metals.
ii. Evaluate the scope to refine the relative ranking thresholds based on the BHM scores where the strongest shifts in macrofauna occur.
iii. Seek to develop 'absolute' thresholds that relate BHM scores to ecological condition, rather than scores relative to other estuaries.
iv. Support proposed work to develop a software package (likely within the software Primer), to enable easy and reliable BHM score calculation by councils and science providers. Simultaneously, it is recommended that training to use any such software is provided, to help ensure consistent application and interpretation.

8.3 TRAITS BASED INDEX (TBI)

Indicator type: Supporting

Metric: Numeric score on continuous 0-1 scale (0 = completely defaunated, 1 = a non-polluted reference value).

Unit of measurement: TBI score.

Spatial scale: Site, Estuary.

Applicability: Limited to intertidal areas of Auckland and Waikato estuaries.

Rationale: Estuarine ecosystems are currently threatened by several anthropogenic pressures, affecting their ecological integrity, including changes in the composition and resilience of benthic macroinvertebrate communities. To understand the responses of ecological communities to anthropogenic disturbance and to manage and mitigate effects, indices for assessing the ecological integrity of estuarine and coastal waters have been created worldwide. The TBI is constructed from the richness of macrofaunal taxa in seven functional groups (e.g., suspension feeders, organisms that live in the surface 0-2cm, etc.), which are well known to respond to changes in sediment mud percentage and heavy metal contaminant concentration gradients below international guidelines. In one study the TBI performed marginally better than indices developed overseas, including the Benthic Index of Biotic Integrity developed in the USA, and the AMBI developed in Europe (Rodil et al. 2013). The TBI successfully tracked the stressors that were the most relevant locally and indicated the relative levels of within-group taxonomic richness at various sites. As within-group richness is a component of functional redundancy and ecological resilience, the TBI offers a trifecta of simplicity, robustness and meaningfulness that will facilitate management (Rodil et al. 2013).

Method: Methods describing the calculation and use of TBI are fully described by (Hewitt et al. 2012; Rodil et al. 2013). Briefly, TBI calculations are done at the site level (not at replicate level), and are based on the list of species present at the site. The calculation of TBI follows 6 steps:

(1) Match species/taxa with species/taxa scores,

(2) Generate presence/absence data for each site,

(3) Multiply each species' traits score by its presence/absence value across the entire data matrix,

(4) Sum the scores in a given site/date column,

(5) Select the relevant SUMmax score (based on the number of replicates used to calculate the SUMactual), and

(6) Divide SUMactual for a given site or time by the appropriate SUMmax value (Hewitt et al. 2012; Rodil et al. 2013).

Measurement considerations: The authors of the TBI strongly advise against its use outside of Auckland and Waikato at this time, and in the comparison of TBI scores from intertidal and subtidal habitats (Rodil et al. 2013; Lohrer et al. 2023).

Calculation of statistic: TBI values are calculated using the following formulas/statistics:

(1) The taxonomic richness in each of the 7 trait groups per site are summed (i.e., NtaxaTop + NtaxaErect + NtaxaSS + NtaxaSedentary + NtaxaSus + NtaxaMedium+ NtaxaWorm) to produce a quantity called SUMactual.

(2) A maximum expected value called SUMmax (i.e., a non-polluted reference value) is determined (expert determined). This quantity varies depending on the number of replicate samples used to calculate SUMactual.

(3) A minimum possible value (i.e., a completely defaunated site) is set at 0.

(4) The TBI formula is 1 – (SUMmax – SUMactual)/ SUMmax, which essentially standardises the index values to fall between 0 and 1. Values near 0 indicate highly degraded sites, and values near 1 indicate the opposite (Rodil et al. 2013; Hewitt et al. 2014).

Potential bands and/or thresholds and rationale (including caveats): The thresholds/bands used for the TBI are scores from 0-1. Scores >0.4 are considered 'good', 0.3 – 0.4 are considered 'moderate', and <0.3 are considered to represent 'poor' health and low functional redundancy.

Summary of proposed thresholds:

Table A1: Summary of the threshold/band values proposed by the TBI in relation to estuarine ecological health.

TBI Score	Estuarine Health	Narrative
>0.4 to 1	Good	High richness in macroinvertebrate functional groups that are sensitive to mud and metals, suggesting a healthy and resilient macroinvertebrate community (high functional redundancy).
≥0.3 to 0.4	Moderate	Moderate level of richness in macroinvertebrate functional groups that are sensitive to mud and metals, suggesting a macroinvertebrate community that may be slightly impact and less tolerant/resilient to additional stressors.
0 to 0.2	Poor	Low richness in macroinvertebrate functional groups that are sensitive to mud and metals, suggesting low resilience and a community affected by persistent or high stress.

Overall confidence in thresholds/ bands: High for Auckland and Waikato regions, but Fair at national scale. Further work and validation are needed for wider New Zealand.

Recommendation: Traits Based Index (TBI)

Adopt as preliminary numeric thresholds/band, but only for Auckland and Waikato estuaries until there is better extrapolation and validation of the index to wider New Zealand.

Links to other indicators: Other indicators linked to the TBI are mud content in sediment, nutrient loads, and metal concentrations in sediment.

Alternative metrics considered: Alternative metrics considered here (discussed as stand-alone indicators) are the National BHM and AMBI.

Additional work recommended:

i. Develop guidance (e.g., methods manual, open-source R code, desktop app.) to enable easy and reliable TBI score calculation by councils and science providers.
ii. Calculate the TBI in other estuaries across New Zealand and compare results with those for Auckland and Waikato estuaries to evaluate national applicability.
iii. Determine the sensitivity of the TBI to changes in key environmental drivers; e.g., sediment mud content, nutrient load, and which are likely to be targeted for management.
iv. Evaluate whether proposed TBI thresholds can be further refined to provide greater discrimination of estuarine health.

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APPENDIX 9. MICROALGAE

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Sediment microalgae describe diverse communities of diatoms, cyanobacteria and unicellular eukaryotic algae among others. Sediment microalgae are important components of coastal ecosystems owing to their high productivity and role in altering sediment stability, biogeochemical gradients and supporting secondary production. Microalgae are known to respond quickly to changing environmental conditions, with numerous factors regulating microalgal biomass. Ultimately this results in biomass being inherently highly variable.

BACKGROUND

Benthic microalgal communities are a combination of diatoms, cyanobacteria, unicellular eukaryotic algae and euglenoids among others which inhabit the sediment surface (MacIntyre et al. 1996). Microalgae are highly productive and are often the dominant primary producer in shallow, unvegetated ecosystems. For example, estimates suggest their productivity can account for up to 50% of total estuarine autochthonous primary production and up to 80% of total benthic carbon fixation (Underwood & Kromkamp 1999). Microalgae additionally contribute a significant food source fuelling secondary production, alter biogeochemical gradients within the sediment through oxygenation and nutrient uptake, and enhance sediment stability through the secretion of extracellular polymeric substances (Miller et al. 1996; Tolhurst et al. 2008; Hope et al. 2020).

Microalgae are known to respond quickly to changing environmental conditions. For example, microalgae undergo diurnal vertical migrations within the sediment to reduce photoinhibition and therefore have natural daily variations in their distribution. The factors regulating microalgal biomass over the longer term are varied, and include light, salinity, pelagic and porewater nutrients, hydrodynamics, sediment type and grazing pressure (Jesus et al. 2009; Aktan et al. 2014). Less is known about the response of microalgae to varying stressors. Some evidence suggests microalgae are responsive to nutrient enrichment (Hope et al. 2020; Zeldis et al. 2020), however, results are highly variable and are often dependent on a number of other environmental factors (Adrienne et al. 2006; Cebrian et al. 2012; Mangan et al. 2022).

Microalgal biomass is measured though a proxy: chlorophyll-*a*. Chlorophyll-*a* is the most common of the six photosynthetic pigments all plants (including microphytes and phytoplankton) use for photosynthesis. A degradation product of algal chlorophyll pigments are non-photosynthetic phaeopigments. Phaeopigment has been used as a proxy for grazing pressure.

PROPOSED METRICS

The proposed metric for monitoring sediment microalgae is:

Chlorophyll-*a* and phaeopigments.

9.1 SEDIMENT MICROALGAE (CHLOROPHYLL-*A* AND PHAEOPIGMENTS)

Metric: Amount of pigment (either chlorophyll-a or phaeopigments) per dry weight of sample.

Unit of measurement: µg/g (dry weight).

Spatial scale: Site-specific.

Applicability: National across all estuary types.

Rationale: Site-specific sampling of surficial sediment can be easily undertaken and carried out with targeted, random or stratified-random sampling across different scales of data resolution. Chlorophyll-*a* is a well-known proxy for microalgal biomass and is used globally.

Method: Chlorophyll-*a* and phaeophytin are measured with a standard methodology. Several sub-samples (~10cm^3) within each sample area are pooled, kept in the dark and frozen until analysis. Once thawed and homogenised, a sub-sample is freeze-dried, extracted with 90% buffered acetone and measured using a fluorometer before and after the addition of hydrochloric acid, which removes phaeophytin (Sartory & Grobbelaar 1984).

Measurement considerations: Samples are typically taken from the upper 1-2cm of surficial sediments using small cores. Samples should be kept in the dark and ideally analysed within one month. Due to high natural spatial and temporal variability of chlorophyll-*a* and phaeophytin both within and between sites within an estuary, site selection can have a major influence on results.

Calculation of statistic: Chlorophyll-a and phaeopigments are calculated using sensitivity calibration factors calculated using pure chlorophyll-a of known concentration, fluorescence before and after acidification, sample weight and volume extracted.

Potential bands and/or thresholds and rationale (including caveats): While increases in chlorophyll-*a* have been reported with increased nutrient availability, values are highly site- and season-specific. Such inherent spatial and temporal variability makes it difficult to quantify thresholds indicative of ecological shifts. Consequently, neither sediment chlorophyll-*a* nor phaeophytin has been previously banded into regional or national thresholds.

Summary of proposed thresholds: No thresholds are being proposed.

Overall confidence in thresholds/ bands: Low. There are some studies/data, but large spatial and temporal variability make banding into thresholds inaccurate.

Recommendation: Sediment microalgae (Chlorophyll-*a* and phaeopigments)

Further work would be required to discern natural variability from anthropogenic changes in order to facilitate the banding of sediment chlorophyll-*a* and phaeophytin.

Links to other indicators: Could increase utility if used in conjunction with other sediment information such as sediment nutrient concentrations and mud content (sediment chlorophyll-*a* often has a positive relationship with mud content) and ecological information such as macrofauna community.

Alternative metrics considered: Chlorophyl-*a* is an appropriate metric for sediment microalgae.

Additional work recommended:

i. Collection and analysis of existing national data (e.g., from regional authorities) to understand variability in chlorophyl-a and phaeophytin seasonally and nationally and across impacted to non-impacted estuaries.

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APPENDIX 10. MUD CONTENT

Author: Barrie Forrest (Salt Ecology)

The combination of accelerated sediment accretion rates and increased sediment mud content have long been recognised as major stressors on estuaries and other coastal ecosystems, and which can significantly alter the hydrodynamic, geomorphology, and ecological characteristics of the receiving system impacting its ecological health. Of particular concern are the accumulation of silt and clay particle size fractions <63µm which 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), and which can result in widespread mud deposition zones developing in upper estuary tidal flats (Robertson et al. 2016b). The following background text is drawn from Zaiko et al (2018) and MfE (2022).

BACKGROUND

The gradual infilling of estuaries with sediment eroded from land is a natural process, but sediment accumulation rates have increased by orders of magnitude since European settlement in many places (e.g., Handley et al. 2017, Hunt 2019a, Ministry for the Environment and Stats NZ 2019, Parliamentary Commissioner for the Environment 2020, Ministry for the Environment 2022). This was initially caused by widespread catchment deforestation, much of this during the mid-1800s to early-1900s, but some current human activities and land use practices can also increase rates of soil erosion and resulting sediment loads delivered to waterways and estuaries. Some activities within estuaries, such as aquaculture, channel dredging, and structures such as causeways can also affect sediment mud content (an important measure of sediment quality) and sediment accretion rates. Further, legacy effects of previous land use decisions mean sediment impacts may be borne for decades or centuries after management changes are made, with some effects nearly impossible to reverse (Townsend and Lohrer, 2015).

Excessive fine (silt and clay) sediment inputs can affect biodiversity by altering habitats, smothering benthic species, and suppressing important biological and biogeochemical processes such as feeding, respiration, photosynthesis, reproduction, recruitment, and denitrification (e.g., Kennish 2002, Thrush et al. 2004, Lohrer et al. 2003, O'Meara et al 2017, Thrush et al. 2021). Impacts also include the loss or degradation of shellfish beds that are harvested recreationally and commercially (e.g., Thrush et al 2013). It can also alter microbial activities (which are critical for organic matter degradation and nutrient regeneration), diminish benthic primary productivity, and reduce the oxygenation of surficial sediments (by capping the seabed, clogging sediment pore spaces, and depriving micro- and macrophytes of light) (Berkenbusch et al. (2002). Townsend and Lohrer (2015) report consequences from short-term 'event' deposition (primarily burial) that include lethal effects on benthic biota, changes in benthic species composition, loss of sensitive species, decline in diversity, and modification of animal behaviours (Hewitt et al. 2003, Thrush et al. 2004, Lohrer et al. 2004, Norkko et al. 2002a).

NEMP monitoring data from New Zealand show that many estuaries now contain extensive areas dominated by sediments with high sediment mud content and have elevated (relative to pre-catchment deforestation baseline) sedimentation rates. Salt Ecology national data from 766 stations across 36 estuaries with paired macroinvertebrate and sediment mud data indicate the most diverse and abundant macrobenthic communities occur in sediments with mud concentrations of less than ~20-25%. Therefore, sediments with >25% mud have been defined as 'mud-elevated' and indicative of likely ecological degradation. For these reasons, mud-elevated (>25% mud) sediment is considered a key attribute for management and a useful supporting indicator for the assessment of estuary trophic status (mud-elevated sediments increase susceptibility to eutrophication).

PROPOSED METRICS

The following indicator metric is proposed for monitoring sediment mud content.

Percent mud content in sediment.

It should be used as part of a multi-faceted approach that includes 'mud-elevated' (>25% mud) sediment extent, mud sedimentation rate, and predicted sediment loads to the estuary (see Appendices 4 and 10).

10.1 SEDIMENT MUD CONTENT

Indicator type: Primary.

Metric: Percent of sediment fraction <63µm (rounded from 625µm) (Wentworth 1922; Folk 1954).

Unit of measurement: Percent (g/100g dry weight).

Spatial scale: Site-specific.

Applicability: National across all estuary types.

Rationale: Sediment mud content comprises the silt and clay particle size fraction <63µm. Sediment mud content is included by most councils in their SOE monitoring programmes, due mainly to concerns over impacts on estuary biota from catchment-derived muddy sediment. Sediment sampling in depositional areas provides an integrated measure of episodic inputs (e.g., during storms), which could be missed when undertaking spot water-quality sampling of related parameters such as turbidity and total suspended solids. As well as mud itself being a stressor, the contaminant-holding capacity of sediments tends to increase with decreasing particle grain size. Hence the concentration of contaminants, as well as nutrients and organic matter, is typically greater in mud than in coarser sediment fractions. As mud can block interstitial spaces, muddy sediments may also be depleted in oxygen compared with sandier sediments (Robertson et al. 2016). Sediment mud content is a simple and inexpensive indicator that can be reliably measured by well-established laboratory methods. Where mud arises from anthropogenic sources, targeting reductions in muddy sediment inputs from catchment activities provides the main avenue for mitigating adverse ecological effects. However, due to lags between mud delivery from catchments, accumulation in sediments, and resuspension and flushing by waves and currents, the measurable benefits to estuaries of a reduction in muddy sediment inputs may takes years (even decades) to manifest.

Method: The NEMP method is based on analysis of sediment samples collected from the surface 20mm of sediment. The method considered the most appropriate for long-term monitoring of mud content, and preferrable to laser diffraction, is wet sieving (Hunt & Jones 2019). This method involves washing sediment through a series of sieves, with the material retained on each sieve dried, weighed and calculated as a percentage of the total. The Hill Labs method involves is based on ASTM 5th edition. It involves prior removal of large objects (e.g., sticks, stones) that are not considered representative of the sample, and application of a dispersant to facilitate wet sieving. Hill Labs do not pre-treat samples with 10% hydrogen peroxide to remove organic material, as described by Hunt and Jones (2019).

Assessment baseline: The baseline can be considered as the first reliable measurement of surface sediment mud content. Note that historical baselines may be established through the analysis of vertical sediment profiles in conjunction with sediment aging techniques.

Measurement considerations: The measured sediment mud content in any particular sample may vary considerably according to the sample pre-treatment and the specific analytical method used (Hunt & Jones 2019, and references therein). As such, for long-term monitoring it is desirable that the same method is used consistently across time in a particular estuary, and preferably among councils and providers nationally (Zaiko et al. 2018). When comparing national data, or considering temporal change in an estuary, any differences in method should be recorded as part of the metadata.

As Hill Labs is used by most councils, a key point to note is that sediment mud content analysis is conducted on samples 'as received' by the laboratory, after removal of stones, etc, considered not to be representative of the sample. As such, it can be expected that organic matter (e.g., fragments of seagrass, macroalgae), detrital material, and fragments of shell etc. that are not readily visible could contribute to the larger particle size fractions (e.g., be included in the ≥2mm fraction classified as gravel), thereby leading to an underestimate of sediment mud content. As such, care should be taken in the field to minimise the non-sediment material that is inadvertently included in the sample. Additionally, the Hill Labs method uses a dispersant which, among other things, will break down sediment flocs into smaller particles. As such, the analytical results may overestimate what Hunt and Jones (2019) refer to as environmentally available sediment (i.e., the sediment that macroinvertebrates are exposed to *in situ*).

Statistic calculated from analysis of site-specific discrete sediment samples: Based on dried samples, sediment mud content is calculated as the weight loss of the total sample according to the percentage of the sample that passes through a 63µm sieve, (i.e., total sample weight minus the weight of material retained on sieves ≥63µm.

Potential bands and/or thresholds and rationale (including caveats): Sediment mud content is suitable for band development. Mud is considered a major stressor on estuary macrofauna and other biota, and many studies in New Zealand have investigated the ecological responses of macroinvertebrates using estuary-specific data (e.g., Norkko et al. 2002; Ellis et al. 2017; Bulmer et al. 2022), regional estuary data (e.g., Hewitt et al. 2012; Rodil et al. 2013; Douglas et al. 2019), and inter-regional or national datasets (Thrush et al. 2003; Robertson et al. 2015; Berthelsen et al. 2019; Clark et al. 2020). Although most of these analyses have not focused on the development of thresholds, they provide insight into levels of sediment mud content that are associated with changes in macroinvertebrate assemblages, due for example to the loss or density reduction in sensitive species and/or the increased prevalence of resilient species. Complementing these studies are Salt Ecology national data from 766 stations across 36 estuaries with paired macroinvertebrate and sediment mud data. From this collective information, the following broad patterns relevant to threshold development can be derived:

- The most sensitive species show a rapid density decline over <1% to ~10% mud content.
- The most diverse and abundant macrobenthic communities occur in sediments with mud concentrations of less than ~20-25%.
- The upper range of the mud-optimum for larger-bodied suspension feeding bivalves (e.g., cockle) is ~50% mud or less (e.g., 40% for pipi, Robertson et al. 2015).
- Above ~40-60% mud, the most sensitive species may be eliminated, and some resilient species can thrive.

Based mainly on Robertson et al. (2015), an output from the New Zealand Estuary Trophic Index Toolbox project was the proposal of four bands for sediment % mud content for discrete sites in shallow intertidally-dominated lagoon type estuaries. These bands had thresholds defined as: <5% mud, 5-15% mud >15-25% mud and >25% mud. In the absence of refined national guidance, these bands have been implemented by Salt Ecology for many NEMP surveys conducted for councils (e.g., Roberts et al. 2021; Forrest et al. 2023; O'Connell-Milne & Forrest 2023). More recently, drawing on available studies and expert opinion, Bulmer and Hewitt (2020) defined five 'state ranges' of sediment mud content as: 'very low' (< 5%), 'low' (5 to 20%), 'moderate' (21 to 50%), 'high (51 to 90%), and 'very high' (>90%).

For the purposes of council guidance, we propose a compromise between the above approaches that incorporates five threshold bands, but:

- Includes a 5 and 10% mud threshold to reflect the range over which dramatic declines in the most sensitive species can occur.
- Recognises 25% as an approximate threshold above which unacceptable declines in species and their abundances may occur.
- Uses 50% as a threshold to represent the mud content above which resilient species can thrive and/or sensitive species may be eliminated.

There is no great value in differentiating thresholds between 50% and 100% mud, as community composition is expected to show little change across this range. In practical terms, very few sites are likely to exceed 90% mud anyway. Also, the experience of walking in an estuary with mud content across this range will be similar; i.e., the sediment will often be very soft to walk on (e.g., people will sink to ankles or knees). Thresholds of <10%, ~10-50% and >50% also have intuitive appeal as they correspond to accepted geological criteria for sand, muddy sand, and sandy mud, respectively (Wentworth 1922; Folk 1954). As part of the revised NEMP, a >25% threshold for sediment mud content is being proposed as a biologically-relevant threshold for mapping the extent of 'mud-elevated' sediment, with >50% mud described as 'mud-dominated'.

The key caveats to recognise in relation to these provisional thresholds (see summary in next section) include the following:

1. The recommendations are drawn from a blend of studies to date whose primary goal has not been the development of mud content thresholds. Care needs to be taken when generalising about mud sensitivities from small datasets, or datasets with unresolved taxonomic classification. For example, even though the national analysis of species-mud relationships described by Robertson et al. (2015) was based on a reasonably comprehensive dataset, some of the generic species names (e.g., Amphipoda sp. 1), have subsequently been revealed to contain more than one species (with potentially different sensitivities). Additionally, the Salt Ecology data show that mud tolerance for 84% of species in the Robertson et al. (2015) was greater than indicated in the data set used in the 2015 study. There would clearly be a benefit is combining datasets from different regional and national studies into a single dataset for analysis of species sensitivities, provided species identifications have been subject to QA/QC procedures.
2. Application of site-specific thresholds for sediment mud is clearly context dependent. Care should be taken when making inferences about estuary health from individual sites without reference to how representative the sites are in a wider estuary context. Prescribing absolute thresholds based on site-specific values will provide a misleading picture of estuary health, unless sediment sampling is representative of the dominant soft-sediment habitats in the estuary overall. In practice, the selection of sites will be determined by monitoring purpose; for example, monitoring may target the parts of an estuary subject to the greatest impacts, but may not be representative of the main tidal flats that are removed from the immediate influence of catchment pressures.

Hence, the above thresholds are a site-specific guide only. Important additional considerations are: (i) whether there is a directional trend in mud content at each site, which can be informed by establishing a baseline (see Alternative metrics considered); and (ii) the extent to which site-specific trends are consistent with changes in other indicators such as the spatial extent of elevated (>25%) mud sediments (as determined by estuary wide mapping).

Summary of proposed thresholds: Table A10-1 summarises proposed thresholds, which for now are recommended as preliminary, subject to more extensive analyses of collated national data.

Table A10-1. Recommended thresholds for sediment mud content.

Sediment mud content (%)	Ecological Quality Status				
	Very Good	Good	Moderate	Poor	Very Poor
	<5%	5 to <10%	10 to <25%	25 to <50%	≥50%
Narrative	None to minor stress on benthic fauna. Community intolerant of elevated sediment mud, with sensitive species thriving.	Minor to moderate stress on benthic fauna. Community tolerant of slight elevation of sediment mud.	Moderate to high stress on benthic fauna. Community tolerant of moderate elevation of sediment mud.	High stress on benthic fauna. Community tolerant of elevated sediment mud.	Persistent, high stress on benthic fauna. Community tolerant of very high sediment mud, with resilient species thriving.

Overall confidence in thresholds/ bands: High – general agreement, but limited data/studies. Although comprehensive national analysis/syntheses have been undertaken, the development of thresholds would benefit from a focused analysis of collated national data from multiple sources and providers.

Recommendation: Sediment mud content

Adopt as preliminary numeric thresholds pending data analysis/review.

Links to other indicators: Catchment sediment load, sedimentation rate (e.g., from sediment plates), and spatial (estuary-wide) extent of elevated (>25%) mud sediments. The location and extent of vegetated habitats (seagrass, nuisance macroalgae, salt marsh) can also be important; for instance, seagrass traps fine sediment and can release mud into estuaries during die-back.

Alternative metrics considered: A complementary site-specific metric is the change in sediment mud content relative to a baseline state. The latter could be established in various ways, including:

- By monitoring surface sediment at discrete sites, such that the status quo mud content is used as the baseline for future comparisons.
- Hindcast methods such as deep sediment coring (and dating) to estimate the 'natural' (pre-human) sediment mud content.

Additional work recommended:

i. More extensive analyses of collated existing national data, to specifically focus on threshold development, with consideration of factors that may influence ecological community sensitivity to mud, such as estuary typology.

ii. For future monitoring, seek agreement among councils and providers to ensure consistent and comparable analytical methods for sediment mud content. It is assumed that revisions to the NEMP will provide a means of fostering consistency in methods for sample collection.

iii. Consider the most appropriate way(s) to determine baseline state with respect to sediment mud content and investigate the development of related thresholds. As an interim measure, Zaiko et al. (2018) recommended 'bottom-line' guidance that 'sediment mud content at representative sites should not increase from its current extent'.

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Waikawa Estuary, Southland

APPENDIX 11. NUTRIENTS (SEDIMENT N AND P)

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Nitrogen, and to a lesser extent phosphorus, are important nutrients utilised in plant growth (e.g., macroalgae, seagrass, salt marsh and mangroves). Elevated nutrient concentrations can cause blooms in the growth of phytoplankton and macroalgae in estuaries and therefore play a key role in eutrophication. While direct measures of primary productivity (e.g., macroalgae growth) are likely a more suitable indicator for management purposes, sediment nutrient content can provide useful contextual information on benthic health in estuaries.

BACKGROUND

Sediment nutrients represent sediment bound fractions of nitrogen and phosphorus, and they are typically measured as mg /kg dry sediment weight Total Nitrogen (TN) or Total Phosphorus (TP), representing the combined total concentration of inorganic and organic forms associated with particulate matter (Table A11-1). There is a strong coupling between water column nutrients (see 'Water Column Nutrients' indicator summary) and sediment nutrients, with changes in redox condition potentially altering the equilibrium between the two (e.g., release of iron bound phosphorus under anoxic conditions). Nonetheless, compared to water column nutrients, sediment nutrients are more stable and tend to better integrate nutrient sources over time. Nitrogen is often the limiting nutrient (i.e., nutrient controlling the level of primary productivity) in estuaries, however several studies have highlighted the importance of managing both nitrogen and phosphorus (e.g., Howarth & Roxanne 2006).

Table A11-15. Components of sediment nutrients adapted from Sutula et al. (2011).

Components of Total Nitrogen	Components of Total Phosphorus
Organic nitrogen (detritus left from undecayed or partially decayed organic matter)	Organic phosphorus (detritus left from undecayed or partially decayed organic matter)
Inorganic nitrogen (a minor feature in natural waters and usually not considered)	Inorganic phosphorus (typically associated with minerals)

Sediment nutrients are derived from two main sources: external inputs (e.g., catchment inputs) or internal inputs (e.g., breakdown of plant material). Catchment sediment loads are a significant source of TN and TP in estuaries, with high concentrations generally recorded in estuary depositional areas near freshwater inputs (e.g., Ellis et al. 2015). Sediment grainsize is also an important determinant of sediment nutrient content, with finer (muddier) particles promoting adsorption of nutrients onto mineral surfaces, meaning finer sediments tend to have higher nutrient concentrations (Heap et al. 2001; Sutula 2011). This trend of increasing nutrients with increasing mud content has been observed in most estuaries across New Zealand (e.g., Salt Ecology NZ dataset, unpubl.), and within estuaries over time (e.g., Hale et al. 2024). Because of the influence of grainsize on sediment nutrient concentration, many studies normalise TN and TP content to grainsize (e.g., Sutula 2011).

Sediment Total Organic Carbon (TOC) is often used as an indicator of eutrophication, with quantitative thresholds established to characterise ecological conditions (see 'Organic Matter' indicator summary). There are limited studies on the utility of sediment nutrients as benthic health indicators in estuaries, possibly because they often strongly correlate with sediment TOC (Sutula et al. 2014; Robertson et al. 2016; Salt Ecology NZ dataset, unpubl.). A few studies have explored the use of sediment TN content as a eutrophication indicator in estuaries (e.g., Sutula et al. 2014; Berthelsen et al. 2019; Walker et al. 2022) and, to a lesser extent, sediment TP content (e.g., Ellis et al. 2017).

In California estuaries, sediment TN content was a good descriptor of sediment oxygen depletion (i.e., a common consequence of eutrophication; Sutula et al. 2014). Another study suggested that sediment TN content may be a better indicator of sediment eutrophication because it tends to be highly labile (i.e., more bioavailable) and more indicative of recent biogeochemically active organic material than TOC which can be more refractory (i.e., less bioavailable; Walker et al. 2022). Lability is important because the degree at which the nutrients are bioavailable may be a more important control on eutrophication than the nutrient quantity itself (i.e., a high nutrient content does not necessarily mean it is bioavailable; Sutula 2011). Bioavailability can be dependent on several factors: nutrient form (organic vs inorganic), solubility, redox state, adsorption/ desorption processes, mineralisation/ immobilisation processes and other nutrient cycling pathways. The TOC:TN ratio can also be used as a crude indicator of organic matter source (i.e., terrestrial or marine source) and its subsequent lability. For example, terrestrial plants have a higher TOC:TN and TOC:TP ratio than marine algae because vascular plants have compounds rich in carbon (e.g., cellulose) that often breakdown more slowly than other sources (Heap et al. 2001).

The New Zealand Estuary Trophic Index (Robertson et al. 2016) designated sediment TN as a supporting eutrophication indicator with sediment TP requiring further development. Further, a review of New Zealand estuary indicators for the "Managing Upstream" project recommended sediment nutrients be developed further as a state variable (Cornelisen et al. 2017). Based on the information above, we propose the use of sediment nutrients (TN and TP) as supporting indicators, where their influence on eutrophication should be considered alongside other indicators (e.g., macroalgae, muddiness, Redox Potential Discontinuity, TOC).

PROPOSED METRICS

The following indicator metrics is proposed for monitoring sediment nutrients:

- Total sediment nitrogen (mg-N/kg of dry sediment)
- Total sediment phosphorus (mg-P/kg of dry sediment)

11.1 TOTAL NITROGEN (SEDIMENT)

Indicator type: Supporting.

Metric: Total nitrogen concentration in estuary sediment.

Unit of measurement: mg-TN/kg of total sample dry weight.

Spatial scale: Site-specific, or estuary-wide estimates made from multiple site-specific samples of sediment.

Applicability: All New Zealand estuarine and coastal waters with soft sediments, sandy to muddy, including tidal lagoon (SIDE), tidal river (SSRTRE), intermittently closing and opening lakes and lagoons (ICOLLs), and deep bay (DSDE) estuaries.

Rationale: Sediment TN content is sensitive to broad spatial and temporal changes and has been linked to symptoms of eutrophication in sediment (e.g., RPD; Sutula et al. 2014) and the subsequent health of benthic macrofauna (e.g., Walker et al. 2022). However, the degree of sediment TN lability and its availability for microbial degradation can be variable irrespective of the quantity within the sediment (i.e., TN content). Nevertheless, it can be a useful supporting indicator when used alongside other indicators of eutrophication (e.g., macroalgae, muddiness, RPD, TOC).

Method: Detailed methods for sampling sediment for nutrient content are outlined in the NEMP (Robertson et al. 2002, Stevens et al. in prep). Sediment TN samples are typically taken in surficial sediments using either surface scrapes down to 20mm using a trowel (Robertson et al. 2002) or small cores down to 20mm deep (Sutula et al. 2014). The samples are refrigerated or frozen until laboratory analysis. The recommended laboratory method involves samples being homogenised, dried and analysed via catalytic combustion (900°C in the presence of oxygen) and detection via an elemental analyser. The detection limit should be no greater than 250mg/kg.

Other methods include mass spectroscopy where TN is determined in tandem with coincident stable isotopes (e.g., $^{14}N/^{15}N$ concentrations; Hale et al. 2024), or the addition of Total Kjeldahl Nitrogen (TKN) and Total Oxidized Nitrogen (TON) which are determined through two separate extractions (i.e., sulphuric acid using copper sulfate and potassium chloride, respectively) and then the extractants are analysed on a flow injection analyser (e.g., Ellis et al. 2015).

Assessment baseline: The most ecologically relevant baseline for a site-specific monitoring of sediment TN concentration is 'natural' (pre-human) conditions. This can be determined by using hindcast methods such as deep sediment coring (and dating) to estimate baseline sediment TN content (e.g., Hale et al. 2024).

In the absence of a 'natural' state, alternatives include measuring TN in similar habitats of predominantly unmodified estuaries (i.e., native forest catchments), or measuring contemporary sediment TN content over consecutive years to determine a site baseline that can be used for future comparisons.

Measurement considerations: Detailed considerations for sampling sediment nutrient content are outlined in the NEMP revision (Stevens et al. in prep) and are briefly outlined below. When assessing long-term trends, or making spatial comparisons, it is important that the sampling approach (i.e., sample depth) and laboratory analysis method remain consistent. Artefacts from method changes can compromise the interpretation of long-term trends. The routine detection limit for TN is often relatively poor (e.g., up to 500mg/kg), therefore it is important to request the lowest possible (i.e., trace) detection limit, particularly for sandy sediments where nutrient concentrations are expected to be low.

The objective of monitoring and type of estuary will likely determine the type of sampling approach (e.g., targeted, random or stratified-random). While most SOE monitoring undertaken by councils is focused on intertidal areas, sediment TN can also be collected from subtidal sediments using remote sampling devices (e.g., Eckman grab, corers). Supporting field metadata requirements include date, time, tide height, GPS coordinates, substrate type and substrate condition (i.e., RPD). Other indicators that will likely aid in data interpretation include RPD, TOC, TP, grainsize, sedimentation rate and (where applicable) macrofauna, epifauna and surface growths of algae (macro- or micro-).

Calculation of statistic: TN is typically expressed as mg-TN/kg dry weight of sediment.

It can be converted to %TN by dividing mg-TN/kg by 100.

Potential bands and/or thresholds and rationale (including caveats): In a review of estuarine indicators for assessing the health of California estuaries, Sutula et al. (2011) concluded that sediment TN was a 'supporting indicator' meaning it fell short of meeting evaluation criteria for being a primary indicator, but it could be used in conjunction with other indicators to describe estuary health. Since, there has been limited improvement on this classification, however, a few more studies are available for threshold development.

Based on the work of Sutula et al. (2011), the New Zealand Estuary Trophic Index (ETI) proposed sediment TN as a supporting eutrophication indicator, to be used in combination with other indicators (Robertson et al. 2016). That study proposed preliminary thresholds for sediment TN content in New Zealand estuaries at both the site level, and extrapolated to spatial extent (e.g., 'Poor', >2000mg/kg TN over 10% of the estuary or >30ha). Site level thresholds for the 'Very good' classification were based on the laboratory detection limit (<250mg/kg) and the 'Fair' threshold was determined from the Sutula et al. (2014) exhaustion threshold for shallowing aRPD (~1000mg/kg TN). As no additional work has been undertaken to validate the spatial extent thresholds since they were first proposed, they have not been considered further here.

Additional studies that have proposed sediment TN thresholds are presented in Table A11-2. Thresholds have been developed through an assessment of sediment TN content on benthic habitat quality in three ways;

(1) the effect of TN on aRPD,

(2) the correlation between TOC thresholds and TN, and

(3) the relationship between macrofauna indices (e.g., abundance, richness, traits, AMBI) and TN.

Given the link between aRPD, as a habitat quality indicator, and macrofauna indices (Robertson et al. 2016) it is not surprising that the thresholds are comparable across the two approaches.

Table A11-16. Summary of sediment TN thresholds proposed in other studies.

TN (mg/kg)		Very good	Good	Fair	Poor	Very Poor
Robertson et al. (2016)	TOC/literature	<250	250-1000	1000-2000	>2000	
Sutula et al. (2014)	aRPD		500-700	700-1100	1100-1400	>1400
Walker et al. (2022)	Macrofauna (taxa & AMBI)			1200	2600	3700
Ellis et al. (2017)	Macrofauna (abundance)	185	630			
Ellis et al. (2017)	Macrofauna (traits)	245	955			
Salt Ecology unpubl.*	aRPD		~250-800	~800-1500		~3500
Salt Ecology unpubl.*	Macrofauna (richness & AMBI)		~250-800	~800-1200	>1200	
Salt Ecology unpubl.*	TOC		~500-1000	~1000-2000	~2000	

*Represents only a preliminary appraisal of data with further analysis required to confirm estimates presented here.

The key points in the studies outlined in Table A11-2 are as follows:

- Sutula et al. (2014) defined reference conditions as the physical, chemical or biological characteristics of sites found in the best available condition according to the response variable of interest. In that study, a sediment TN content of 500-700mg/kg was identified as a transition from reference conditions toward a moderately impacted state (i.e., at TN >700mg/kg the sediments are moderately impacted, a condition rating of 'Fair'), equating to a condition rating of 'Good' (Table 2). The 'slope' threshold was defined as the sharp transition to a zero slope where the response variable reached a natural limit (i.e., aRPD of 0cm). This transition toward maximum benthic degradation (i.e., aRPD of 0cm) was between 1100-1400mg/kg sediment TN content (a condition rating of 'Poor'), beyond which aRPD was anoxic to the sediment surface (i.e., >1400mg/kg TN; a condition rating of 'Very poor').

- Walker et al. (2022) identified that a sediment TN content of 1200mg/kg would have ‘reduced likelihood of supporting a desirable, intact benthic community’ (i.e., several sensitive species are lost beyond this threshold). At ~2600mg/kg most species were lost and at 3700mg/kg the sediment condition would have ‘minimal likelihood or supporting a desirable, intact community’.
- Ellis et al. (2017) assessed the effect of sediment nutrient content on taxa abundance and functional traits. That study found that taxa response to nutrient loading was often species-specific (e.g., pipi had a small tolerance range relative to cockles), however, the optima for most taxa was within the sediment TN range of 185-630mg/kg. The optima for most functional traits were in the range of 245-955mg/kg. Because these are considered optimal ranges, they have been classified as ‘Very good’ to ‘Good’ in Table A11-2. These thresholds were supported by Berthelsen et al. (2019).
- Salt Ecology has compiled a fine-scale monitoring dataset of 34 estuaries (multiple sites) from across New Zealand (Salt Ecology NZ dataset, unpubl.). Comparison of sediment TN content and aRPD (Fig. A11-1) shows a similar pattern to Sutula et al. (2014) and the threshold breakpoints approximated in Table 2. Comparison of common macrofauna indices (richness, abundance and AMBI, Fig. A11-2) shows a distinct shift in both richness and AMBI score at ~1000mg/kg TN.
- Because sediment TOC and TN are strongly correlated (Pearson R^2=0.91; Fig. A11-3), it is possible to assess the TN content corresponding to each %TOC threshold (see ‘Organic Matter indicator’ summary). These results indicate similar thresholds to those in the ETI (Robertson et al. 2016).

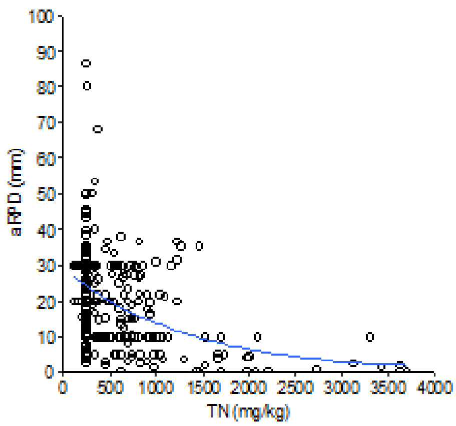

Fig. A11-1. Site average aRPD plotted against average sediment TN content with log-normal smoothing line. Salt Ecology internal fine-scale monitoring dataset of 34 estuaries.

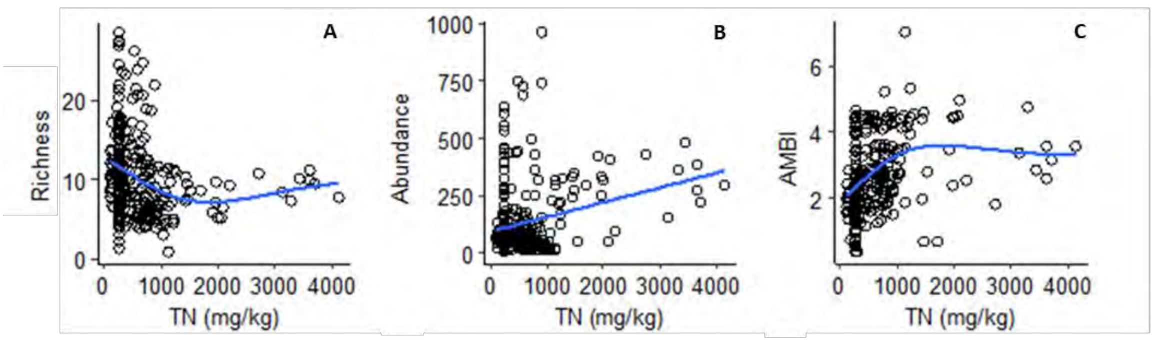

Fig. A11-2. Site average for (A) richness, (B) abundance and (C) AMBI with respect to sediment TN with a log-normal smoothing line. Salt Ecology internal fine-scale monitoring dataset of 34 estuaries.

As discussed, there are interactive effects between sediment grainsize, %TOC and sediment TN (Fig. A11-3, Salt Ecology NZ dataset unpubl.). Both %mud content (Pearson R^2=0.67) and %TOC (Pearson R^2=0.91) are strongly correlated with sediment TN. These interactions were also evident in a study of New River Estuary (Southland) where combined sediment coring, nutrient load modelling, and ecological modelling were used to hindcast ecological state (Hale et al. 2024). That study showed a significant increase in sediment TN and TOC were coincident with an increase in mud content, particularly since the late 1990's. That study also showed a shift in TN from terrestrial sources to algae, coincident with significant blooms of macroalgae since the early 2000's (i.e., TOC:TN ratio decreased). Several other indicators reacted with similar trajectories, including sediment accumulation rate, stable isotopes, modelled aRPD, seagrass decline and macrofauna health.

Overall, the findings suggest that TN impacts sediment health, though its effect can be influenced by other estuary characteristics such as grain size and TOC. This supports using TN as a supporting indicator, where its role in eutrophication should be evaluated alongside other indicators (e.g., macroalgae, TOC, muddiness, aRPD).

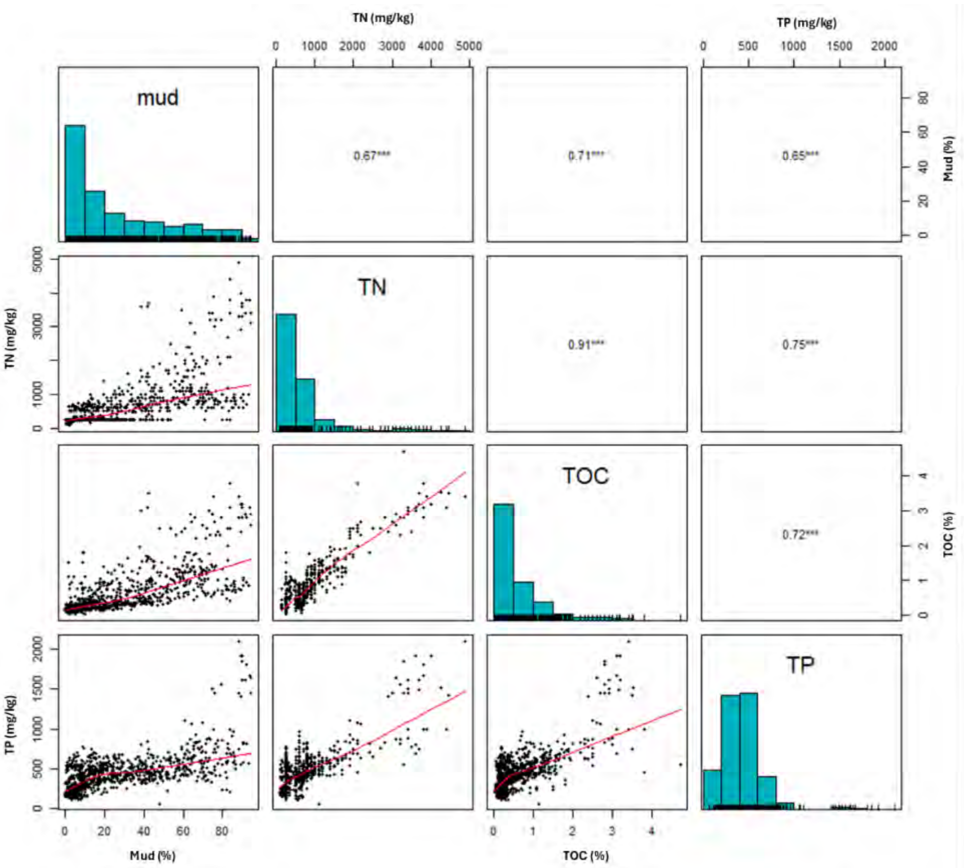

Fig. A11-3. Correlation coefficients for pairwise comparisons of key variables mud (%), sediment TN (mg/kg), sediment TOC (%) and sediment TP (mg/kg). Salt Ecology internal fine-scale monitoring dataset of 34 estuaries.

Summary of proposed thresholds:

The proposed thresholds (Table A11-3) are based on the following:

- The 'Very good' threshold of <250mg/kg is based on the ETI and supported by the lower TN range presented in Ellis et al. (2017) for macrofauna abundance and functional traits. It also aligns with the laboratory detection limit.
- The 'Good' threshold of 250-800mg/kg is based on a combination of the Ellis et al (2017) macrofauna response, Sutula et al. (2014) aRPD response, and unpublished Salt Ecology data. This represents a narrower range than the ETI thresholds presented in Robertson et al. (2016) because effects have been documented at concentrations lower than 1000mg/kg.
- The 'Fair' threshold of 800-1200mg/kg is based on a combination of Sutula et al. (2014), Walker et al. (2022) and unpublished Salt Ecology data. This represents a decrease compared to the thresholds presented in

the ETI, however, the intent of the 'Fair' rating is to represent moderate stress with a risk of sensitive macrofauna species being lost.

- The 'Poor' (1200-2000mg/kg) and 'Very poor' (>2000mg/kg) thresholds represent two levels of risk where there is the potential for severe sediment degradation (i.e., shallow aRPD) and a shift in the macrofauna community to more eutrophication tolerant species.

Table A11-17. Recommended thresholds for total nitrogen (TN) in sediment.

TN (mg/kg)	Ecological Quality Status				
	Very Good	Good	Fair	Poor	Very Poor
	<250	≥250 to <800	≥800 to <1200	≥1200 to <2000	≥2000
Narrative	Sediments are well oxygenated and ecological communities are health and resilient.	Sediments are well oxygenated and ecological communities are at low risk of losing sensitive species.	Ecological communities are at moderate risk of losing sensitive species.	Likely decrease in sediment oxygenation. Ecological communities are at high risk of a shift in macrofauna community to more tolerant species.	Poor sediment oxygenation. Ecological communities likely comprise eutrophication tolerant species.

Overall confidence in thresholds/ bands: High.

Recommendation: Total nitrogen in sediment

Adopt as preliminary numeric thresholds pending data analysis/review of New Zealand data.

Links to other indicators: Indicators that serve as explanatory variables for changes in sediment TN include grainsize or substrate type, sedimentation rate sediment quality (e.g., TOC, TP, TS, RPD), benthic macrofauna and primary producers (e.g., macroalgae, phytoplankton). Complementary stressor indicators include nutrient and sediment loads, land use types and hydrodynamic characteristics such as flushing time, tidal exchange, deposition.

Alternative metrics considered: Stable isotopes of nitrogen, TOC:TN ratio, TOC:TN:TP ratio, denitrification efficiency are related indicators and have been used internationally to assess estuary condition, there were not assessed because it was beyond scope of the current project.

Additional work recommended:

i. Collate standardised national data (and associated metadata) on sediment TN including supporting indicators (grainsize, TOC, TP, TS, RPD, macrofauna). For example, Salt Ecology have sediment TN data for ~34 estuaries (some with multiple years). It would be useful to combine this with other national datasets (e.g., Cawthron, NIWA and councils) in preparation for more comprehensive analyses.
ii. Undertake a comprehensive analysis of a national dataset to improve confidence in the preliminary thresholds proposed for TN.
iii. Strategies addressing how site-specific sampling can be scaled to estuary-wide characterisations should be developed (e.g., stratified-random designs). These approaches, with more data collection, can then be used to assess spatial thresholds for sediment TN.

11.2 TOTAL PHOSPHORUS (SEDIMENT)

Indicatory type: Indicator not endorsed

Metric: Total Phosphorus (TP) concentration in estuary sediment.

Unit of measurement: mg-TP/kg of total sample dry weight.

Spatial scale: Site-specific, or estuary-wide estimates made from multiple site-specific samples of sediment.

Applicability: All New Zealand estuarine and coastal waters with soft sediments, sandy to muddy, including tidal lagoon (SIDE), tidal river (SSRTRE), intermittently closing and opening lakes and lagoons (ICOLLs), and deep bay (DSDE) estuaries.

Rationale: While phosphorus plays a role in primary productivity and can contribute to eutrophication, very few studies have utilised sediment TP as an indicator in estuaries. This is because phosphorus is rarely a limiting nutrient in coastal environments, and, therefore, is not typically the nutrient driving eutrophic responses (e.g., algal blooms, oxygen depletion). Although some studies have shown TN and TP can be co-limiting, particularly in upper estuary environments, TP alone is not a useful indicator in estuaries.

Method: Detailed methods for sampling sediment for nutrient content are outlined in the NEMP (Robertson et al. 2002, Stevens et al. in prep). The recommended method is extraction using a combination of nitric acid and hydrochloric acid on a dry sample, with heating 85-95°C (USEPA 200.2 Digestion; Martin et al. 1994). The TP concentration in the extractant is analysed by inductively coupled plasma mass spectrometer (ICP-MS). Detection limit should be no greater than 40mg/kg.

Assessment baseline: The most ecologically relevant baseline for site-specific monitoring of sediment TP content is 'natural' (pre-human) conditions. This can be determined by using hindcast methods such as deep sediment coring (and dating) to estimate the 'natural' (pre-human) sediment TP content.

In the absence of a 'natural' state, alternatives include measuring TP in similar habitats of predominantly unmodified estuaries (i.e., native forest catchments), or measuring contemporary sediment TP content over consecutive years to determine a site baseline that can be used for future comparisons.

Measurement considerations: Detailed considerations for sampling sediment nutrient content are outlined in the NEMP revision (Stevens et al. in prep) and are briefly outlined below. When assessing long-term trends, or making spatial comparison, it is important that the sampling approach (i.e., sample depth) and laboratory analysis method remain consistent. Artefacts from method changes can compromise the interpretation of long-term trends.

The objective of monitoring and type of estuary will likely determine the type of sampling approach (e.g., targeted, random or stratified-random). While most SOE monitoring undertaken by councils is focused on intertidal areas, sediment TP can also be collected from subtidal sediments using remote sampling devices (e.g., Eckman grab, corers). Supporting field metadata requirements include date, time, tide height, GPS coordinates, substrate type and substrate condition (i.e., RPD). Other indicators that will likely aid in data interpretation include RPD, TOC, TN, grainsize, sedimentation rate and (where applicable) macrofauna, epifauna and surface growths of algae (macro- or micro-).

Calculation of statistic: TP is typically expressed as mg-TP per kg dry weight of sediment.

It can be converted to %TP by dividing mg-TP/kg by 100.

Potential bands and/or thresholds and rationale (including caveats): In a review of estuarine indicators for assessing the health of California estuaries, Sutula et al. (2011) concluded that sediment TP was a 'supporting indicator' meaning it fell short of meeting evaluation criteria for a primary indicator but could be used in conjunction with other indicators to describe estuary health. Sediment TP was also explored briefly in the ETI (Robertson et al. 2016), however, it was not recommended as a supporting indicator because TP is rarely the limiting nutrient in coastal environments and other indicators such as TN and TOC were better predictors of eutrophication.

To our knowledge the only ecological thresholds available for TP in sediments are from a study undertaken in Tauranga Harbour. In that study, Ellis et al. (2017) assessed the effect of sediment nutrient content on taxa abundance and functional traits. That study found the optima for most taxa was within the sediment TP range of 75-215mg/kg. While the optima for most functional traits was <320mg/kg. These thresholds were supported by Berthelsen et al. (2019).

Salt Ecology has compiled a fine-scale monitoring dataset of 34 estuaries (multiple sites) from across New Zealand (Salt Ecology NZ dataset, unpubl.). Using the same principles discussed under the sediment TN indicator, TP was plotted against aRPD and macrofauna indices for richness, abundance and AMBI (Fig. A11-4 & A11-5). Data showed that the relationship between TP and indicators of sediment health (i.e., aRPD and macrofauna) can be variable across a broad concentration range (Fig. A11-4 & A11-5), making it difficult to identify potential thresholds of change based on the available data.

There are interactive effects between sediment grainsize, %TOC and sediment TP (Fig. A11-3), with both %mud content (Pearson R^2=0.65) and %TOC (Pearson R^2=0.72) correlated with sediment TP. Sediment TP was also correlated with TN (Pearson R^2=0.75). Sediment TP overall was not as strongly correlated to other indicators as sediment TN (Salt Ecology NZ dataset, unpubl.; Fig. A11-3).

Overall, the findings suggest that TP is correlated with several sediment characteristics (grainsize, TOC, TN), but the relationship to sediment health indicators (i.e., aRPD and macrofauna) can be variable. At present, there is insufficient information available to set thresholds for sediment TP. This is not unexpected, given TP is rarely the limiting nutrient in coastal environments and therefore unlikely to be a driver of sediment condition. For these reasons, even with more data collection and analysis, sediment TP may never be a suitable indicator in estuaries.

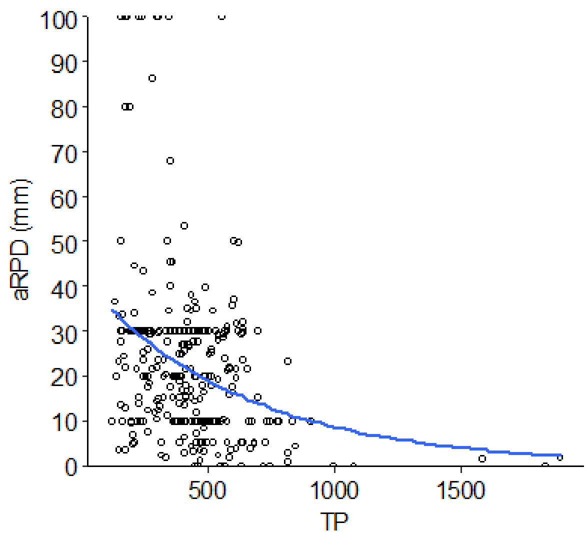

Fig. A11-4. Site average aRPD plotted against average sediment TP content with log-normal smoothing line. Salt Ecology internal fine-scale monitoring dataset of 34 estuaries.

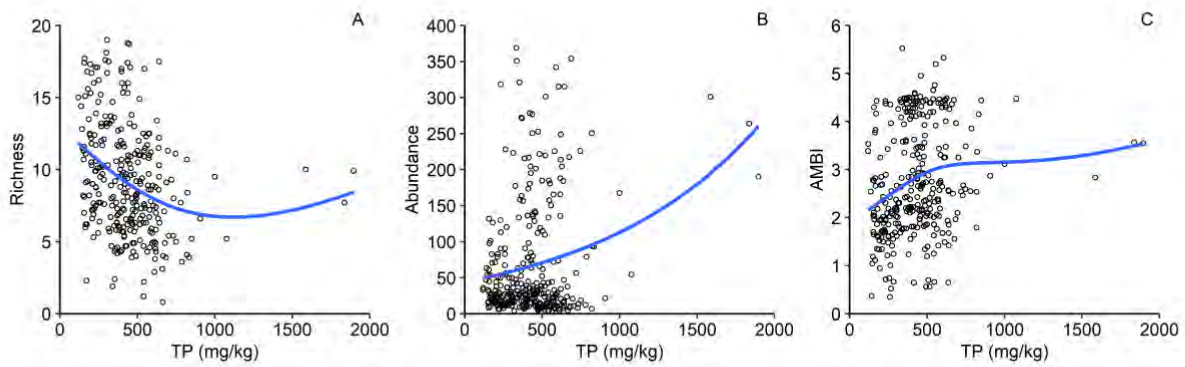

Fig. A11-5. Site average for (A) richness, (B) abundance and (C) AMBI with respect to sediment TP with a log-normal smoothing line. Salt Ecology internal fine-scale monitoring dataset of 34 estuaries.

Summary of proposed thresholds: No thresholds are proposed.

Overall confidence in thresholds/ bands: Low.

Recommendation: Total phosphorus in sediment

Further analysis/review of data from NZ and elsewhere is required to properly assess thresholds for sediment TP. A possible outcome of that analysis could be that TP is not a suitable estuarine indicator.

Links to other indicators: Indicators that serve as explanatory variables for changes in sediment TP include grainsize or substrate type, sedimentation rate, sediment quality (e.g., TOC, TN, TS, aRPD), benthic macrofauna and primary producers (e.g., macroalgae, phytoplankton). Complementary stressor indicators include nutrient and sediment loads, land use types and hydrodynamic characteristics such as flushing time, tidal exchange, deposition.

Alternative metrics considered: TN:TP ratio, TOC:TN:TP ratio, bioavailable fractions of sediment P (e.g., P-bound to iron-oxyhydroxides) are related indicators and were not assessed because it was beyond scope of the current project.

Additional work recommended:

i. Collate standardised national data (and associated metadata) on sediment TP including other indicators (grainsize, TOC, TN, TS, aRPD, macrofauna). For example, Salt Ecology have sediment TP data for ~34 estuaries (some with multiple years). It would be useful to combine this with other national datasets (e.g., Cawthron, NIWA and councils) in preparation for more comprehensive analyses.
ii. Undertake a comprehensive analysis of a national dataset to determine whether TP thresholds are suitable for use in estuaries.
iii. Strategies addressing how site-specific sampling can be scaled to estuary-wide characterisations should be developed (e.g., stratified-random designs).

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APPENDIX 12. ORGANIC MATTER

Author: John Zeldis (NIWA, Christchurch)

Organic matter in estuarine sediment includes carbon derived from plant and animal matter and is typically measured as percent Total Organic Carbon (%TOC). TOC production and decomposition of labile organic matter are at the heart of the eutrophication problem because of their influence on oxygen depletion and other biogeochemical processes including sulphide and ammonium production in sediments and overlying waters.

BACKGROUND

The rate of organic matter production and supply, and consequent microbial respiratory demand, are key elements of the estuarine eutrophication problem, associated with adverse sedimentary conditions including depleted oxygen, depressed denitrification and excessive ammonium and hydrogen sulfide concentrations (Gray et al. 2002; Hyland et al. 2005; Sutula 2011). Hyland et al. (2005) expanded upon the Pearson and Rosenberg (1978) model (which describes benthic community response along an organic enrichment gradient) by using it for defining lower and upper thresholds in %TOC concentrations corresponding to low versus high levels of benthic species richness in samples from seven coastal regions of the world. Specifically, it was shown that risk of reduced macrobenthic species richness from organic loading and other associated stressors in sediments should, in general, be relatively low where %TOC values were <1%, and relatively high where values were >3.5%.

Sediments with high %TOC are often associated with chronic macroalgal blooms which, upon decomposition, contribute locally produced (autochthonous) organic matter to sediments (Green et al. 2014; Sutula et al. 2014). High %TOC is also commonly associated with muddy, cohesive sediments (Pelletier et al. 2011; Pratt et al. 2014), which are less likely to be well irrigated compared to more permeable, sandy sediments (Huettel & Rusch 2000; Engelsen et al. 2008; Zeldis et al. 2019) leading to oxygen depletion and retention of toxic sulphide and ammonium (Sutula 2011). Excessive %TOC production and deposition can lead to shallowing of the boundary of oxic and hypoxic/anoxic conditions in the sediment profile (commonly measured using Redox Potential Discontinuity (RPD) depth: see RPD indicator), and attendant undesirable shifts in biotic community compositions in studies made in NZ (Pratt et al. 2014; Robertson et al. 2015; Stevens et al. 2022; Hale et al. 2024) and elsewhere (Pearson & Rosenberg 1978; Sutula et al. 2014).

PROPOSED METRICS

The following indicator metric is proposed for monitoring organic matter.

- Total Organic Carbon (%TOC).

12.1 TOTAL ORGANIC CARBON (%TOC)

Indicatory type: Supporting.

Metric: %TOC of total sample dry weight

Unit of measurement: %

Spatial scale: Site-specific, or estuary-wide estimates made from multiple site-specific samples of sediment.

Applicability: All New Zealand estuarine and coastal waters with soft sediments, sandy to muddy, including tidal lagoon (SIDE), tidal river (SSRTRE) (including intermittently closed cases for both), coastal lakes, and deep bay (DSDE) estuaries.

Rationale: Site-specific sampling of surficial sediment can be easily taken and can be carried out with targeted, random or stratified-random sampling across different scales of data resolution. %TOC is an accurate indicator of carbon content that is sensitive to broad spatial and temporal changes. There are reservations in its use as a general eutrophication indicator, related to degree of organic matter lability and its availability for microbial degradation (Sutula 2011). Irrespective, %TOC has been routinely assayed in estuarine studies in New Zealand (e.g., Pratt et al. 2014; Hale et al. 2024) and elsewhere (e.g., Hyland et al. 2005; Pelletier et al. 2011; Sutula et al. 2014) often allowing an understanding of its ecological role and attendant thresholds of impact. Temporal and spatial changes in %TOC can be assessed by repeat mapping and related to the effects of anthropogenic inputs of nutrient and sediment, reclamation, or physical disturbance.

Method: Analytical methods for determining sediment %TOC are standardised, using combustion techniques via elemental analyser (flash combustion; Verardo et al. 1990). TOC has also been determined using mass spectroscopy in tandem with coincident determinations of stable isotope (^{13}C, ^{15}N concentrations: Zeldis et al. 2019; Hale et al. 2024). TOC:TN ratios may be useful to determine degree of lability of the organic matter (Heap et al. 2001).

Measurement considerations: %TOC samples are typically taken in surficial sediments using small cores, 1-2cm in depth (Sutula et al. 2014; Zeldis et al. 2019), at sites.

Calculation of statistic: %TOC is typically expressed as the percent of TOC per unit dry weight of sediment.

Potential bands and/or thresholds and rationale (including caveats): Sutula (2011) reviewed a suite of indicators for development of numeric endpoints for nutrients and other management controls in California estuaries. They concluded that as an indicator of eutrophication, organic carbon was likely to have indirect linkages to management. They nominated it as a 'supporting indicator' meaning it fell short of meeting evaluation criteria (including signal to noise ratio) but may be used as a supporting line of evidence. Some of the shortcomings involved uncertainty in the sensitivity of the indicator *vis a vis* primary producers (macroalgae vs microalgae), or introduced by confounding factors including sediment morphology (muddiness/sandiness) and the lability of the organic matter. Sutula (2011) suggested that ratios of TOC with total sulphur or degree of pyritization could be better indicators, citing work in Australia where those indicators had been used to classify degrees of eutrophication. They recommended that experiments and field work be done to address these knowledge gaps and synthesise findings into an assessment framework.

There have been subsequent studies which appear to increase confidence in use of %TOC as a eutrophication indicator. Sutula et al. (2014) derived relationships between macroalgal biomass and measures of sediment %TOC and apparent RPD (aRPD) depth, measured at 16 sites in eight California estuaries. They showed a 'step' threshold of aRPD depth at < 0.5%TOC, below which the effect on aRPD depth was at a 'reference' (low impact) level (aRPD > ~4cm from the sediment surface, indicating healthy oxic conditions). At the other end of the scale, they showed a 'slope' threshold of 1.1-1.2%TOC, where 'exhaustion' levels of impact on aRPD depth were first detected (aRPD depth approaches zero cm from the sediment surface). They also noted that aRPD depth had a closer relationship with %TOC values than with macroalgal biomass, which they attributed to the former indicator's more intimate relationship with the diagenetic processes that determine aRPD.

Analysis of TOC sediment data collected in the EMAP-Virginian Province Study (USA) (Paul et al. 1999) indicated that %TOC in the 1 to 3% range was associated with impacted benthic communities, while values less than 1% were not. Pelletier et al. (2011), using EMAP datasets, identified unimpacted 'reference' conditions for healthy sediments at %TOC levels of 0.2 to 0.9% across hundreds of east coast USA estuary samples, with clear increasing associations of %TOC with %mud. Their data supported the hypothesis that sites designated as enriched (high %TOC) were eutrophied: dissolved oxygen levels were reduced and sediment chlorophyll-*a* and nutrients were higher at enriched sites, suggesting that the relationship of organic carbon to grain size can be used as a screening tool to diagnose eutrophication.

In New Zealand, Robertson et al. (2016) investigated interactions of grain size (muddiness) and %TOC with macrobenthic health (measured as locally-calibrated AMBI). They developed regression trees that identified threshold values of %mud and %TOC delimiting macrobenthic taxon abundance and richness at 21 NZ tidal lagoon and tidal river estuaries. They identified thresholds ranging from 'Normal' to 'Transitional to pollution', to 'Polluted' with increasing % mud. Percent TOC was only important as a criterion for abundance and richness indices if %mud was high (>~34%). This corresponded with %TOC exceeding 1.2% within the 'Transitional to pollution' band but indicated a dominant effect of muddiness below 34% mud. Piecewise regression performed on the TOC distribution data suggested an additional breakpoint in AMBI at ~3%TOC with most scores beyond this threshold fitting the 'Polluted' condition. The analysis indicated that %TOC had the strongest association with AMBI in elevated mud and TOC situations.

An example showing interactive effects between grain size, macroalgae and %TOC was Zeldis et al. (2019) on the Avon-Heathcote Estuary (Christchurch), showing that while the estuary was highly eutrophic due to outgrowths of macroalgae and benthic microalgae (before the diversion of its wastewater inputs to an ocean outfall), its sediments were sandy and supported low %TOC throughout. Another example was that of Hale et al. (2024), who used sediment coring, nutrient load modelling and ecological modelling to hindcast ecological state in New River Estuary (Southland). They showed clear increases of %TOC with increasing muddiness through the 20th century (related to catchment land use intensification mainly from sheep and beef farming), followed by steep increases in %TOC in the late 20th and early 21st centuries associated with intensified dairy farming and rapid increases in macroalgal biomass. Several other indicators reacted with similar trajectories, including sediment accumulation rate, isotopic compositions, and modelled aRPD depth, seagrass decline and macrobenthic health.

Taken together, the findings above indicate an operative effect of TOC on sediment health, but one that is conditioned by other estuary characteristics including grain size and primary biomass, potentially operating independently or in concert. This supports use of %TOC as a supporting indicator, wherein its influence in driving eutrophication should be considered along with other indicators (including macroalgae, muddiness, and RPD depth). Such interactions were built into the Bayesian network analysis of estuary trophic health by Zeldis and Plew (2022).

Summary of proposed thresholds: The %TOC thresholds given here (Table A12-1) were assigned four bands, to be consistent with the four-band scoring system used for secondary indicators in the ETI (Zeldis & Plew 2022; Hale et al. 2024) and the National Policy Statement for Freshwater Management (New Zealand Government 2020). The banding uses results of Sutula et al. (2014) who described %TOC expected to elicit effects ranging between 'reference' (<0.5 %TOC) and 'exhaustion' (>1.2 %TOC) impacts, and by considering %TOC effects on eutrophication indicators (Paul et al. 1999; Pelletier et al. 2011) and macrobenthos (Robertson et al. 2016). Hale et al. (2024) showed steadily worsening of several ecological health indicators through band C in the mid-to-late 20th century, reaching band D in the late 20th and early 21st centuries, when %TOC crossed the 2.0% threshold. Banding above 2% was also allocated accounting for the relationships between macroalgal biomass and %TOC shown in Figure 6 of Sutula et al. (2014), where this level of %TOC occurs at high macroalgal biomass, and is intermediate with respect to the 1-3%TOC range found by Paul et al. (1999) to associate with impacted benthic communities. Addition of a fifth band (Severe, band E), with %TOC >3.5%, to resolve extremely degraded conditions more fully, could be considered.

Table A12-1: Recommended %TOC thresholds for New Zealand estuaries.

% TOC	Ecological Quality Status			
	Very Good (A)	Good (B)	Fair (C)	Poor (D)
	<0.5%	0.5 to 1.2%	>1.2% to 2%	>2%
Narrative	No to minor stress on sensitive organisms.	Moderate stress on some species and a risk of sensitive macroinvertebrate species being lost.	Significant, persistent stress on a wide range of aquatic organisms.	A likelihood of local extinctions of keystone species and loss of ecological integrity.

Overall confidence in thresholds/ bands: High.

Recommendation: Total Organic Carbon – TOC

Adopt as preliminary numeric thresholds pending analysis/review of data from NZ and elsewhere.

Links to other indicators: As discussed above, other commonly measured indicators serve as explanatory variables for changes in %TOC including sediment grain size, aRPD depth and abundance of primary producers. Several of these links have been parameterised within the Bayesian network model of Zeldis and Plew (2022).

Alternative metrics considered: Organic matter can also be measured as Loss On Ignition (LOI). Pribyl (2010) provided an approximate conversion between %TOC and LOI (~ %TOC x 2 = %LOI), although he points out issues with lower reliability of LOI than direct TOC measures. LOI seems less useful than %TOC, due to lack of reliable comparisons of LOI with ecological responses in literature, which focus mainly on %TOC.

Additional work recommended:

i. Supporting data needs include nutrient and sediment loads, and macroalgal, microphytobenthic and RPD indicator monitoring along with %TOC. These should be done across estuary types where appropriate. Sampling strategies addressing how site-specific sampling can be scaled to estuary wide characterisations should be developed (e.g., stratified-random designs).

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APPENDIX 13. REDOX POTENTIAL DISCONTINUITY (RPD)

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The Redox Potential Discontinuity (RPD) is the boundary between oxic near-surface sediment and the underlying suboxic or anoxic sediment. As an indicator of ecosystem condition, an RPD close to the sediment surface has been related to reduced volume and quality of habitat for benthic infauna and alteration in community structure. These effects have been linked to reduced availability of forage for fish, birds and invertebrates, as well as to undesirable changes in biogeochemical cycling.

BACKGROUND

The boundary where conditions in the seabed or estuary sediment change from oxidizing to reducing is termed the Redox Potential Discontinuity (RPD). Above the RPD, the sediment is oxygenated. Shallowing of RPD is related to deleterious alterations in macrobenthic community structure (Sutula et al. 2014) and undesirable changes in sedimentary biogeochemical cycling (Eyre & Ferguson 2009; Sutula 2011). As reviewed by Walker et al. (2022), the production and accumulation of labile organic matter associated with eutrophication stimulates heterotrophic bacterial communities in sediments, increasing water column and benthic oxygen demand while simultaneously decreasing sediment redox potential. Zones of sediment anoxia and sulfate reduction become shallower across the sediment horizon, sometimes extending to the sediment–water interface. This leads to an increase in pore water ammonia and sulfide concentrations that are toxic to benthic fauna. Tolerance to changes in organic matter concentrations and low oxygen conditions varies widely among the taxonomically diverse macrobenthic community, though persistent anoxic conditions will eventually kill all metazoans.

Pearson and Rosenberg (1978) suggested four major stages of change in response to organic enrichment of the benthos along a gradient of organic enrichment: (1) Normal: (RPD deep below the sediment surface) characterised by the presence of large, deep-burrowing species such as decapods and echinoids, (2) Transitory: (RPD moderate depth below the sediment surface) characterised by the presence of smaller organisms usually deposit-feeding bivalves, which replace the large deep-burrowing species, (3) Polluted: (RPD very close to the sediment surface) characterised by a very shallow RPD depth and dominance of small tube-building polychaetes; indicating severe eutrophication, and (4) Grossly Polluted: (RPD and sulphide patches at the sediment surface) no macrofauna, with only nematodes surviving.

The RPD can be located visually (e.g., by using *in situ* digital Sediment Profile Imaging (SPI) cameras or transparent cores) as the level where sediment first changes colour to grey/green or black. The colour change is due to, in the absence of oxygen, microbial sulphate reduction resulting in the precipitation of Fe-sulphides and producing a grey/green or black sediment colouration. Where the RPD is identified visually, the depth to the RPD is termed aRPD depth

Alternatively, the RPD can be located by measuring redox potential (Eh) with a probe (here, termed pRPD), where Eh represents a composite of multiple redox equilibria measured at the probe surface (Rosenberg et al. 2001). The electrode is inserted at different depths into the sediment (usually using cores) and the extent of reducing conditions at each depth recorded. The RPD is inferred from the change in measured Eh through the sediment column; typically, the RPD is taken as the level at which Eh redox potential first undergoes rapid decline into the range -100 to -150mV, which indicates long term, heavy organic carbon pollution (Pearson & Stanley 1979). As noted for the % TOC indicator (Appendix 12), such conditions are associated with adverse sedimentary conditions including depleted oxygen, depressed denitrification and excessive ammonium and hydrogen sulfide concentrations (Gray et al. 2002; Hyland et al. 2005; Sutula 2011).

PROPOSED METRICS

The aRPD method has been preferred to the pRPD method in several studies. Marked core-to-core variability and inconsistency between aRPD and pRPD methods have been described in published studies that have compared the visual and probe methods for identifying the RPD (Forrest & Creese 2006; Gerwing et al. 2013) as well as in several recent NEMP surveys (e.g., Forrest et al. (2021)). Gerwing et al. (2013) discussed how redox (Eh) electrodes measure the instantaneous redox potential of the sediment, which can be very variable. In contrast, the visual method represents an integrated long-term average of redox conditions, which is likely to be of greater interest in estuary health monitoring programmes. Rosenberg et al. (2001) concluded that aRPD imagery afforded the additional advantages of the ability to visualise the sediment colour and faunal profile, as well as introducing less likelihood of disturbing the sediment-column redox pattern.

The variation often found between aRPD and pRPD depths likely reflects the occurrence of oxic zones throughout the core profile, such as caused by the mixing of surface and deeper sediments by bioturbation, and where it is a matter of chance whether the Eh probe encounters these areas when it is inserted. Also, if core holes become part-flooded, the infiltration of ambient water will influence readings, necessitating removal of the core from the sediment. In sandy sediments, this can drain the core, making it too dry for a reliable Eh reading. These methodological issues undermine the utility of the probe method, at least for routine field monitoring purposes (L. Stevens, B. Forrest, Salt Ecology pers. comm. May 2024; Forrest et al. (2021)).

Sutula (2011) reviewed a suite of indicators for development of numeric endpoints for nutrients and other management controls in California Estuaries. Although they did not evaluate RPD directly as an indicator, they did discuss how in cohesive sediments molecular fluxes tend to be less than in larger grain sized sediments and these sediments tend to have redox profiles reflecting high oxygen demand and potential for sediments to become anoxic at very shallow sediment depth. In contrast, sandy, non-cohesive sediments are permeable and interstitial water movements increase the transport rates of oxygen and other solutes including dissolved organic matter by orders of magnitude. In such sandy situations, it may be difficult to clearly identify an aRPD, and Eh probe measures may be called for. In highly cohesive muds (for example, silts) conditions may occur where the sediment profile exhibits very shallow RPD but is in fact not loaded with significant organic matter (i.e., is not eutrophic) a feature seen in some NZ estuaries (e.g., Whanganui Estuary (Horizons region: Forrest et al. (2021); (L. Stevens, B. Forrest, Salt Ecology pers. comm. May 2024)). In addition, the exchange between water column and sediment is also influenced by benthic fauna that contributes to bioturbation and bio-irrigation of the sediment which can also complicate determination of the RPD depth.

Taken together, the findings above indicate an operative effect of depth of the RPD on sediment health, but one that is conditioned by other estuary characteristics including grain size, organic content, and primary producer and faunal community compositions, potentially operating independently or in concert. This supports use of depth of the RPD as a supporting indicator, wherein its influence in indicating eutrophication should be considered along with other indicators (including macroalgae, grain size, % TOC and macrofaunal composition).

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

The following indicator metric is proposed for monitoring RPD:

Depth from sediment surface to the RPD

- aRPD determined by sediment profile imagery SPI (preferred) or visually,
- pRPD determined by measuring redox potential (Eh) with a probe.

13.1 REDOX POTENTIAL DISCONTINUITY (RPD)

Indicator type: Supporting.

Metric: The vertical distance (depth) between the RPD and the sediment surface. Using a visual method (aRPD) the RPD is depth where the sediment profile first changes colour from brown to grey-green or black (e.g., Forrest et al. 2021). Using the probe method (pRPD), the RPD is the depth where Eh first undergoes rapid decline into the range -100 to -150mV.

Unit of measurement: mm.

Spatial scale: Site specific. Estuary-wide estimates obtained from statistics on multiple site-specific samples, or within spatial strata (if a stratified survey design is used).

Applicability: All New Zealand estuarine and coastal waters with soft sediments, sandy to muddy, including tidal lagoon (SIDE), tidal river (SSRTRE) (including intermittently closed cases for both), coastal lakes, and deep bay (DSDE) estuaries.

Rationale: Sampling can be carried out with targeted, random or stratified-random sampling across different scales of data resolution. This is potentially important where fine scale horizontal patchiness occurs in RPD depth (as described by Rosenberg et al. 2001).

Methods: Visual methods for determining aRPD depth have commonly been carried out using SPI camera and computer digitisation of colour areas (Sutula et al. 2014), or from direct visual assessment. For Eh probe determinations, measures are made using a redox potential electrode coupled to a millivolt meter (often called an ORP meter) that detects whether the sediment tends to receive or donate electrons. The electrode is inserted to different depths into the sediment and the extent of reducing conditions at each depth recorded.

Measurement considerations: Intertidal, site-specific sampling of sediments is straightforward using SPI, or transparent cores and camera. For Eh probe determinations coring is often required. For subtidal sampling, a vessel is required for both methods and ability to deploy the equipment. For Eh profiling, sampling can be complicated by the issues described above for the pRPD method (Proposed metrics section).

Calculation of statistic: Expressed as an average value of the depth of the RPD across estuary zone of interest.

Potential bands and/or thresholds and rationale (including caveats):

Thresholds inferred from sediment organic carbon

Sutula et al. (2014) used sediment profile imagery at 16 sites across eight California estuaries, to identify thresholds of adverse effects of macroalgal biomass, sediment organic carbon (% TOC) and nitrogen (% N) concentrations on aRPD depth. They showed that aRPD depth decreased linearly with % TOC increases until a 'break point' beyond which further increases in % TOC caused no further decrease in aRPD depth (Sutula et al. (2014): their Figure 5). This relationship can be written as:

aRPD depth =5.8−3.84 %*TOC* (Zeldis & Plew 2022).

This relationship showed aRPD depth levelling off at ~1.1cm which Sutula et al. (2014) interpreted as an 'exhaustion' threshold of aRPD depth where severe adverse effects occur. Rounding this, Zeldis and Plew (2022) set a C-D band threshold at aRPD depth <1cm. Sutula et al. (2014) also defined a 'cut value' for %TOC which separated reference ('unimpacted') sites from non-reference sites of 0.46%TOC (their Figure 3). Using the above equation, this equated to an aRPD depth of ~ 4.0cm, representing aRPD depth at the reference (A-B band) threshold (cf. Figure 4 of Sutula et al. (2014)).

Sutula et al. (2014) described how controls on aRPD formation are complex, responding to a variety of driving factors. These included overlying water oxygen concentrations, bioturbation, sediment %TOC, carbonate and iron content, physical energy, all of which vary temporally and spatially within estuarine sediments. However, they considered that their estimated 'exhaustion' threshold in organic matter accumulation appeared to override other factors controlling aRPD depth, driving it to near zero levels. They considered that the ranges associated with 'reference' and near zero aRPD depth represented 'bookends' of a gradient of increasing organic matter loading along which increasing adverse effects can be documented. These values were in concordance with studies documenting macroalgal biomass effects on aRPD depth, including work of the European Union Water Framework Directive (EU WFD; (Scanlan et al. 2007)) and in California (Green et al. 2014), at their 'reference' and 'exhaustion' thresholds, respectively. In terms of %TOC effects on aRPD depth, Pelletier et al. (2011) predicted subtidal impairment and enrichment thresholds at values above 1–1.5%TOC for the three Atlantic Coast regions, agreeing well with the 'exhaustion' thresholds of 1.1–1.2%TOC found in Sutula et al. (2014). Pelletier et al. (2011) also defined a 'reference' envelope of %TOC at 0.2–0.9%, values that also agreed well with the 0.2–0.7%TOC 'reference' transition range identified by Sutula et al. (2014).

Thresholds inferred from macrofaunal habitat quality indices

Nilsson and Rosenberg (2000) related SPI-determined aRPD depths to Benthic Habitat Quality (BHQ) indices for macrofauna. The resulting values of aRPD depth and BHQ broadly reflected the range and discretisation of aRPD depth determined above (ranging between 0 and 5cm, corresponding to BHQ indices between 0cm (extremely impacted) and 5cm (healthy bioturbating communities) with BHQ cutoffs at 1, 2 and 3.5cm. That study was, however, made in deep Swedish coastal waters (60 – 118m depths) in very muddy sediments, unlike many applications likely in New Zealand. In a shallower coastal system (~1.0m low water depth) in an eastern USA embayment Grizzle and Penniman (1991) used SPI (acrylic box core) and Eh probe-determined RPD depth showing values ranging from <1cm at sites closest to pollution sources with impacted macrofauna, to ≥4cm at sites farthest away, where macrofauna included healthy deep-burrowing species. In this case, the Eh probe-determined measurements compared well with aRPD from the box cores, with the colour discontinuity occurring between -110 and -150mV, consistent with Pearson and Stanley (1979) (although, see 'Alternative metrics considered' section).

Summary of proposed thresholds: The depths of RPD thresholds given here (Table A13-1) were assigned four bands, to be consistent with the four-band scoring system used for secondary indicators in the ETI (Zeldis & Plew 2022; Hale et al. 2024) and the National Policy Statement for Freshwater Management (New Zealand Government 2020). The thresholds employ the 'reference' (A-B) and 'exhaustion' (C-D) thresholds of Sutula et al. (2014), with the addition of an intermediate B-C threshold set at 25mm, being the midpoint between 'reference' and 'exhaustion' RPD depths (Zeldis and Plew (2022)) and roughly consistent with intermediate thresholds of Nilsson and Rosenberg (2000). Addition of a fifth band (Severe, band E), to resolve extremely degraded conditions more fully, could be considered.

Table A13-1. Recommended RPD depth thresholds for New Zealand estuaries.

RPD depth (mm)	Ecological Quality Status			
	Very Good (A)	Good (B)	Fair (C)	Poor (D)
	>40	40 to >25	25 to 10	<10
Narrative	No to minor stress on sensitive organisms.	Moderate stress on some species and a risk of sensitive macroinvertebrate species being lost.	Significant, persistent stress on a wide range of aquatic organisms.	A likelihood of local extinctions of keystone species and loss of ecological integrity.

Note that the thresholds are in mm rather than cm, to improve measurement resolution between bands.

Overall confidence in thresholds/ bands: High.

Recommendation: Redox Potential Discontinuity - RPD

Adopt as preliminary numeric thresholds pending analysis/review of data from NZ and elsewhere.

Links to other indicators: As discussed above, other commonly measured indicators serve as explanatory variables for changes in depth of RPD including sediment grain size, %TOC and compositions of primary producer and macrofaunal communities. Several of these links have been parameterised within the Bayesian network model of Zeldis and Plew (2022).

Alternative metrics considered: As described above, the RPD can be identified using either electrode based (Eh) probe or visual methods using SPI or transparent cores, although the visual methods are preferred in most cases.

For cases where the RPD is either indistinct or unlikely to indicate eutrophication in spite of near zero aRPD depth (as discussed above) consideration could be made of a narrative metric which simplifies the distinctions between moderately impacted sediments and those severely impacted, i.e., simply whether or not the sediment has an intense black colour throughout the entire profile, smells of sulphide, and possibly has *Beggiatoa* or intense microalgal growth on the surface (i.e., shows obvious symptoms of strong enrichment). This would also accommodate cases where severe eutrophication has occurred with complete loss of ecological integrity (i.e., RPD depth = zero).

Additional work recommended:

i. Supporting data needs to include nutrient and sediment loads, and macroalgal, microphytobenthic biomass, %TOC and macrofaunal monitoring, along with depth of RPD. Work to determine estuary health responses at depth of RPD values intermediate between ~10 and 40mm (i.e., B-C threshold of Table A13-1) would be useful to firm up thresholds.

ii. Additional work should be done across estuary types where appropriate. Sampling strategies addressing how site-specific sampling for RPD depth can be scaled to estuary wide characterisations should be developed (e.g., stratified-random designs).

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APPENDIX 14. SEDIMENT SULPHUR

Author: Keryn Roberts (Salt Ecology)

Depletion of oxygen in sediments is a symptom of eutrophication and has been linked to nutrient loading. During prolonged periods of anoxia, sulphate reduction becomes significant in the breakdown of organic matter, producing, hydrogen sulphide, elemental sulphur and sulphur-iron minerals. The subsequent build-up of porewater sulphide under prolonged anoxia can be toxic to benthic macrofauna. As such, "sediment sulphur indicators" (e.g., TOC:TS, degree of pyritization and acid-volatile sulphide), can be good proxies for sulphate reduction and provide useful insights into the persistence and severity of sediment anoxia, a eutrophication symptom.

BACKGROUND

Sulphur is an important macronutrient for plants, plays a role in the breakdown of organic matter, is important in the biogeochemical cycling of nutrients and can determine the speciation, bioavailability and toxicity of heavy metals (Sutula 2011; Jasińska et al. 2012).

In estuaries, eutrophication leads to excessive organic matter deposition, and its decomposition decreases sediment oxygenation, resulting in the accumulation of toxic compounds (e.g., ammonia, sulphides) in porewater which can have negative effects on benthic macrofauna (Walker et al. 2022). Several sediment indicators (e.g., RPD, TOC, TN) have been used to characterise eutrophic sediments and associated changes in macrofauna communities (Sutula et al. 2014; Walker et al. 2022). Additionally, 'sediment sulphur indicators' (e.g., TOC:TS, degree of pyritization and acid-volatile sulphide) can be used to provide further insight into the persistence and severity of sediment anoxia, a eutrophication symptom (Sutula 2011).

When surficial sediments are well oxygenated, organic matter is broken down primarily by aerobic respiration, in which oxygen is used as the primary electron acceptor. Under these conditions there is a natural gradient of oxygen depletion and simultaneous decrease in redox potential (i.e., conditions become more reducing) with increasing sediment depth. As oxygen becomes depleted down the sediment profile, other electron acceptors come to be sequentially utilised in the breakdown of organic matter, resulting in the gradual replacement of aerobic respiration (well oxygenated sediments near the surface) to anaerobic respiration (poorly oxygenated and anoxic sediments lower down in the sediment; Fig. A14-1; Aller 2014).

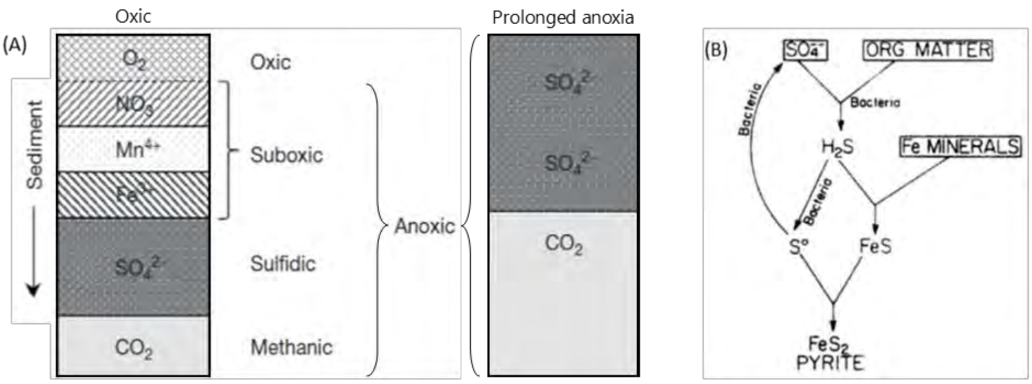

Fig. A14-1 (A) Representation of oxic and anoxia sediments (modified from Aller 2014). In the absence of oxygen, other electron acceptors are used in the decomposition of organic matter. (B) Decomposition of organic matter using sulphate (SO_4^{2-}) as the electron acceptor (from Berner 1984). The hydrogen sulfide (H_2S) byproduct reacts with iron (Fe) minerals to form iron sulphides (FeS and pyrite; FeS_2).

The increased accumulation of labile organic matter associated with eutrophication leads to increased rates of oxygen consumption and the zone of oxygen depletion (i.e., anoxic zone; Fig. A14-1 A) rises closer to the sediment surface. Under prolonged anoxia, a sulfidic zone (Fig. A14-1 A) can extend to the sediment surface. In the sulfidic zone, organic matter is broken down by sulphate reduction, in which sulphate is the primary electron acceptor. Sulphate reduction leads to the production of toxic hydrogen sulphide (H_2S), which can remain in the porewater or react with iron minerals, forming iron monosulfide, an unstable compound, that is usually converted over time to a more stable form, pyrite (FeS_2; Fig. A14-1 B; Berner 1984). While lack of sulphate can limit sulphate reduction in freshwaters, it is a major constituent of seawater and is therefore abundant in estuarine environments. Further, the link between the iron and sulphur cycles means that the amount of reactive iron in the sediment can also regulate sulphur speciation (Sutula 2011).

Sulphate reduction in sediments is generally not measured directly because it requires costly and more complex approaches (e.g., porewater extraction in the absence of oxygen, $\delta^{34}S$ stable isotopes) making it impractical for routine council monitoring. Conversely, sediment total sulphur (TS) content alone is not a useful indicator because the amount of sulphur can be influenced by factors such as source, redox condition and grainsize. Therefore, TS requires contextual information to be useful. As such, international monitoring programmes use other 'sediment sulphur indicators' as proxies for sulphate reduction, including:

(1) TOC:TS ratio - High rates of sulphate reduction under anoxic conditions leads to the accumulation of sulphide minerals (e.g., pyrite; Fig. A14-1 B) and subsequently higher concentrations of total sulphur in the sediment. As such, a low TOC:TS ratio can indicate persistent anoxic conditions.

(2) Degree of pyritization (DOP) - Iron in the sediment can regulate sulphur speciation. Understanding the amount of pyrite formed relative to reactive iron can provide an indication of sediment condition and persistent anoxia (i.e., if a large amount of pyrite has been formed relative to reactive iron it suggests conditions have been anoxic and sulfidic for some time).

(3) Acid-volatile sulphide (AVS) – Represents the fraction of sulphide as hydrogen sulphide (H_2S) gas and iron monosulfide (FeS) and is associated with the bioavailability of some metals including cadmium, copper, lead, nickel and zinc (Hall & Anderson 2022). Several international studies have debated its usefulness as an indicator (e.g., Rickard & Morse 2005; Sutula 2011; Hall & Anderson 2022). These issues are unresolved, so we do not discuss AVS further here.

Several co-factors control the importance of sulphur in the decomposition of organic matter (Sutula 2011). For example, the indicators listed above are reliant on:

(1) Sediment redox state (i.e., prolonged anoxia is required to produce pyrite; Berner 1984; Aller 2014).

(2) Availability of dissolved sulphate, which is not limiting in estuaries (Sutula 2011).

(3) Pool of reactive iron minerals in the sediment (Hedges & Keil 1995).

(4) Amount of reactive organic matter (Berner 1984).

These factors can also be altered by bioturbation and subsequent oxygenation of sediments by macrofauna, sedimentation rates, and/or reactivity of iron in the sediment (Sutula 2011). For these reasons Sutula (2011) recommended 'sediment sulphur indicators' be used as supporting indicators only. Similarly, 'sediment sulphur indicators' are only used qualitative indicators of sediment redox condition in Australian estuary monitoring (ozcoasts.org.au). Further, because sulphate reduction occurs in surficial sediments after prolonged anoxia, it represents severe levels of degradation in the sediments and significant stress on benthic macrofauna. Therefore, while it can be used to characterise the persistence and severity of anoxia, other indicators (e.g., RPD, TOC, TN) will likely detect sediment degradation, caused by eutrophication, prior to 'sediment sulphur indicators'.

PROPOSED METRICS

The following indicator metrics are proposed for monitoring sediment eutrophication.

1. TOC:TS
2. Degree of Pyritization (DOP)

14.1 TOC:TS

Indicator type: Supporting.

Metric: Ratio between %TOC and %TS.

Unit of measurement: no units.

Spatial scale: Site specific

Applicability: All New Zealand estuarine and coastal waters, including tidal lagoon (SIDE), tidal river (SSRTRE), intermittently closing and opening lakes and lagoons (ICOLLs), and deep bay (DSDE) estuaries.

Rationale: TOC:TS ratio serves as a proxy for sulphate reduction, and is a qualitative indicator of severe and prolonged eutrophication in estuarine sediments (Sutula 2011; ozcoasts.org.au). Despite the influence of several co-factors on sulphate reduction and the decomposition of organic matter, it can be a useful supporting indicator when used alongside other indicators of eutrophication (e.g., macroalgae, muddiness, RPD, TOC), particularly in instances of persistent anoxia.

Method: Detailed methods for sampling sediment for TOC and TS content are outlined in the NEMP (Robertson et al. 2002, Stevens et al. in prep). Sediment TOC and TS samples are typically taken in surficial sediments using either surface scrapes down to 20mm using a trowel (Robertson et al. 2002) or small cores down to 20mm deep (Sutula et al. 2014). The samples are refrigerated or frozen until laboratory analysis.

The recommended laboratory method for %TOC is acid pre-treatment to remove carbonates followed by catalytic combustion (~1050°C with oxygen), the carbon dioxide released is analysed via an elemental analyser (Verardo et al. 1990). The recommended laboratory method for %TS is combustion (~1350°C) in oxygen rich environment and the sulphur dioxide released is analysed via infra-red detector (ASTM Method 4329).

Assessment baseline: The TOC:TS ratio is an indicator of oxygen status which can be variable over time therefore a 'natural' (pre-human) baseline is unlikely to be suitable in this instance. However, hindcast methods such as deep sediment coring (and dating) could be used to assess qualitative changes in oxygen status over time (e.g., Akhil et al. 2013). For example, this information could be used to assess the frequency and duration of low sediment oxygen conditions historically.

Measurement considerations: When assessing long-term trends, or making spatial comparison, it is important that the sampling approach (i.e., sample depth) and laboratory analysis method remain consistent. Artefacts from method changes can compromise the interpretation of long-term trends.

The objective of monitoring and type of estuary will likely determine the type of sampling approach (e.g., targeted, random or stratified-random). While most SOE monitoring undertaken by councils is focused on intertidal areas, sediment, TOC:TS can also be collected from subtidal sediments using remote sampling devices (e.g., Eckman grab, corers). Supporting field metadata requirements include date, time, tide height, GPS coordinates, substrate type and substrate condition (i.e., RPD). Other indicators that will likely aid in data interpretation include RPD, iron, TN, TP, grainsize, sedimentation rate and (where applicable) macrofauna, epifauna and surface growths of algae (macro- or micro-).

Calculation of statistic: TOC:TS ratio is typically expressed as the ratio of %TOC and %TS per unit dry weight of sediment.

Potential bands and/or thresholds and rationale (including caveats): International studies have shown that well oxygenated sediments typically have a TOC:TS ratio >5 (Sutula 2011), while sediments undergoing sulfate reduction below an oxygenated water column typically have a TOC:TS ratio in the range of 1.5 to 5 (Berner 1984; Hedges & Keil 1995; Sutula 2011; Akhil et al. 2013). Under severe eutrophic conditions, where the water column is anoxic and sulfidic (i.e., euxinic) and sediments are undergoing high rates of sulfate reduction, the TOC:TS ratio is <1.5 (Berner 1984; Sutula 2011).

The TOC:TS ratio does not apply in the presence of iron limitation (i.e., H_2S does not react with iron and instead diffuses upward and is rapidly oxidised meaning the TS concentration remains low as it is not bound in iron sulphides) or when TS concentrations are low (Raiswell et al. 1987). Because of these factors Sutula (2011) recommended that it would be more appropriate to use TOC:TS in depositional habitats, where concentrations of both iron and TS are likely high.

Summary of proposed thresholds:

Table A14-1: Recommended TOC:TS thresholds for New Zealand estuaries.

TOC:TS	Ecological Quality Status (TOC:TS)		
	Good	Fair	Poor
	>5	≤5 to 1.5	<1.5
Narrative	No signs of eutrophication in the sediment. Sediments, and bottom waters in subtidal areas are well oxygenated. Healthy macrofauna community.	Moderate signs of eutrophication in the sediment. Oxygenated sediment surface, with moderate rates of sulfate reduction in deeper layers. Bottom waters in subtidal areas are oxygenated. Moderate stress on macrofauna.	Severe, likely persistent, eutrophic conditions. Sediments are devoid of oxygen with high rates of sulfate reduction. Bottom waters in subtidal areas are anoxic and sulfidic. Conditions likely uninhabitable for macrofauna.

Overall confidence in thresholds/ bands: Fair.

Recommendation: TOC:TS

While studies generally agree, we recommend undertaking further data collection and analysis in New Zealand estuaries to determine whether the proposed thresholds for TOC:TS are appropriate, and to determine whether this indicator should be restricted to depositional areas or applied estuary-wide. This recommendation remains consistent with the ETI recommendations (Robertson et al. 2016).

Links to other indicators: As discussed above, other commonly measured indicators serve as explanatory variables for changes in TOC:TS including RPD, iron, TN, grainsize, sedimentation rate, and abundance of primary producers.

Alternative metrics considered: Degree of pyritisation and acid-volatile sulphides.

Additional work recommended:

i. Sediment TS is not routinely collected in fine-scale monitoring therefore there are limited data available for a national analysis. Additional data collection across different substrate types should be considered before comprehensive data analysis is undertaken. The data analysis should address whether the proposed thresholds for TOC:TS are appropriate and whether it is applicable to all substrate types or restricted to depositional areas.
ii. Strategies addressing how site-specific sampling can be scaled to estuary-wide characterisations should be developed (e.g., stratified-random designs). These approaches, with more data collection, can then be used to assess spatial thresholds for sediment TOC:TS.

14.2 DEGREE OF PYRITIZATION (DOP)

Indicator type: Indicator not endorsed.

Metric: DOP = %pyrite iron/(%pyrite iron + %reactive iron).

Unit of measurement: no units.

Spatial scale: Site specific.

Applicability: All New Zealand estuarine and coastal waters, including tidal lagoon (SIDE), tidal river (SSRTRE), intermittently closing and opening lakes and lagoons (ICOLLs), and deep bay (DSDE) estuaries.

Rationale: DOP serves as a proxy for sulphate reduction and is a qualitative indicator of eutrophication in estuarine sediments (Sutula 2011; ozcoasts.org.au). DOP is often used as a paleoenvironmental indicator of bottom water oxygenation (e.g., Cooper & Brush 1993); however, other studies have utilised it in surficial sediments for estuary monitoring (Kilminster 2010). However, laboratory methods are likely cost prohibitive to councils and with other sediment eutrophic indicators (e.g., RPD, TOC, TN, TOC:TS) available, DOP is not recommended for routine SOE monitoring.

Method: Sediment TOC and TS samples are typically taken in surficial sediments using either surface scrapes down to 20mm using a trowel (Robertson et al. 2002) or small cores down to 20mm deep (Sutula et al. 2014). Samples are refrigerated or frozen until laboratory analysis.

The recommended laboratory method for %pyrite is the determination of chromium reducible sulphides (i.e., pyrite plus a negligible fraction of elemental sulphur), with the extraction approach described previously (e.g., Canfield et al. 1986; Fossing & Jørgensen 1989; Liu et al. 2020) and extractant analysed spectrophotometrically. The recommended laboratory method for reactive iron is extraction with 1N hydrochloric acid at room temperature for 24 hours and the extractant is analysed spectrophotometrically (Leventhal & Taylor 1990).

Assessment baseline: The DOP is an indicator of redox status which can be variable over time therefore a 'natural' (pre-human) baseline is unlikely to be suitable in this instance. However, like the TOC:TS ratio, hindcast methods such as deep sediment coring (and dating) could be used to assess changes (i.e., frequency, duration) in redox status over time (e.g., Cooper & Brush 1993).

Measurement considerations: When assessing long-term trends, or making spatial comparison, it is important that the sampling approach (i.e., sample depth) and laboratory analysis method remain consistent. Artefacts from method changes (e.g., extraction method or conditions like time or acid concentration) can compromise the interpretation of long-term trends.

The objective of monitoring and type of estuary will likely determine the type of sampling approach. While most SOE monitoring undertaken by councils is focused on intertidal areas, sediment DOP can also be collected from subtidal sediments using remote sampling devices (e.g., Eckman grab, corers). Supporting field metadata requirements include date, time, tide height, GPS coordinates, substrate type and substrate condition (i.e., RPD). Other indicators that will likely aid in data interpretation include RPD, TOC, TN, grainsize, sedimentation rate and (where applicable) macrofauna, epifauna and surface growths of algae (macro- or micro-).

Calculation of statistic: DOP is typically expressed as a proportion from 0 to 1, with values closer to 1 indicating more pyrite formation and prolonged anoxic conditions.

Potential bands and/or thresholds and rationale (including caveats):

There are variable thresholds for DOP presented in the literature:

- Leventhal and Taylor (1990) and references therein suggested a DOP of sediment below well oxygenated waters is generally <0.4, while sediments below sub-oxic waters (i.e., low oxygen but not sulfidic) are 0.5 to 0.7 and sediments below euxinic (i.e., no oxygen and sulfidic) waters have a DOP >0.7.
- Raiswell et al. (1987) reported DOP <0.42 for sediments deposited under aerobic conditions and 0.46 to 0.8 for sediments deposited under restricted oxygen conditions in the water column. That same study found that

sediments deposited under no oxygen conditions overlapped with restricted oxygen conditions with a DOP range of 0.55 to 0.93.

- Kilminster (2010) found in the Peel-Harvey Estuary and Leschenault Inlet (Western Australia) DOP values were 0.37 and 0.27, respectively. With Peel-Harvey Estuary experiencing periods of water column anoxia, coincident with more iron being convert to pyrite (i.e., higher DOP).
- While DOP thresholds are used qualitatively elsewhere, the large range of DOP values, some of which are not exclusively related to a particular redox status, indicate that thresholds are currently unreliable. Further, the lack of New Zealand data prevents any comparison to literature values. Sutula (2011) also highlighted that because DOP represents the saturation of buffering capacity (i.e., pyrite production is limited by iron availability), it has limitations as an indicator (i.e., cannot provide information beyond a saturation point).

Summary of proposed thresholds: No proposed thresholds.

Overall confidence in thresholds/ bands: Low.

Recommendation: Degree of pyritization (DOP)

There are inconclusive thresholds proposed in international literature and to our knowledge no local data is available to make a further assessment. Further the complexity of the laboratory approach could potentially be cost prohibitive to councils. Given other sediment eutrophic indicators (e.g., RPD, TOC, TN, TOC:TS) are available DOP is not recommended for further development.

Links to other indicators: As discussed above, other commonly measured indicators serve as explanatory variables for changes in TOC:TS including RPD, iron, TN, TP, grainsize, sedimentation rate, and abundance of primary producers.

Alternative metrics considered: TOC:TS and acid-volatile sulphides.

Additional work recommended:

i. Not recommended for further development at this stage.

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APPENDIX 15. TRACE METALS

Authors: Barrie Forrest & Don Morrisey (Salt Ecology)

Trace metals (sometimes call 'heavy' metals) are a class of potential contaminants that have natural sources, but concentrations in bed sediments can be elevated in areas of anthropogenic development, in particular around point sources in urban environments. Excessive levels of trace metals in bed sediments have the potential to cause toxic effects on estuary biota, with some metals having the potential to bioaccumulate in aquatic organisms and biomagnify up successive trophic levels.

BACKGROUND

Sources of trace metals in estuaries include industrial processes, diffuse source inputs via stormwater, agricultural activities, and natural sources that derive from catchment geology or geothermal activity (Smith 1985; Morrisey et al. 2000; Williamson & Morrisey 2000). For example, zinc is a ubiquitous trace metal derived from many anthropogenic sources including galvanised roofs, vehicle tyre wear, and sacrificial anodes on boats. Copper, chromium and arsenic have widespread use in timber preservation (Smith 1985), with copper also ubiquitous in port and marina environments, as it is the most common biocide used in vessel hull antifouling coatings (Dafforn et al. 2011). Lead has widespread historical use as an additive in petrol and paint. Certain trace metals (arsenic, cadmium, copper, lead, mercury, zinc) may also arise from agricultural land use, as they may also be associated with horticultural compounds (e.g. pesticides, fungicides) and fertiliser application (Gaw et al. 2006).

Several New Zealand studies have revealed that significantly elevated concentrations of trace metals in estuary sediments can arise from natural catchment sources. For example, estuaries and shallow subtidal zones in the top of the South Island have very high concentrations of nickel (and other metals to a lesser extent), due to the geology of catchment rocks (Robinson et al. 1996; Forrest et al. 2022). Otago schist rock is enriched with arsenic, which can be released if the rock is exposed, for example due to mining (Blake et al. 2019). In parts of the North Island, arsenic, mercury and a range of other metals can be elevated in sediments as a result of catchment geothermal activity (Smith 1985; Rumsby 2009).

PROPOSED METRICS

The following indicator metric is proposed for monitoring trace metals.

Bed-sediment concentrations of the suite of trace metals recommended in the revised NEMP.

These are commonly occurring elements that are of interest as ecological toxicants, namely: arsenic (which is technically a metalloid), cadmium, chromium, copper, mercury, nickel, lead, and zinc. Depending on local circumstances, it may be appropriate to include other trace metals in a monitoring programme. In such instances, it is likely that the approach described below (i.e., 'rules') for defining thresholds will be similarly applicable.

15.1 TRACE METAL CONCENTRATION IN BED-SEDIMENT

Indicator type: Primary.

Metric: Total recoverable trace metal concentration.

Unit of measurement: mg/kg bed sediment (<2mm particle size) dry weight.

Spatial scale: Site-specific.

Applicability: National across all estuary types.

Rationale: Most councils measure trace metals as part of their SOE monitoring programmes, due mainly to concerns over metal toxicity to estuary biota. Trace metals from anthropogenic sources can also be regarded as a screening indicator of the potential presence of other contaminants (e.g., hydrocarbons, pesticides, plasticisers) that have similar general sources such as stormwater. Trace metals bind to mud (i.e., silt and clay) particles, and are associated with various other sediment constituents (e.g., organic matter, sulphide), hence concentrations can be elevated in deposition zones around point source inputs. Sediment sampling provides an integrated measure of trace metal contaminants that may have episodic inputs (e.g., during storms), which could be missed when undertaking spot water-quality sampling. Bed-sediment trace metal concentrations can be reliably measured by well-established laboratory methods. Where trace metals arise from anthropogenic sources, managing contaminant sources directly (e.g., industrial discharges), and targeting reductions in muddy sediment inputs, provide avenues for mitigating ecologically significant concentrations. However, due to trace metal accumulation in sediments, there is likely to be a lag between source reductions and benefits.

Method: The NEMP method is based on analysis of sediment samples collected from the surface 20mm of sediment (to reflect recent deposition), with further sampling considerations described in the revised NEMP document. For trace metal analysis, the original NEMP (Robertson et al. 2002) recommended nitric/perchloric acid digestion and flame atomic absorption spectrophotometry to measure concentrations of metals in the digest. Inductively coupled plasma mass spectrophotometry (ICP-MS) is now the standard procedure, capable of measuring a wider range of elements at lower concentrations and allowing samples to be processed more rapidly.

The recommended approach going forward is to adopt the USEPA 200.2 method (e.g., used by Hill Laboratories), which involves a strong nitric/hydrochloric acid digestion of the sediment fraction <2mm and analysis by ICP-MS. Screening to the sediment fraction <2mm is the recommended approach for comparison with sediment quality guidelines, so that the potential contaminant risk is not diluted by a large mass of gravel and other debris (ANZG 2018).

Measurement considerations: The contaminant capacity of sediments tends to increase with decreasing particle grain size, hence the concentration of contaminants is typically greater in muddier sediment fractions (Förstner & Wittmann 1979; ANZG 2018). In the absence of a change in muddy sediment inputs (or a change in metal sources), trace metals can be sampled infrequently (e.g., annually or at longer intervals).

Supporting data requirements include date, time, site name and GPS coordinates, sampling depth and collection method, relevant field observations (sediment texture, colour, presence of obvious organic enrichment), method of storage before analysis (this should be by freezing), date of laboratory analysis, analytical limits of detection (LoD), and any relevant notes in laboratory analysis reports. Concurrent analysis of sediment mud content is necessary to assist interpretation, preferably along with sediment organic matter (%TOC).

Note that the above recommended analytical method uses a strong acid digest and is regarded as a 'total recoverable' procedure, as distinct from a total extraction. The latter requires the use of hydrofluoric acid to extract all metals (e.g., from silicaceous material), and is not used due to health and safety concerns. Nonetheless, the recommended total recoverable procedure represents a conservative screening level analysis as it will include metals that are not biologically available. As such, if significant metal concentrations are detected (i.e., significant in the context of the thresholds described below), ANZG (2018) describes further tiered analyses, and recommends a weight-of-evidence approach, to assess ecological risk from high metal concentrations. For example Simpson et al.

(2013) recommend analysis of the mud (<63µm) fraction of the sediment with weak-acid digestion (e.g., 0.5M HCl) as representing the most reactive and biologically available component of the total recoverable metal concentration.

Calculation of statistic: Raw laboratory data on total recoverable metals in the <2mm grain size fraction should be compared to the threshold values for each analyte described in Table A15-1 below. If concentrations are below the analytical Limit of Detection (LoD), by convention half the LoD can be substituted (e.g., to calculate mean values for a site).

Potential bands and/or thresholds and rationale (including caveats): In 2000, Australia and New Zealand published joint marine and freshwater quality guidelines that included interim sediment quality guidelines for trace metals (ANZECC 2000). The interim guidelines were developed using ecological effects and laboratory toxicity data from North America (Long et al. 1995). The guidelines were updated in a publication released in 2018 (ANZG 2018), which recommended sediment quality guideline values (SQGVs) based on two effects thresholds, as follows:

- Default Guideline Value (DGV): indicates the sediment concentrations below which there is a 'low risk' (but not zero risk) of unacceptable effects occurring. To protect aquatic ecosystems, DGVs are intended to be used with other lines of evidence.
- Guideline Value-High (GV-high): is an 'upper' guideline value that provides an indication of concentrations at which toxicity-related adverse effects may be observable. The GV-high is intended as indicator of potential high-level toxicity problems, not as a guideline value to ensure protection of ecosystems.

A key issue in developing thresholds for council SOE monitoring is that there are many uncertainties associated with the SQGVs (Simpson et al. 2013), there is a potential for adverse ecological effects to manifest at trace metal concentrations much less than the DGV, and additive effects from multiple stressors may arise (Long et al. 1995; MacDonald et al. 2000; Hewitt et al. 2009; Simpson et al. 2013; Tremblay et al. 2017). As such, Auckland Council has used field-based species sensitivity distributions (field data on macrofaunal distributions and contaminant concentrations) to develop sediment quality guidelines (Environmental Response Criteria) that are generally far lower (i.e., more conservative) than the ANZG guidelines (Anderson et al. 2006; Hewitt et al. 2009). However, Simpson et al. (2013) point out that unless there is a distinct contaminant concentration gradient, relating observed effects to specific contaminants is confounded by co-occurring contaminants, factors that affect contaminant bioavailability, and other physical and chemical factors including other stressors (Simpson et al. 2013). Simpson et al. (2013) suggest that such studies are more appropriate as part of an ecological 'lines of evidence' approach.

Nonetheless, given the above findings it is important that councils have thresholds that reflect potential effects, and provide an early warning of the development of declining conditions (e.g., trends towards increased trace metal concentrations). Hence, we recommend that ANZG (2018) SQGVs form the foundation for threshold development, with the thresholds scaled consistently for all metals relative to the DGV and GV-High. In this respect we propose the same five-band scale for all monitored trace metals, based on 'rules' for thresholds as follows:

Very good = concentrations <25% DGV

Good = concentrations 25% to <50% DGV

Fair = concentrations 50% DGV to <DGV

Poor = concentrations DGV to <GV-High

Very poor = concentrations ≥GV-high

In this scale, the three lower bands are <DGV, which is an approach suggested for two main reasons:

i. They account for the possibility of locally-observed field effects concentrations that are very low relative to DGVs, as identified in New Zealand and overseas studies (Hewitt et al. 2009; Simpson et al. 2013; Tremblay et al. 2017).

ii. Mean trace metal concentrations in New Zealand estuaries appear in many cases to be <25% of the DGV, except for estuaries in urban catchments or where there are significant natural sources of trace metals.

The latter assessment is based on a cursory review of Salt Ecology data obtained from NEMP sampling (n=766 samples) from 90 estuary sites (across 36 estuaries) nationally. Sites are predominantly located in mid-estuary areas away from contaminant point sources and are not considered to be adversely impacted by heavy metal inputs. Under the proposed threshold scheme, such estuaries would be classified as 'Very good' based on their trace metal concentrations (see next section), providing councils with a benchmark from which to track long-term degradation (e.g., in estuaries with catchments that become increasingly developed or industrialised over decadal time scales).

Summary of proposed thresholds: A summary of the proposed threshold rules and band values for trace metals is provided in Table A15-1. Monitoring to ascertain the rating should be based on sampling the surface 20mm of sediment (consistent with the NEMP) and analysis of the fraction <2mm (i.e., excluding gravel and larger material).

Table A15-1: Recommended thresholds for trace metal concentrations in bed sediments in New Zealand estuaries.

Metal mg/kg	Ecological Quality Status (Trace metal concentration in bed sediment)				
	Very Good	Good	Fair	Poor	Very Poor
	<25% DGV	25% to <50% DGV	50% DGV to <DGV	DGV to <GV-High	≥GV-high
As	<5	5 to <10	10 to <20	20 to <70	≥70
Cd	<0.38	0.38 to <0.75	0.75 to <1.5	1.5 to <10	≥10
Cr	<20	20 to <40	40 to <80	80 to <370	≥370
Cu	<16	16 to <32.5	32.5 to <65	65 to <270	≥270
Hg	0.038	0.038 to <0.075	0.075 to <0.15	0.15 to <1	≥1
Ni	<5.25	5.2 to <10.5	10.5 to <21	21 to <52	≥52
Pb	<12.5	12.5 to <25	25 to <50	50 to <220	≥220
Zn	<50	50 to <100	100 to <200	200 to <410	≥410
Narrative	Ecological communities are healthy and resilient.	Minor stress on sensitive organisms.	Moderate stress on some species and a risk of sensitive macroinvertebrate species being lost.	Significant, persistent stress on a wide range of aquatic organisms.	A likelihood of local extinctions of keystone species and loss of ecological integrity.

Overall confidence in thresholds/ bands: High – the ANZG (2018) guidelines are based on limited field-based data/studies, with recognition that there is a potential for adverse ecological effects to manifest at trace metal concentrations much less than the DGV, and additive effects from multiple stressors may arise. While the proposed thresholds are expected to be conservative, more in-depth analysis of New Zealand field data would assist in the understanding whether they are appropriate nationally.

Recommendation: Trace metal concentration in bed sediment

Adopt Table A15-1 as preliminary numeric thresholds pending data analysis/review.

Links to other indicators: Sediment mud content, organic matter (% total organic carbon) and sulphide levels, as well as catchment sediment mass loads; for example, loads predicted from models (Hicks et al. 2019).

Alternative metrics considered: A more comprehensive assessment would ideally include other ubiquitous contaminants for which ANZG (2018) guidelines exist (i.e., from which thresholds can be set), in particular polycyclic aromatic hydrocarbons and some of the main organochlorine pesticides.

Additional work recommended: Further analysis of nationally available data on trace metals and associated sediment quality parameters (e.g. sediment mud content) would help to:

i. Elucidate the thresholds at which adverse ecological effects occur, and whether there exist regional differences.

ii. Better understand the current status of trace metals in New Zealand estuaries, and the extent to which differences within and among regions can be related to factors such as catchment land use.

iii. Provide insight into 'reference' conditions for trace metals in New Zealand estuaries.

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APPENDIX 16. CYANOBACTERIA

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Cyanobacteria can form blooms in freshwater, transitional and marine waters. In addition to the ecological impacts of cyanobacterial blooms, the species responsible for blooms can produce toxins which pose a health risk for both humans and animals. Incidences of cyanobacterial blooms have increased in response to anthropogenic drivers and will likely worsen with climate change.

BACKGROUND

Cyanobacteria are a type of photosynthetic bacteria, commonly called blue-green algae. There are two forms: planktonic cyanobacteria which live in the water column, and benthic cyanobacteria which live on substrate (i.e., both soft sediments and hard substrates). They can be a concern because some species produce toxins (cyanotoxins) that pose a health risk to humans and animals through ingestion (e.g., contaminated water and seafood), inhalation or dermal contact (WHO 2021). Toxin-producing species differ between freshwater (e.g., *Microcystis* sp.) and coastal environments (e.g., *Nodularia* sp.), leading to variable levels of risk associated with blooms in these environments. As such, the "New Zealand guidelines for cyanobacteria in recreational fresh waters" may not be directly applicable to all estuary types. Other negative effects of cyanobacterial blooms include low dissolved oxygen, poor water clarity, benthic smothering, fish kills and altered biogeochemical cycling (Sutula 2011; Paerl & Paul 2012).

Like other forms of algae (e.g., macroalgae and other phytoplankton), many cyanobacteria rapidly respond to fluctuations in nutrient (nitrogen and phosphorus) concentrations. Excess nutrients often lead to planktonic cyanobacterial blooms (Paerl & Paul 2012 and references therein). While international studies have linked benthic cyanobacterial proliferation to excess nutrients (e.g., Paerl & Paul 2012), in New Zealand, they often occur under low nutrient conditions in freshwater environments (MfE & MoH 2009). While most primary production in coastal waters is nitrogen limited (i.e., nitrogen is the nutrient limiting growth), some cyanobacteria species can fix atmospheric nitrogen, alleviating the need for an external nitrogen supply (Marino et al. 2006; Cook & Holland 2010). As a result, both nitrogen (TN) and phosphorus (TP) are important drivers of growth, including the ratio between the two (i.e., a low TN:TP ratio may promote the proliferation of nitrogen fixing cyanobacteria; Marino et al. 2006; Cook & Holland 2010; Funkey et al. 2014). The source (e.g., freshwater input or internal sources) of nutrients and their relative importance can also vary depending on the estuary type, with both physical (e.g., stratification, mixing) and biological (e.g., oxygen status) characteristics influencing nutrient availability.

Several factors can influence cyanobacterial bloom formation in addition to nutrient inputs and availability, including time since last flushing flow (i.e., residence time), salinity, stratification, oxygen status, light availability, temperature, and grazing pressure (Chan et al. 2006; Marino et al. 2006; Havens 2008; Cook & Holland 2010; Sutula 2011). Consequently, the drivers of cyanobacterial blooms are often complex, making the application of general stressor-response relationships more challenging.

Adding to this complexity are the large fluctuations in salinity within an estuary and the capacity for both marine and freshwater species to co-exist. Several studies have detected typically freshwater species in estuaries, and many freshwater species can tolerate brackish waters (e.g., *Microcystis* sp.; Preece et al. 2017). Further, studies have shown that intact colonies of freshwater species have been detected in estuary sediments which can act as an inoculum for future cyanobacterial blooms (Bormans et al. 2020).

The scarcity of data on both planktonic and benthic cyanobacteria in New Zealand estuaries limits the ability to use it as an indicator at present. The influence of cyanobacteria on ecological health has been well documented internationally (as discussed above) but the complexity of drivers may require an estuary-specific understanding to implement effective management. For example, in a brackish lake in Australia (Gippsland Lakes, Victoria), *Nodularia* sp. (a nitrogen-fixing cyanobacterium) blooms were associated with physical factors, such as mixing and stratification, that led to the release of legacy phosphorus from the sediment (Cook & Holland 2010). This example highlights that reducing catchment nutrients alone may not be enough to reduce blooms, as legacy effects and the physical characteristics of an estuary may override catchment interventions. Despite these complexities cyanobacteria as a human health indicator in estuaries should be considered and is briefly discussed below.

PROPOSED METRICS

We propose developing metrics for two different types of cyanobacteria, both relevant to human health, not ecological health:

1. Planktonic cyanobacteria (biovolume; mm^3/L or cell counts; cell/mL)
2. Benthic cyanobacteria (% cover)

16.1 PLANKTONIC CYANOBACTERIA (HUMAN HEALTH)

Indicator type: Supporting.

Metric: Biovolume or cell count. Biovolume is used when there is limited knowledge of the actual toxin producing species and there are many species that differ in volume. Cell count should be used when there is robust knowledge on toxin producing species.

Unit of measurement: biovolume (mm^3/L) or cell count (cells/mL)

Spatial scale: Site specific+

+Estuary wide estimates could be obtained from multiple site-specific samples or remote-sensing (e.g., Cannizzaro et al. 2019 and references therein).

Applicability: All New Zealand estuarine and coastal waters, including tidal lagoon (SIDE), tidal river (SSRTRE), intermittently closing and opening lakes and lagoons (ICOLLs), and deep bay (DSDE) estuaries.

Rationale: The risk posed to human health requires effective monitoring approaches and thresholds to trigger a management response (e.g., media release, warning signage) in a timely manner (e.g., immediate). Cyanobacterial biomass, measured as biovolume or cell count, is amenable to low cost (~$150 per sample) approaches to obtaining the data relative to the benefit, and is already measured as part of routine freshwater (i.e., lakes) quality monitoring, making it an accessible indicator for most councils.

Method: The "New Zealand guidelines for cyanobacteria in recreational fresh waters" (2009) describes a method for collecting water samples for planktonic cyanobacteria in freshwater and this approach is currently being used in New Zealand ICOLLs (e.g., Waituna Lagoon, Te Waihora/Lake Ellesmere, Wairewa/Lake Forsyth). At the time of writing, the 2009 New Zealand guidelines are being updated (MfE & MoH 2024 in press) and ICOLLs are being considered in further detail. While the method might be able to be adapted to different estuarine types it would require a thorough review and further consideration of tidal state, mixing status, stratification, and depth to be reliable.

Assessment baseline: An assessment baseline is not applicable to human health indicators.

Measurement considerations: Cyanobacteria method considerations include the type of monitoring required (e.g., surface vs depth profile), frequency (e.g., weekly, monthly), time of year (e.g., summer, spring) and where to monitor (e.g., where there are a high number of recreational users). For example, water-column profiling may be required in deep bays while surface grabs may be adequate for shallow systems. Blooms can also be spatially and temporally variable and therefore more flexible approaches to site selection may be required. Where applicable, tide state should be considered.

Supporting metadata requirements include date, time, site name and GPS coordinates, visual characteristics of the site, sampling depth and collection method. Additional *in situ* water quality measures such as temperature, salinity, dissolved oxygen, turbidity, water clarity, chlorophyll-*a* (proxy for phytoplankton), phycoerythrin (proxy for blue-green algae), and discrete water quality (e.g., nutrient concentrations) are possible supporting indicators that can be used to understand both the extent of the problem (e.g., phycoerythrin) in addition to potential drivers (e.g., nutrient concentrations, temperature, salinity stratification, water clarity).

Calculation of statistic:

ICOLLs: Weekly or fortnightly visual inspections and sampling of waterbodies where cyanobacteria are known to proliferate between spring and autumn (MfE & MoH 2024 in press). Because the indicator is related to human health a one-off sample can trigger a management response (MfE & MoH 2024 in press).

Other estuary types: Requires development.

Potential bands and/or thresholds and rationale (including caveats):

ICOLLs: Regional council water quality monitoring undertaken in ICOLLs already utilise the "New Zealand guidelines for cyanobacteria in recreational fresh waters" (2009) to trigger a management response. The 2009 thresholds were originally developed for freshwater species and therefore are particularly relevant after long periods of closure when salinities are low and residence times are extended. However, the main toxin producing species in ICOLLs (i.e., brackish waters), have been considered further in the updated "New Zealand Guidelines for Cyanobacteria in Recreational Freshwaters" (MfE & MoH 2024 in press), due to be released in 2024. The descriptions of threshold bands and the rationale are provided in that document and were not repeated here because it was still "in press" at the time of writing.

Other estuary types: In principle, planktonic cyanobacteria biomass or cell count are feasible human health indicators in estuaries, however, there are limited studies in estuaries and coastal waters in New Zealand, particularly for estuary types other than ICOLLs (i.e., in estuaries with higher salinities). In an indicator summary prepared by Biessy and Wood (2024) they stated that there were no thresholds relating to specific effects on human health in estuaries across New Zealand. Specifically, there is a lack of understanding of the cyanobacteria species present (including toxin-producing species) and how species might vary across different estuary types and salinities. While some species may overlap with those identified in ICOLLs, characterising which species are present and producing toxins is a precursor to assessing risk to human health in other estuary types.

Summary of proposed thresholds:

ICOLLs: Adopt the "New Zealand Guidelines for Cyanobacteria in Recreational Freshwaters" due to be released in 2024 (MfE & MoH 2024 in press). Thresholds are not presented here because the document was still "in press" at the time of writing.

Other estuary types: No thresholds have been proposed.

Overall confidence in thresholds/ bands:

ICOLLs: Very High

Other estuary types: Fair

In principle planktonic cyanobacteria biomass or cell concentrations are useful human health indicators, however a data deficit in New Zealand estuarine and coastal waters currently limits their development.

Recommendation: Planktonic cyanobacteria (human health)

Adopt the ICOLLs thresholds presented in the New Zealand Guidelines for Cyanobacteria in Recreational Freshwaters (in press). For all other estuary types, significant further work is required to identify cyanobacteria species present (including toxin-producing species) and how species might vary across different estuary types and salinities before thresholds can be established.

Links to other indicators: Blue-green algae biomass (measured as phycoerythrin) serves as a field measure of cyanobacteria, while phytoplankton biomass (measured as chlorophyll-*a*), encompasses a wider range of planktonic species that bloom. *In situ* field measurements, being quick and easily repeatable during a single site visit can offer valuable insights into the extent of the issue.

Alternative metrics considered: Toxin concentration measured in water and/ or biota (e.g., shellfish). Qualitative measures using *in situ* water quality sensors (e.g., phycoerythrin blue-green algae sensor).

Additional work recommended:

i. Review available international literature to assess the feasibility of developing planktonic cyanobacteria guidelines for estuaries. A project titled "Managing marine harmful algal blooms (HABs) in recreational settings", currently being undertaken by Cawthron Institute, alongside Health New Zealand (Te Whatu Ora) and the New Zealand Ministry of Health (Manatū Hauora), goes some way toward achieving this (Smith 2024 in prep).
ii. Data collection is likely required across a range of estuary types to assess the most common cyanobacteria species and cyanotoxins present before toxicological studies can be undertaken to develop thresholds. This data should be collected alongside *in situ* water quality indicators in addition to nutrient loads to improve local stressor-response relationships.

iii. Further research is required to assess the use of cyanobacteria as an ecological health indicator (e.g., effects on seagrass, macrofauna, fish, birds etc.).
iv. Explore estuary-wide measures of cyanobacteria (e.g., remote-sensing).

16.2 BENTHIC CYANOBACTERIA (HUMAN HEALTH)

Indicator type: Supporting.

Metric: Average percent cover of a designated site or Affected Area (AA).

Unit of measurement: % cover of benthic cyanobacteria across a site or AA.

Spatial scale: Site specific+

+Estuary wide estimates could be obtained from multiple samples (e.g., Ahern et al. 2007) or remote-sensing (e.g., Roelfsema et al. 2006 and references therein).

Applicability: All New Zealand estuarine systems where benthic algae can grow. Likely most relevant to tidal lagoon (SIDE), tidal river (SSRTRE) and intermittently closing and opening lakes and lagoons (ICOLLs).

Rationale: The risk posed to human health requires effective monitoring approaches and thresholds to trigger a management response (e.g., media release, warning signage) in a timely manner (e.g., immediate).

Method: Methods for monitoring benthic cyanobacteria in estuaries require further development in New Zealand and should consider both intertidal and subtidal monitoring. To our knowledge, only Auckland Council has previously undertaken benthic cyanobacteria monitoring in coastal environments following blooms of toxic benthic marine cyanobacteria (i.e., *Lyngbya majuscula*; Tricklebank & Hay 2007), however it has now stopped, and methods were not developed further for national use. There are international examples in Australia (e.g., DES 2018) and the US (e.g., Krimsky & Staugler 2023) where monitoring and surveillance of benthic cyanobacteria in coastal waters are undertaken. Other studies have also complemented field-based monitoring (e.g., transect monitoring) and remote-sensing in larger estuaries (e.g., Roelfsema et al. 2006).

Further, in macroalgal monitoring, biomass, in addition to percent cover, is a more representative measure of degradation caused by macroalgal blooms (e.g., opportunistic macroalgal blooming tool; WFD-UKTAG 2014). It may be necessary to consider biomass, in addition to percent cover, or toxin concentration to accurately assess human health risk.

Assessment baseline: An assessment baseline is not applicable to human health indicators.

Measurement considerations: Cyanobacteria method considerations include the type of monitoring required (e.g., intertidal vs subtidal), frequency (e.g., weekly, monthly, response), time of year (e.g., summer, spring) and where to monitor (e.g., where there are a high number of recreational users or whole estuary). Benthic proliferations can also be spatially and temporally variable and therefore monitoring at fixed sites may not be suitable and more flexible approaches (e.g., monitoring in response to public reports) may be required.

Supporting field metadata requirements include date, time, tide height, GPS coordinates for point based data (e.g., percent cover), substrate type and quality (i.e., TOC, TN, TS, aRPD), and where applicable water quality (e.g., clarity, turbidity, salinity, nutrients). Furthermore, complementary stressor indicators include climate variables (e.g., wind, temperature, etc.), nutrient and sediment loads, land use types and hydrodynamic characteristics such as flushing time, tidal exchange, and dilution.

Calculation of statistic: Requires development.

Potential bands and/or thresholds and rationale (including caveats): There are no thresholds that relate to specific effects on human health in coastal waters in New Zealand (Biessy and Wood 2024). In principle, benthic cyanobacteria cover is a feasible human health indicator in estuaries, however, there are limited to no studies on benthic cyanobacteria in estuaries and coastal waters in New Zealand. This data deficit means there is a lack of understanding of the cyanobacteria species present (including toxin-producing species) and variation across different estuary types and habitats (e.g., intertidal vs subtidal). Characterising species and understanding how toxin concentration relates to percent cover (and biomass) is a precursor to developing thresholds and understanding toxicological effects.

Summary of proposed thresholds: No thresholds are being proposed.

Overall confidence in thresholds/ bands: Low. In principle, benthic cyanobacteria is a useful human health indicator, however a data deficit in New Zealand estuarine and coastal waters currently limits its development.

Recommendation: Benthic cyanobacteria (human health)

Significant further work is required to establish thresholds for benthic cyanobacteria (human health) in estuaries.

Links to other indicators: Other indicators that serve as explanatory variables for changes in benthic cyanobacteria include substrate type and quality (e.g., TOC, TN, TS, aRPD), water quality indicators (e.g., clarity, turbidity, salinity, nutrients) and climate variables (e.g., wind, temperature, etc.). Furthermore, complementary stressor indicators include nutrient and sediment loads, land use types and hydrodynamic characteristics such as flushing time, tidal exchange, and dilution.

Alternative metrics considered: Toxin concentration measured in the mat or biota (e.g., shellfish). A multi-metric index for benthic cyanobacteria (like macroalgae) which might include biomass.

Additional work recommended:

i. Review available international literature to assess methods for measuring benthic cyanobacteria in estuaries. Further consideration of the effects of biomass on toxin concentration may also be necessary.
ii. Review available international literature to assess the feasibility of developing benthic cyanobacteria guidelines for estuaries. A project titled "Managing marine harmful algal blooms (HABs) in recreational settings", currently being undertaken by Cawthron Institute, alongside Health New Zealand (Te Whatu Ora) and the New Zealand Ministry of Health (Manatū Hauora), goes some way toward achieving this (Smith 2024 in prep).
iii. Data collection is likely required across a range of estuary types to assess the most common benthic cyanobacteria species and cyanotoxins present. This data should be collected alongside *in situ* field measures (e.g., water quality, substrate type, sediment quality) in addition to climate variables and nutrient loads to improve local stressor-response relationships.
iv. Further research is required to assess the use of benthic cyanobacteria as an ecological health indicator (e.g., effects on seagrass, macrofauna, fish, birds etc.).
v. Explore estuary-wide measures of benthic cyanobacteria (e.g., remote-sensing).

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APPENDIX 17. DISSOLVED OXYGEN (WATER COLUMN)

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Oxygen is essential for aquatic ecosystems because it enables organisms to extract energy from organic matter. When respiratory consumption of estuarine water column oxygen becomes greater than replenishment by photosynthesis or hydrodynamic and atmospheric exchange, oxygen concentrations are decreased and can become stressful for biota (hypoxia). In extreme cases, hypoxia can be catastrophic for biota and normal biogeochemical functioning of coastal ecosystems. Thus, there is need to develop robust thresholds for assessing dissolved oxygen (DO) levels in estuaries for the maintenance of estuarine health.

BACKGROUND

Oxygen is required for many chemical and biological processes in the ocean and even periodic declines in oxygen levels cause changes in coastal productivity, biodiversity, and biogeochemical cycles (Howarth et al. 2011). Coastal and estuarine waters with the greatest tendency to become hypoxic are those that receive high inorganic and/or organic nutrient loads (Caffrey 2004; Salisbury et al. 2008; Bianchi & Allison 2009), and those that density stratify (Buzzelli et al. 2002; Scully 2016). Relatively deep and long-residence time estuaries have a greater tendency to stratify (Lowery 1998) and thus deep waters within these systems have a greater tendency to become hypoxic, as found in Firth of Thames (NZ: Zeldis et al. 2022). Hypoxia can, however, also occur in well-mixed estuaries that receive high nutrient and/or organic matter loads from land (Caffrey 2004; Verity et al. 2006; Salisbury et al. 2008; Zeldis et al. 2022). Hypoxia may also be driven by aquaculture (fish farming) that adds organic matter to aquaculture cages, in NZ and elsewhere (Plew 2019; Burke et al. 2021), which impacts both the environment and the fish welfare itself (Oldham et al. 2018).

Increased seawater temperatures with climate change are likely to exacerbate hypoxia impact by reducing the solubility of oxygen in seawater, increasing organismal and ecosystem metabolism, and increasing the tendency of the ocean to stratify (Vaquer-Sunyer & Duarte 2011; Statham 2012). Cumulative effects of ocean acidification and hypoxia can co-occur in coastal waters because respiration of organic matter increases carbon dioxide (CO_2) within seawater, increasing its hydrogen ion concentration and reducing its pH (Sunda and Cai (2012). This acidification has reached levels that are deleterious to marine life in NZ and elsewhere (Wallace et al. 2014; Law et al. 2019; Zeldis et al. 2022) and can act in concert with hypoxia (Gobler et al. 2014; Tomasetti et al. 2021) as a cumulative stressor.

Understanding of sensitivity of marine organisms and ecosystems to hypoxia is reasonably well developed (Boynton & Kemp 2008; Vaquer-Sunyer & Duarte 2008; Sutula et al. 2012). This has led to development of useful thresholds of hypoxia for application in coastal resource management (Sutula 2011).

PROPOSED METRIC

The following indicator metric is proposed:

Dissolved oxygen (DO)

Site specific, estimates for whole estuary obtained from statistics on multiple samples.

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Indicator type: Primary.

Metric: Dissolved oxygen (DO)

Unit of measurement: mg/L DO or % DO of air saturation (most commonly), µmol/kg DO (less commonly).

Spatial scale: Estuary-wide or within spatial strata (if a stratified survey design is used), made with multiple site-specific samples. It will often be necessary to carry out sub-surface sampling to account for stratified physical and oxygen conditions.

Applicability: All New Zealand estuarine and coastal waters including tidal lagoon (SIDE), tidal river (SSRTRE) (including intermittently closed cases for both), coastal lakes, and deep bay (DSDE) estuaries.

Rationale: While coastal managers have found it difficult to identify thresholds with respect to some ecosystem health indicators (e.g., for nutrients and phytoplankton), thresholds with respect to DO are more certain. For example, within the California Nutrient Numeric Endpoint (NNE) assessment, Sutula (2011) considered the DO indicator to have strong linkages to beneficial uses (i.e., with respect to responses of biota to DO stress), well-vetted means of measurement, well-modelled relationships between the indicator and nutrient management, and acceptable measurement precision for eutrophication assessment. There remain, however, uncertainties on precision of settings of DO thresholds and time and space scales over which they are assessed (Sutula et al. 2012) (see below).

Method: There are standard methods for measuring DO (NEMS 2020) in coastal waters. Most monitoring of coastal waters carried out by regional council scientists uses in-situ probe measurement of DO concentration (Dudley et al. 2017; Dudley & Todd-Jones 2018). These samples are almost always taken during the day, in the top 30cm of the water column, most commonly at monthly frequency. This sampling may miss problems associated with de-oxygenation of bottom water in sub-tidal parts of estuaries and, even in well-mixed estuaries may miss oxygen minima that often occur at night (Zeldis et al. 2022). Measurement of DO across depth profiles (e.g., using boat-deployed instruments) is carried out more rarely (Dudley et al. 2017). Continuous monitoring, e.g., using sensors deployed on moorings or attached to submerged structures, is carried out by some councils (e.g., WRC, BOP) and the feasibility of this approach is being investigated by others (Gadd et al. 2020). However, to date, most continuous DO monitoring using moored sensors has been conducted by research organisations, including a 20-year time series in Firth of Thames (Zeldis et al. 2022).

Measurement considerations: Sampling can be carried out with targeted or systematic sampling across different scales of data resolution. Intertidally, wading or sampling from structures (wharves, bridges, etc.) may be feasible (NEMS 2020). Values are likely to be affected by tide state, so uniform sampling with respect to tide may be advised (NEMS 2020). For subtidal sampling, a vessel may be required. Regional council state of the environment (SoE) sampling for coastal water quality is typically conducted from shore or helicopter (Dudley et al. 2017), precluding high frequency and depth profile sampling. As noted, diurnal to annual DO changes tracked over time using moored, continuously recording DO sensors or over depth profiles using probes (e.g., Zeldis et al. (2022)) constitute the most comprehensive methods for monitoring coastal DO, but expense hinders their use.

Calculation of statistic: Dissolved oxygen data are typically acquired using discrete, probe-based measurements, but they can be summarised or averaged over various temporal scales from minutes to days, weeks, or seasons, or with respect to the temporal scale of the threshold against which DO is being assessed. For example, in this document, thresholds discussed use 'acute' and 'chronic' time scales, which have single reading and weekly averaged time scales, respectively. This is most robust if data are collected by continuously recording sensor deployments.

Potential bands and/or thresholds and rationale (including caveats): Sutula et al. (2012) described minimum DO criteria for California estuaries for 'chronic' (long-term) effects, ranging between 5.8 and 6.3mg/L DO, with the higher thresholds applying for systems sustaining fast swimming fish (salmonids). For 'acute' (short-term) effects the minimum limits ranged from 2.3 to 4.0mg/L DO, with the lower values applying to intermittently closed estuaries. Sutula et al. (2012) did not provide formal advice on averaging periods over which these standards apply but recommended that this be a subject of further analysis (see additional work recommended, below). Sheldon and

Alber (2010) designated minimal DO criteria of 3mg/L DO for a 'fair/poor' threshold and 5.5mg/L DO for a 'good/fair' threshold based on the assessments for USA estuaries of Bricker et al. (2003) (although no temporal durations were indicated). Batiuk et al. (2009) designated DO criteria for Chesapeake Bay (USA), including a 30-day mean of 5mg/L DO applied to open-water habitats, with a 7-day mean of 4mg/L DO and an instantaneous minimum of 3.2mg/L DO.

For NZ, the New Zealand National Objective Framework (NOF) (NZ Ministry for Environment, unpublished) designated a 'fair-poor' threshold at 5mg/L DO. Green and Cornelisen (2016) described an 80% DO saturation (5.9mg/L DO)* criterion for Waikato Regional Council (to avoid unsatisfactory conditions), and for Auckland Council (average of all samples except bottom samples which may reach 65% saturation (4.8mg/L DO).

*Calculated as % air saturation at typical NZ autumn surface conditions (21°C, 34 psu salinity, 1013.25 atm barometric pressure) using loligosystems.com/resources/online-oxygen-converter/

A recent review of water quality targets for estuaries in the Horizons region 'One Plan' (Dudley et al. 2024), recommended that the minimum DO standard for its estuary management subzone should be 70% DO saturation, and 90% for its seawater (open coast) management zone (5.2 and 6.6mg/L DO, respectively). Again, averaging time frames did not accompany these criteria. Limits designated within the ETI (Robertson et al. 2016), were based largely on the 'chronic' California estuary criteria (Sutula et al. 2012) described above. The Estuary Trophic Index (ETI) (Robertson et al. 2016). designated values dependent on the duration of exposure, including a 7-day minimum threshold of ≥7mg/L DO indicating no stress/minor stress on aquatic organisms and a threshold of <5.0mg/L DO indicating significant, persistent stress with likelihood of local extinctions and loss of ecological integrity.

Organism responses to hypoxia have been summarised in meta-analyses by Gray et al. (2002) and Vaquer-Sunyer and Duarte (2008). Gray et al. (2002) outlined a taxonomic progression of decreasing sensitivities to hypoxic stress across a range of effects (growth, metabolism, mortality) progressing from fish → crustaceans→ annelids → bivalves. The findings of Vaquer-Sunyer and Duarte (2008) were largely in agreement with those of Gray et al. (2002) but were conducted within a formal statistical framework that used 872 experimental assessments across 206 marine benthic organisms. They found that fish and crustaceans had the highest lethal concentration thresholds (i.e., they were most susceptible to hypoxic stress), followed by bivalves. Sublethal thresholds associated with life-giving factors such as reduced growth and reproduction, increased physiologic stress, forced migration, reduction of suitable habitat, increased vulnerability to predation, and disruption of life cycles were found to be highest for fish and crustacea, followed by molluscs. More resistant taxa were generally also those with greatest potential mobility, although this did not necessarily extend to fish. Lethal times (after exposure to acute hypoxia) were shortest for crustacea and fish (order of few hours to a few days), while times for molluscs were order of a few hundred hours. Early ontogenetic stages were often considerably more susceptible to hypoxia than later stages. In a subsequent meta-analysis Vaquer-Sunyer and Duarte (2011) showed that survival times under hypoxia were reduced by on average 74% and that median lethal concentration increased by 16% when marine benthic organisms were exposed to warmer temperatures.

Vaquer-Sunyer and Duarte (2008) questioned the widespread use of the 2mg/L DO poor-fair threshold in conventional applications (e.g., U.S. Environmental Protection Agency (2000)), and recommended its upward revision. They showed that the 2mg/L DO threshold is below the empirical sublethal and lethal oxygen thresholds for half the species they tested. They recommended a level of 4.6mg/L DO as "a precautionary limit to avoid catastrophic mortality events, except for the most sensitive (e.g., crab) species, and to effectively preserve biodiversity".

In NZ, adult Greenshell™ mussels (*Perna canaliculus*) are resilient to low oxygen in intertidal habitats and can sustain themselves through long emersed periods (Marsden & Weatherhead 1998), although this incurs a metabolic cost. *Perna canaliculus* larvae showed large reductions of survival and settlement at 6mg/L DO, in experimental tanks (Alfaro 2005), although spat were not affected.

As noted above, the most sensitive group to low O_2 (in terms of sublethal effects) is fish, particularly active swimmers. Cultured juvenile kingfish show strongly impaired growth rates (by 39%) at 24°C in low O_2 treatments ranging between 2.9 and 4.9mg/L DO (Bowyer et al. 2014), and Pirozzi et al. (2019) showed significantly reduced nutrient utilisation in juvenile kingfish at 5.4mg/L DO. Tolerances for farmed Chinook salmon (*Oncorhynchus tshawytscha*), like kingfish, are active swimmers. They have high DO requirements, with recommended minimum DO concentration of 6mg/L DO (Sim-Smith & Forsythe 2013) and concentrations below that are defined as hypoxic because they cause a decrease in blood O_2, chronic stress and reduced growth.

In terms of ecosystem biogeochemical responses to low O_2 concentrations, the rate at which reactive nitrogen is naturally removed by denitrification within estuaries is sensitive to O_2 conditions (Eyre & Ferguson 2009). Boynton and Kemp (2008) demonstrated a consistent decline in denitrification as near-bottom O_2 concentrations decreased, including levels of 3–5mg/L DO.

Summary of proposed thresholds: The proposed DO thresholds (Table A18-1) were assigned four bands, to be consistent with the four-band scoring system used the ETI (Robertson et al. 2016; Zeldis & Plew 2022) and the National Policy Statement for Freshwater Management (New Zealand Government 2020). The thresholds range from no stress/minor stress on aquatic organisms at ≥7mg/L DO, to significant, persistent stress with likelihood of local extinctions and loss of ecological integrity at <5.0mg/L DO (Table A17-1). The discussion above documents that DO levels below ~5mg/L DO indicate extreme conditions to be avoided for welfare of marine taxa and ecosystems, so this level is recommended here as a C-D threshold. That threshold excludes cultured finfish, for which higher levels are necessary (*ca*. minimum 6mg/L O_2). The 5mg/L DO limit is above the precautionary limit recommended by Vaquer-Sunyer and Duarte (2008) (4.6mg/L DO) and is therefore considered appropriate. The range of the C band includes the minimum 'chronic' threshold nominated by Sutula et al. (2012) (5.8mg/L DO). The A-B threshold (7mg/L DO) achieves reasonable DO levels with respect to minimum healthy limits suggested by Sutula et al. (2012) and Vaquer-Sunyer and Duarte (2008). The thresholds are designated (Table A17-1) as 7-day mean minimum values meaning they are derived from organismal responses over that length of exposure, and therefore could be considered 'chronic' exposures (*sensu* Sutula et al. (2012)). However, estuary DO sampling by resource managers will usually be periodic (e.g., once per month) and made by discrete grab sampling (i.e., not temporally averaged), so there is a need to be able to interpret such discrete 'instantaneous' DO results in terms of longer term ('chronic') effects. This could entail use of the 7-day minimum thresholds as precautionary, conservative limits which, if breached in discrete sampling, raises the need for more intense investigation for those cases (see: Additional work recommended, below).

Table A17-1: Recommended 7-day mean minimum DO thresholds (mg/L DO) for New Zealand estuaries (adapted from Robertson et al. 2016).

7-day mean minimum (mg/L DO)	Ecological Quality Status			
	Very Good (A)	Good (B)	Fair (C)	Poor (D)
	≥7.0	7.0 to ≥6.0	6.0 to ≥5.0	<5.0
Narrative	No stress caused by low O_2 on any aquatic organisms that are present at near-pristine sites.	Occasional minor stress on sensitive organisms caused by short periods of lower O_2. Risk of reduced abundance, performance and welfare of sensitive fish and macroinvertebrate species.	Moderate stress on aquatic organisms caused by O_2 less than preference levels. Risk of sensitive fish and macroinvertebrate species being lost.	Significant, persistent stress on aquatic organisms caused by O_2 less than tolerance levels. Likelihood of local extinctions of keystone species and loss of ecological integrity.

Overall confidence in thresholds/ bands: High

Recommendation: Dissolved oxygen (Water column)

Adopt as preliminary numeric thresholds pending analysis/review of data from NZ and elsewhere, particularly for NZ native species oxygen tolerances (*cf*. Sutula et al. 2012).

Links to other indicators: As discussed above, estuary DO varies as functions of nutrient loads, water column stratification presence/absence, dynamics of phytoplankton biomass and productivity, and flushing times. Several of these links have been included within the Bayesian network model of Zeldis and Plew (2022).

Alternative metrics considered: Biochemical Oxygen Demand (BOD) measures rate of oxygen consumption, conducted on water samples. It is a chemical procedure for determining the rate of consumption of DO by aerobic biological organisms in a body of water to break down organic material present. It is usually expressed in mg O_2 consumed L^{-1} of sample over 5-day incubations at 20°C. BOD has been used commonly in wastewater monitoring and is often used as a surrogate of the degree of organic matter loading in water. Another measure, Chemical Oxygen Demand (COD) is less specific, measuring everything that can be chemically oxidized, rather than just levels of biologically active organic matter. BOD and COD are measured as rates, whereas DO concentration measures the sum of physical (e.g., stratification.), chemical (e.g., oxidation of reduced compounds) and biological processes (photosynthesis and respiration). Thus, high rates of BOD and COD may not necessarily imply hypoxia, because of the processes determining DO concentration. Therefore, direct measures of DO concentrations have a clear linkage to beneficial uses, while BOD and COD are linked, but only indirectly.

Additional work recommended:

i. A protocol should be developed that specifies where, when, and how samples should be collected, e.g., standardised protocols and/or guidance for measurement of DO, including the spatial (across estuary, surface vs. bottom v. integrated) and temporal density of data collection.

ii. Develop an assessment framework that clearly articulates how data would be applied to decide whether the estuary is impaired. Consideration should be given to formalise guidance for monitoring programmes and interpretation of DO data. Part of this guidance should include methodologies to interpret temporal/spatial representation data in the context of 'acute' and 'chronic' threshold limits. Related to this, appropriate averaging periods for 'acute' and 'chronic' criteria should help in establishing defensible thresholds (*c.f.*, Sutula et al. 2012)). From this perspective, New Zealand datasets collected at high frequency (minutes) for long periods (months to years) could be interrogated statistically to determine to the optimum balance of averaging periods for DO criteria, and associated sampling designs for frequency and duration of DO monitoring. An example of such a dataset is that of Zeldis et al. (2022) for Firth of Thames.

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APPENDIX 18. NUTRIENTS (WATER COLUMN N AND P)

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Nitrogen (N) is the key nutrient of concern with regards to estuarine and coastal eutrophication, acting as the dominant limiting nutrient for growth of phytoplankton and 'nuisance' species of macroalgae (Valiela et al. 1997; Howarth & Marino 2006; Barr 2007). Models suggest that phosphorus (P) may limit phytoplankton blooms in a smaller fraction of New Zealand estuaries (Plew et al. 2018a; Plew et al. 2020b). As the process of eutrophication progresses, the excessive production of aquatic plants and algal biomass results in an over-accumulation of labile organic matter in surface waters and sediments, altering the balance of basic biogeochemical cycles in the sediments and surface waters and leading to a cascade of adverse effects (Vitousek et al. 1997; Sutula et al. 2011). Expression of these symptoms in coastal waters is moderated by physical characteristics of the water body. Water bodies with high dilution rates of freshwater, rapid flushing and high wave action tend to express fewer and less intense eutrophication symptoms relative to nutrient loads. So, while eutrophication is a nationwide issue, it is of most concern in estuaries where nutrient loads are high, and moderating physical characteristics are less pronounced (Plew et al. 2020b).

BACKGROUND

In New Zealand and globally, increased nutrient inputs to land and their subsequent passage via freshwater flows (from both diffuse and point sources) have reduced the ecological integrity of coastal waters (Vitousek et al. 1997; Fowler et al. 2013; Plew et al. 2020b; Hale et al. 2024). Nutrient loads to land in New Zealand have increased greatly in the past two centuries, and much of this additional load passes via freshwater flows to increase nutrient concentrations in coastal waters (Parfitt et al. 2012; Plew et al. 2018a; Snelder et al. 2018; Dudley et al. 2020). At a regional scale, spatial extent and magnitude of coastal degradation follows patterns of increased nutrient availability in New Zealand estuaries (Plew et al. 2018a; Dudley et al. 2020; Plew et al. 2020b; Zeldis et al. 2021; Hale et al. 2024) and estuaries globally (Smith 2003; Rabalais et al. 2009; Howarth et al. 2011; Paerl et al. 2014).

Nutrients are present in the waters of estuaries and other coastal waters in a variety of chemical forms; these are summarised in Table A18-1. Several of these forms are directly available as nutrient sources to primary producers (such as plants and algae). Biogeochemical processes occurring within coastal waters can change the chemical form of nutrients, sometimes rapidly and with variation over short spatial scales (Sutula et al. 2011). Furthermore, nutrients entering estuaries from land can be rapidly taken up by primary producers (particularly during summer months), so that water column nutrient concentrations remain low, while eutrophication worsens (Zeldis et al. 2021). For these reasons, measured concentrations of the nutrient forms listed in Table A18-1 can relate poorly to trophic state, including algal growth rates and other symptoms of eutrophication. To contend with these issues, loads of nutrients to estuaries adjusted for dilution and flushing; i.e. 'potential nutrient concentrations' (Zeldis et al. 2017; Plew et al. 2020b; Zeldis & Plew 2022), or biological indices of nutrient availability (Sutula et al. 2011; Borja et al. 2012; Barr et al. 2013; Stevens et al. 2022) are commonly preferred metrics for quantifying relationships between nutrient pressure and trophic response in coastal waters.

When interpreting state or trends of measured nutrient concentrations or potential nutrient concentrations, we can assess the impact of dilution by examining salinity in estuaries. We can assess the effects of biogeochemical processes (such as rapid nutrient uptake by algae, or denitrification) by examining trends in potential nutrient concentrations in estuaries, as well as measured nutrient concentrations in estuaries.

PROPOSED METRICS

The following indicator metrics are proposed for monitoring nutrient availability in coastal waters.

1. Potential nutrient concentrations: DIN, SRP, TN, TP
2. Measured nutrient concentrations: DIN, SRP, TN, TP

Table A18-18. Nutrient species components of nutrient loads to coastal waters. Adapted from Sutula et al. (2011).

Form	Components of Total Nitrogen	Components of Total Phosphorus
Dissolved Inorganic	Nitrate (NO_3^-) + nitrite (NO_2^{2-}) Ammonium (NH_4^+; in dynamic equilibrium in natural waters with unionized or free ammonia).	Ortho-phosphate (PO_4^{-2}) is considered freely dissolved. Measurements of phosphate are "soluble reactive phosphorus (SRP)," which includes ortho-phosphate plus P that is loosely adsorbed to particles.
Dissolved Organic	Dissolved organic nitrogen. Typically, nitrogen attached to organic macromolecules (often a large portion of total nitrogen in natural waters especially those less impacted by human activities, and especially during periods of active decomposition of organic matter (e.g. algal bloom die-off)).	Dissolved organic phosphorus (can be a large portion of total phosphorus in natural waters less impacted by human activities, and especially during periods of active decomposition of organic matter (e.g. algal bloom die-off).
Particulate	Particulate organic nitrogen (detritus left from undecayed or partially decayed organic matter). Particulate inorganic nitrogen (insignificant in natural waters and usually not considered).	Particulate organic phosphorus (detritus left from undecayed or partially decayed organic matter). Particulate inorganic phosphorus (typically associated with minerals).

18.1 POTENTIAL NUTRIENT CONCENTRATIONS (NITROGEN AND PHOSPHORUS)

Indicator type: Primary.

Metric: State: Median and other quantiles

Trend: Percent change in load per year

Unit of measurement: Milligrams (of N or P) per litre (mg/L).

Spatial scale: Site specific.

Applicability: All New Zealand estuarine waters. Not applicable in open coastal waters.

Rationale: Spatial patterns of modelled potential nutrient concentrations in New Zealand estuaries relate to spatial patterns of eutrophication impacts (Plew et al. 2020b). Potential nutrient concentrations therefore provide a metric that is sensitive enough to detect eutrophication impacts of loads to estuaries in New Zealand.

Nutrient load data to estuaries across New Zealand is available through 'steady state' models (e.g. NZRiver Maps (Whitehead & Booker 2019), and CLUES (Elliott et al. 2016)). However, while the concentration and flow measurements required to assess temporal changes in nutrient availability can be assessed by repeat sampling at terminal river reaches, there are currently few freshwater monitoring stations in such locations nationally. Potential nutrient concentrations nevertheless provide a robust metric of nutrient pressure especially suited to estuaries, relatable to primary production rates, as well as other eutrophication impacts (Sutula et al. 2011; Zeldis et al. 2022).

Method: Methods to model potential nutrient concentrations in estuaries (loads of nutrients to estuaries adjusted for dilution and flushing in the absence of any biogeochemical processes) are provided in documentation to manage estuaries in the United States of America (NRC 2000), and New Zealand (Plew et al. 2018b; Plew et al. 2020b). Load calculations can be acquired from model products, as described above, but load measurements require both concentration and flow measurements at the same river location, ideally situated on or near the terminal river reach (which passes to the coast). Dudley et al. (2022) provide information on selecting which terminal river reaches to monitor to aid integration of estuary management within freshwater management units. Information on measurement of nutrients in rivers is available at the NEMS website at https://www.nems.org.nz/documents/water-quality-part-2-rivers while flow measurement is covered at https://www.nems.org.nz/documents/open-channel-flow-measurement. There are many available methods to calculate nutrient loads from time series of concentration and flow; these are reviewed in the context of water quality management by Snelder et al. (2017). Snelder et al. recommended the L7 method (Cohn et al. 1989) to provide realistic loads with high precision and representativeness.

Measurement Considerations: We would recommend consideration of increasing monitoring at terminal river reaches, and open coastal sites offshore from estuaries. These data are necessary to determine drivers of trophic change in estuaries through time.

Calculation of statistic: There are various methods that can be used to calculate this statistic, ranging from simple dilution models as described in Plew et al. (2018b) to more advanced hydrodynamic modelling approaches (Dudley et al. 2024). All of these methods benefit from estuary specific measurements relating to loads, but also physiographic estuary measurements relating to estuary dilution. Measurements relating to estuary dilution include salinity, estuary volume, tidal prism and freshwater discharge. While estimates of these parameters are available for many New Zealand estuaries (Zeldis et al. 2017), reliability of potential nutrient concentration estimates can be increased by measuring these parameters accurately.

Potential bands and/or thresholds and rationale (including caveats): The priority list provided in the 2018 update to the Australian and New Zealand Environment and Conservation Council (ANZECC) guidelines (ANZG 2018), specifies that guidelines be developed using:

- reference-site data
- laboratory-effects or field-effects data
- multiple lines of evidence based on two or more of these data sources.

The guidelines suggest that priority should be given to laboratory-effects or field-effects data, then local and ecosystem-specific reference site data, then (if insufficient ecosystem-specific monitoring data are available) to default guideline values. For guideline values derived from field and laboratory-effects data, the ecological or biological effects of the stressors are used to define guideline values below which ecologically meaningful changes do not occur. Reference guideline values define a measurable level of change from a natural reference condition that, although the ecological consequences are unknown, is considered unlikely to result in adverse effects. In the absence of reference conditions, the ANZECC (2000) guidelines provide default values for nutrient concentrations in South Australian coastal waters.

In New Zealand, field-effects and laboratory-effects based guideline values for potential nutrient concentrations have been developed for the New Zealand Estuary Trophic Index (ETI) Tools (Plew et al. 2020b; Zeldis & Plew 2022). As described above, potential nutrient concentrations ignore potentially important transformation and uptake processes that determine nutrient concentrations measured in estuary waters (Sutula et al. 2011; Robertson & Savage 2018; Gadd et al. 2020). However, because potential nutrient concentrations can be modelled for every estuary in New Zealand, they permit comparison with eutrophication impacts in estuaries without comprehensive water column nutrient concentration records. This has facilitated development of guideline potential nutrient concentration values to reduce risk of eutrophication impacts. While potential nutrient concentrations do not measure the same parameter as nutrient concentrations obtained by within-estuary grab sampling, they measure nutrients potentially available to primary producers (Plew et al. 2018) with guidelines of potential nutrient concentrations set using measured, co-occurring trophic response (e.g., macroalgal ecological quality rating (EQR; Figure A8-1) and models of phytoplankton response (Plew et al. 2020, Zeldis and Plew 2022).

Further research may be advisable to strengthen potential nutrient concentration bandings in New Zealand's northern regions (Waikato, Auckland, and Northland) that contain mangroves. The ecological data behind ETI eutrophication susceptibility bands are mostly derived from areas of New Zealand outside these regions. Mangroves provide shade and other ecological effects likely to alter relationships between nutrient concentrations in seawater and eutrophication impacts (including to macroalgae and seagrass cover) in New Zealand's northern estuaries.

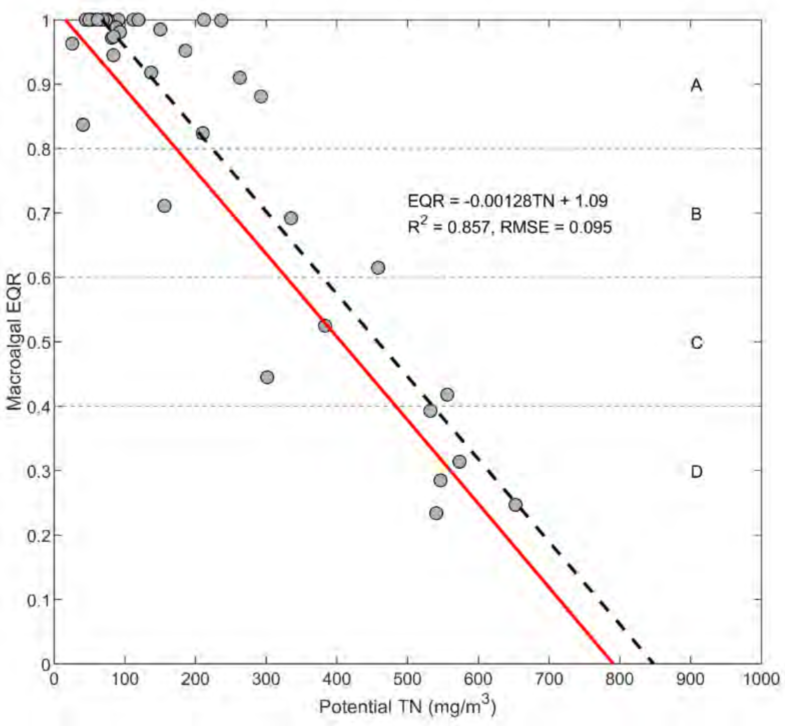

Figure A18-1. Macroalgal EQR vs potential TN relationship. The dashed black line shows a least-squares best fit linear regression through the data, while the solid red line is set at the 25% under-protection level (where 25% of observed values have worse EQR than would be predicted using this relationship).

Comparison between predicted (potential) nutrient concentrations from modelled loads and concentrations measured in estuaries has shown good correlation when hydrodynamic models are used to represent mixing processes (Dudley et al. 2024). Simpler mixing models such as that of Plew et al. (2018b), which represent a whole-of-estuary average nutrient concentration, are likely to produce coarser results. More research is required in this area to improve guidelines for nutrient concentrations measured in New Zealand coastal waters (see recommendations below).

Determining nutrient pressure enables better quantification of anthropogenic impacts on estuary trophic state. The ETI thresholds are considered suitable preliminary thresholds to adopt in New Zealand for potential nutrient concentrations as they are based on international literature and extensive data derived from New Zealand estuaries.

Summary of proposed thresholds: While there is considerable utility in current measurement of nutrient concentrations in New Zealand estuaries, we do not currently have nationally applicable guideline values for measured nutrient concentrations. In the absence of guidelines for measured concentrations, we recommend that potential nutrient concentrations are used to guide management of eutrophication in estuaries. The tables below were originally proposed as a screening tool to identify New Zealand estuaries prone to eutrophication (Plew et al. 2020b; Zeldis & Plew 2022). Estimates of whole-of-estuary potential nutrient concentrations are available for all New Zealand estuaries at https://shiny.niwa.co.nz/Estuaries-Screening-Tool-1/, however, we caution that these use uncalibrated dilution models. Confidence in predictions can be improved by calibrating the models using field studies and/or hydrodynamic modelling (Plew et al. 2020a; Dudley et al. 2024; Hale et al. 2024).

Table A18-3. Recommended Potential TN thresholds corresponding to macroalgal OMBT-EQR bands.

Potential TN concentration (mg/m^3)	Ecological Quality Status			
	Very Good (A)	Good (B)	Fair (C)	Poor (D)
	<175	175 to ≤335	335 to ≤495	>495
Narrative	No to 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 caused by the indicator

Table A18-4. Potential TN and TP thresholds corresponding to phytoplankton bands used in Plew et al. (2020) for estuaries in three salinity classes. Thresholds were determined where flushing time is such that chlorophyll is set solely by nutrient concentration.

Potential TN and TP (mg/m^3) vs phytoplankton bands		Ecological Quality Status			
		Very Good (A)	Good (B)	Fair (C)	Poor (D)
Oligohaline (<5ppt)	TN	<90	≥90 to 225	≥225 to 530	>530
	TP	<12	≥12 to 30	≥30 to 75	>75
Meso/polyhaline (≥5-30ppt)	TN	<45	≥45 to 90	≥90 to 145	>145
	TP	<6	≥6 to 12	≥12 to 20	>20
Euhaline (>30ppt)	TN	<30	≥30 to 75	≥75 to 110	>110
	TP	<4	≥4 to 10	≥10 to 15	>15
Narrative		No to 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 caused by the indicator

Overall confidence in thresholds/ bands: High

Recommendation: Potential nutrient concentrations

Adopt as preliminary numeric thresholds for estuaries pending analysis/review of New Zealand data.

Links to other indicators: Other eutrophication indicators in this report linked to nutrient availability include seagrass health and extent (Li et al. 2019), macroinvertebrate community composition (Clark et al. 2021; Lam-Gordillo et al. 2024), water clarity (Pedersen et al. 2014; Oviatt et al. 2017), phytoplankton / chlorophyll-*a* in water (Painting et al. 2007; Plew et al. 2020b), and dissolved oxygen content of water (Zeldis et al. 2022). Other indicators impacted by nutrient availability in estuaries include sediment organic matter and redox potential depth (Sutula et al. 2014; Zeldis & Plew 2022).

Alternative metrics considered: None.

Additional work recommended: As described below for measured nutrient concentrations.

18.2 MEASURED WATER COLUMN NUTRIENTS (NITROGEN AND PHOSPHORUS)

Metric: State: Median and other quantiles

Trend: Percent change in concentration per year

Unit of measurement: Milligrams (of N or P) per litre (mg/L).

Spatial scale: Site specific.

Applicability: All New Zealand estuarine and coastal waters, particularly open coastal waters.

Rationale: Spatial patterns of nutrient concentrations measured in New Zealand estuaries relate to spatial patterns of eutrophication impacts (Dudley et al. 2020; Plew et al. 2020b). Measured water column nutrient concentrations therefore provide a metric that is potentially sensitive enough to detect broad spatial and temporal changes in nutrient load to estuaries, and eutrophication impacts of those loads. We think this is most likely to be true for sites where nutrient availability is high, and biogeochemical processes (such as uptake by primary producers) within estuaries have proportionally less impact on measured concentrations.

Nutrient concentration data is routinely collected because it is relatively inexpensive, and reproducible (e.g., Dudley et al. 2017; Dudley & Todd-Jones 2018). Temporal changes in nutrient availability can be assessed by repeat sampling. Nutrient concentrations provide a metric of nutrient pressure to estuaries, relatable to nutrient loading, changes in freshwater mixing, or primary production rates, as well as other eutrophication impacts (Borja et al. 2004; Zeldis et al. 2022).

However, we do not have nationally applicable, field-effects based guideline values for measured nutrient concentrations. Therefore, we think that thresholds of measured nutrient concentrations are most appropriate for open coastal waters where potential nutrient concentrations cannot be calculated.

Method: Nationally applicable 'best practice' methods for sampling, measuring and archiving water quality data are available via the National Environmental Monitoring Standards (NEMS) webpage at https://www.nems.org.nz/. The standard for coastal water quality includes methods for monitoring nutrient concentrations (NEMS 2020). That document also includes information on sampling coastal water quality for salinity, which can be applied to aid interpretation of nutrient concentration state and trend data for estuaries. A full description of methods used by regional council scientists to measure water quality in New Zealand coastal waters is provided in Dudley et al. (2017).

Measurement considerations: NEMS documentation provides detailed descriptions of methods that can reduce noise and bias in time series of coastal water column nutrient data (NEMS 2020). For example, uptake of nutrients by primary producers (such as plants and algae) tends to be higher in the summer and lower in the winter. Influence of these seasonal patterns can be reduced in statistical trend analyses, but this is most efficiently done with an even spread of sampling across seasons (e.g., quarterly or monthly sampling frequency). Concentrations also tend to vary tidally; for this reason, for estuary nutrient concentration 'states' to be comparable between estuaries, timing of sampling should be stratified with respect to tide or conducted randomly with respect to tide. Other considerations for providing time series useful for trend analysis are maintaining consistency in methods and site location through the long (decadal) time periods required for trend detection (Dudley et al. 2017).

We would also recommend consideration of accompanying data necessary to interpret drivers of change across time series. Among indicators useful for managing eutrophication in coastal waters, water column nutrient concentrations provide information on nutrient 'pressure', to which other eutrophication indicators respond. However, particularly at low concentrations, measured nutrient concentrations in estuaries can be poor indicators of nutrient pressure for the reasons described in the sections above. To improve the robustness of nutrient concentrations as an indicator of nutrient pressure, we would highly recommend prioritising new nutrient concentration measurement sites in unimpacted open coastal areas, and nutrient concentrations and flow in major terminal river reaches entering estuaries (to calculate loads).

Calculation of statistic:

State: Percentiles (e.g., 5th, 20th, 25th, 50th, 75th, 80th, and 95th) calculated using the Hazen method (see (http://www.mfe.govt.nz/publications/water/microbiological-quality-jun03/hazen-calculator.html) from the distribution of measured values at each site. Data included in state analysis is typically limited to the 5-year period prior to the date of assessment (Dudley et al. 2017; Dudley & Todd-Jones 2018; Fraser et al. 2021).

Trend: Likelihood of positive or negative change in concentration (mg/L) over a specified time period. Because of the tendency of water column nutrient concentrations to vary seasonally, trends are normally assessed as the rate of change of the central tendency of the observations of through time. Because water quality is constantly varying through time, the evaluated rate of change depends on the period over which the trend is assessed. Therefore, trend assessments are specific for a given period (e.g., 10 or 15 years). The most recent trend analysis of New Zealand coastal water quality (Fraser et al. 2021), used statistical methods based on recent guidance for environmental trend assessment (Snelder et al. 2021). The analysis of Fraser et al. (2021) applied either the Mann Kendall assessment or the Seasonal Kendall assessment, using LWP Trends functions in the R statistical computing software.

Potential bands and/or thresholds and rationale (including caveats): The review of Green and Cornelisen (2016) collated nutrient concentration thresholds for coastal waters in both international and domestic literature (e.g., Bricker et al. 2003; Borja et al. 2004). Domestic guidelines include those designated by the National Objective Framework (NOF) process for estuaries, as well as several regional councils (as described below). The review of Green and Cornelisen (2016) highlighted some consistency between guidelines in different studies, but also different thresholds between estuary types, e.g., Borja et al. (2004).

Measurements from clean coastal water in New Zealand have consistently exceeded ANZECC (2000) default guidelines for nutrient concentrations (ANZECC 2000; Dudley et al. 2017). This has necessitated the development of regional reference guideline values based on coastal water nutrient measurements (e.g., Griffiths 2016; Foley 2018; Dudley et al. 2019). These efforts have highlighted regional differences between nutrient concentrations in the open coastal waters of New Zealand. There is (to our knowledge) little available field-effects data on the impacts of nutrient concentrations above reference conditions in the open coastal waters of New Zealand. There has been, however, useful research on impacts of changing nutrient loads. For example, there is considerable evidence to suggest that increased loads of nutrients can strongly impact biogeochemical processes and ecological function of our open (or partly sheltered) coastal seas; importantly, these impacts can occur with minimal change to nutrient concentrations measured in seawater (Zeldis & Swaney 2018; Zeldis et al. 2022; Macdonald et al. 2023).

Region-specific, statistically derived reference guideline values for concentration may still be useful for highlighting and restricting damaging inputs of nutrients at source. For example, an activity may warrant further scrutiny if annual median measured concentrations nearby exceed 80th percentile values of regional reference guideline values (ANZECC 2000).

As an example, Table A18-2 gives regional reference guideline values derived for open coastal waters around Canterbury. A caveat on this table is that there is generally poor understanding of what truly constitutes 'reference' conditions. For example, even offshore sites within the dataset of Environment Canterbury are affected by runoff from large Canterbury rivers (Hadfield & Zeldis 2012). This poor understanding of true reference conditions may also be true nationally; regional scientists report that monitoring at reference sites is hard to justify under rates-based financing, and that most data is collected to monitor likely land use impacts (Dudley et al. 2017). Therefore, for open coastal waters of New Zealand we recommend further development of regional baseline values using repeated open coastal sampling at sites with minimal anthropogenic influence.

Table A18-2: Water quality percentiles for individual management zones in the Canterbury Region. Shown also (far right column) are a range of trigger values used in comparable studies from around New Zealand (Griffiths 2016; Foley 2018; Madarasz-Smith 2018) and ANZECC trigger values for nutrients. Note that NHXN refers to ammonium and ammonia, while NOXN refers to the sum of nitrite and nitrate (see Table A18-1).

Nutrient and Unit (mg/L)		Akaroa Harbour	Lyttelton Harbour	Offshore	Open Coast North	Open Coast South	All groupings combined	Range of NZ open coastal trigger values
DRP	50th %ile	0.008	0.016	0.004	0.008	0.004	0.009	0.010 to 0.012
	80th %ile	0.015	0.023	0.011	0.0162	0.012	0.018	
	90th %ile	0.02	0.026	0.015	0.023	0.0151	0.023	
NHXN	50th %ile	0.01	0.013	0.01	0.016	0.01	0.012	0.015
	80th %ile	0.023	0.034	0.02	0.063	0.024	0.035	
	90th %ile	0.034	0.047	0.0369	0.13	0.04	0.064	
NOXN	50th %ile	0.01	0.011	0.0116	0.0187	0.049	0.015	0.005 to 0.027
	80th %ile	0.0352	0.045	0.07	0.063	0.113	0.063	
	90th %ile	0.066	0.069	0.092	0.09	0.15	0.092	
TN	50th %ile	0.17	0.19	0.18	0.23	0.23	0.2	0.11 to 0.120
	80th %ile	0.22	0.24	0.25	0.31	0.32	0.28	
	90th %ile	0.27	0.28	0.29	0.37	0.4	0.331	
TP	50th %ile	0.022	0.033	0.017	0.036	0.023	0.028	0.025
	80th %ile	0.033	0.044	0.027	0.064	0.0396	0.045	
	90th %ile	0.039	0.052	0.037	0.083	0.049	0.06	

Summary of proposed thresholds: While there is considerable utility in current measurement of nutrient concentrations in New Zealand coastal waters, we do not have nationally applicable guideline values for measured nutrient concentrations.

Overall confidence in thresholds/ bands: Fair – some studies/data but conclusions do not agree.

Recommendation: Measured water column nutrients

Generate region-specific baseline values for open coastal waters using repeated sampling at open coastal sites with minimal anthropogenic influence.

Links to other indicators: As for potential nutrient concentrations, eutrophication indicators in this report linked to measured nutrient concentrations include seagrass health and extent (Li et al. 2019), macroinvertebrate community composition (Clark et al. 2021; Lam-Gordillo et al. 2024), water clarity (Pedersen et al. 2014; Oviatt et al. 2017), phytoplankton / chlorophyll-a in water (Painting et al. 2007; Plew et al. 2020b), and dissolved oxygen content of water (Zeldis et al. 2022). Other indicators impacted by nutrient availability in estuaries include sediment organic matter and redox potential depth (Sutula et al. 2014; Zeldis & Plew 2022).

Alternative metrics considered: None.

Additional work recommended:

Analyse all regional council / research organisation nutrient concentration data sets to:

i. Compare measured nutrient concentrations with macroalgae EQR and chlorophyll-*a* thresholds (and their associated potential nutrient concentrations) for estuaries where such indices are available.

ii. Assess if threshold measured nutrient concentrations are evident, either annually, with respect to season, or with respect to estuary type.

iii. As part of this, assess if 'reference' estuaries exist, with low eutrophication indices; these could provide a basis for minimum nutrient concentration threshold(s) (cf. The TRIX trophic index of Maurizio et al. (2007)).

iv. Assess if consistently eutrophic estuaries exist; these could provide a basis for 'bottom line' nutrient concentration threshold(s).

For open coastal waters of New Zealand, we recommend further development of regional baseline values using repeated open coastal sampling at sites with minimal anthropogenic influence.

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APPENDIX 19. PHYTOPLANKTON

Authors: Keryn Roberts (Salt Ecology) and John Zeldis (NIWA, Christchurch)

Phytoplankton is ubiquitous in fresh, transitional, and marine waters. Its rapid growth in response to fluctuations in nutrient concentrations, tangible link to manageable anthropogenic inputs (e.g., nutrients), and easy measurement as chlorophyll-*a* (a proxy for phytoplankton biomass) make it one of the most common eutrophication indicators.

BACKGROUND

Phytoplankton are microscopic photosynthetic organisms that are primary producers. They form the foundation of the food web and utilise chlorophyll-*a* in photosynthesis to produce energy, oxygen and organic matter. Phytoplankton also play a fundamental role in determining water quality including the regulation of dissolved oxygen, nutrient and carbon cycling, turbidity and total productivity (Sutula 2011).

In situations where nutrients are in excess, phytoplankton can bloom, causing eutrophication that leads to negative effects on the environment (Fig. A19-1; e.g., low dissolved oxygen, poor water clarity, toxins (harmful algal blooms), fish kills, altered biogeochemical cycling). The links between phytoplankton blooms and increases in nutrient inputs and/or nutrient availability have been well documented globally (e.g., Howarth & Roxanne 2006; Smith 2006; Woodland et al. 2015) and in New Zealand (e.g., Zeldis & Swaney 2018; Safi et al. 2022; Zeldis et al. 2022). Because phytoplankton growth rapidly responds to nutrient inputs, it can effectively integrate available nutrients over a period of days, providing a more stable indicator of eutrophication compared to water-column nutrient concentration, which does not necessarily represent true nutrient availability due to consumption and production processes (Sutula 2011).

While phytoplankton growth is regulated by water-column nutrient concentrations (Howarth & Marino 2006; Smith 2006; Woodland et al. 2015; Safi et al. 2022), other factors including physical stratification, flushing time, dilution, salinity, light, temperature, water clarity and grazing pressure are also important (Cloern 1982; Alpine & Cloern 1988; Grzebyk & Berland 1996; May et al. 2003; Ferreira et al. 2005; Edwards et al. 2016; Gall et al. 2024). As a result, phytoplankton biomass can be both spatially (e.g., surface area and/or depth) and temporally variable, particularly across estuaries and among seasons. To effectively utilise phytoplankton as an indicator requires documenting this spatial and temporal variability (e.g., average over a year or season; Sutula 2011) or over longer scales (e.g., interannual (Gadd et al. 2020) to decadal (Safi et al. 2022)).

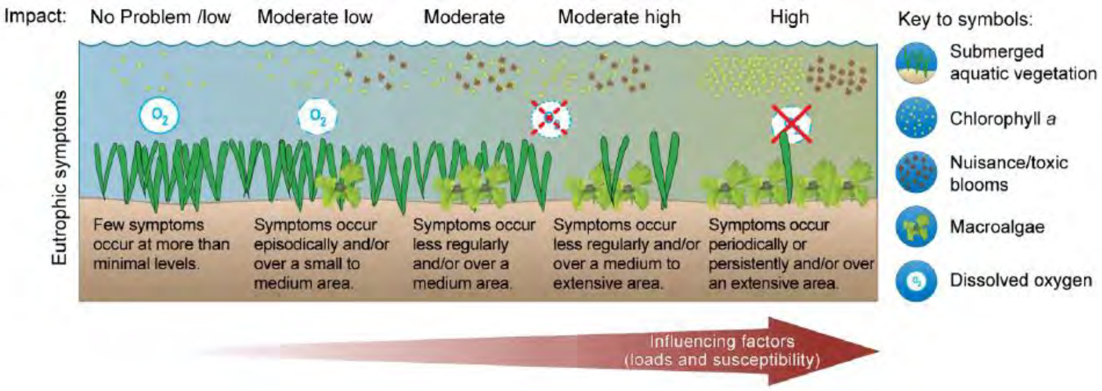

Fig. A19-1. Potential impacts of anthropogenic nutrient enrichment of coastal and marine waters (sourced from Devlin et al. 2014).

PROPOSED METRICS

The following indicator metric is proposed for monitoring phytoplankton.

- Phytoplankton biomass (measured as chlorophyll-*a*; mg/m^3)

19.1 PHYTOPLANKTON BIOMASS

Indicator type: Primary.

Metric: Concentration of chlorophyll-*a* in water

Unit of measurement: mg/m^3

Spatial scale: Site specific⁺

⁺Estuary wide estimates could be obtained from multiple site-specific samples, predictive modelling (e.g., Wild-Allen et al. 2013) or remote-sensing (e.g., Maciel et al. 2023 and references therein). However, phytoplankton biomass varies spatially and therefore there is a risk that estuary-wide measurements could mask or dilute the detection of potential problem areas (e.g., bloom).

Applicability: All New Zealand estuarine and coastal waters, including tidal lagoon (SIDE), tidal river (SSRTRE), intermittently closing and opening lakes and lagoons (ICOLLs), and deep bay (DSDE) estuaries.

Rationale: Several international estuary monitoring programmes have selected phytoplankton biomass (measured as chlorophyll-*a*) as a primary eutrophication indicator in coastal waters (e.g., Sutula 2011; Devlin et al. 2014; NSW 2015; DELWP 2021) because of its important ecosystem function (i.e., food-web support), acceptable signal to noise ratio and measurable response to nutrient loads and other management controls (e.g., flushing time). Further, site-specific water quality monitoring, that includes chlorophyll-*a*, is relatively low cost, and often measured as part of routine freshwater (i.e., lakes) and coastal water quality monitoring, making it an accessible indicator for most councils.

Method: The Coastal Water Quality National Environmental Monitoring Standards (NEMS) (2020) describe two approaches to monitoring chlorophyll-*a* in coastal waters:

(1) *In situ* (field) measurements that use a chlorophyll-*a* fluorescence sensor. The fluorescence sensor provides an indirect measure of algal pigments (chlorophyll-*a* and phycoerythrin*) in relative fluorescence units between 0-100% which is then converted to chlorophyll-*a* concentrations using a post-calibration procedure. The benefits of *in situ* sensors include instantaneous results and the potential for higher resolution sampling (e.g., more sites, depth profiles and/or continuous monitoring).

**Chlorophyll-a is present in all algae while phycoerythrin is a pigment commonly associated with blue green algae (cyanobacteria). Chlorophyll-a content is directly measured in discrete samples in the lab; see method 2 below)*

(2) Discrete water quality sampling, whereby a sample of water is collected in an opaque bottle and sent to a laboratory for analysis. The sample is filtered through a glass fibre filter and the chlorophyll-*a* extracted from the filter paper in a 90% acetone solvent (APHA 2012; Method APHA 10200H). The extraction mix is analysed for chlorophyll-*a* by fluorometry or spectrophotometry.

Assessment baseline: The most ecologically relevant baseline is chlorophyll-*a* concentration under reference conditions (i.e., chlorophyll-*a* levels in estuaries with limited anthropogenic disturbance). This could be established by monitoring unimpacted estuaries (e.g., Freshwater Estuary, Stewart Island), expert opinion or by using hindcast methods such as deep sediment coring (and dating) to estimate 'pre-human' phytoplankton concentrations.

Measurement considerations: Chlorophyll-*a* method considerations (e.g., calibration, sample storage, limit of detection) are discussed in the Coastal Water Quality NEMS (2020). Additionally, the type of monitoring required (e.g., instantaneous, continuous, depth profiles) and where to monitor (e.g., site selection) will be dependent on the objective of monitoring and estuary type. For example, longitudinal sites with depth profiles may be required for a stratified river-dominated estuary (SSRTRE), while fixed surface water sites may be more suitable for other estuary types (e.g., SIDEs) that do not stratify strongly. Some monitoring programmes utilise integrated depth profiles (i.e., sampling with a pvc water pipe down to 50cm from the surface; NSW-OEH 2016) or use composite samples across a large spatial area. While guidance exists for lakes (e.g., NEMS), this will need to be developed further for sampling phytoplankton in estuaries and will need to consider monitoring in both well mixed and stratified systems. Where applicable, tide state and height should also be considered and possibly standardised to reduce the level of noise in temporal datasets caused by varying degrees of dilution and mixing at different tide states.

When using *in situ* (field) measurements that use a chlorophyll-*a* fluorescence sensor, it should be acknowledged in waters with coloured dissolved organic matter, there can be significant interference causing chlorophyll-a concentrations to appear higher than they really are (NSW-OEH 2016). To accurately assess chlorophyll-*a* in these types of systems discrete water quality samples should be collected in preference to *in situ* (field) measurements, or be used to calibrate fluorescence readings (e.g., Zeldis et al. 2022; Gall et al. 2024).

Supporting metadata requirements include date, time, site name and GPS coordinates, sampling depth and collection method. Additional *in situ* water quality measures such as temperature, salinity, dissolved oxygen, turbidity and/or water clarity and discrete water quality (e.g., nutrient concentrations) are possible supporting indicators that can be used to understand both the extent of the problem (e.g., secondary impacts such as dissolved oxygen) in addition to potential drivers (e.g., nutrient concentrations, temperature, salinity stratification, water clarity).

Calculation of statistic:

<u>Intermittently closed/open lakes and lagoons (ICOLLs):</u> Annual median and maximum*

<u>All other estuary types:</u> Annual 90th percentile*

**These statistics assume regular (e.g., monthly) monitoring. The statistics for ICOLLs and all other estuary types are calculated before comparison to the thresholds proposed below.*

For ICOLLs, the median is to be calculated during periods when the ICOLL is open and during periods when the ICOLL is closed. Based on a rolling median of at least 12 samples for each situation (i.e., open or closed). Even though the likelihood of developing poor conditions is greater under closed conditions, the same thresholds apply to both scenarios.

Because the National Policy Statement for Freshwater Management (NPSFM 2020) includes established thresholds for ICOLLs the calculation statistic remains unchanged here. However, we have chosen to use 90th percentile for all other estuary types to reduce the risk of classifying an estuary based on a single exceptional peak that may not reflect long-term water quality.

Potential bands and/or thresholds and rationale (including caveats):

<u>Intermittently closed/open lakes and lagoons (ICOLLs):</u>

Bandings for phytoplankton biomass (measured as chlorophyll-*a*) in ICOLLs (Table A19-1) were developed for the NPSFM (2014; ammended 2020) based on thresholds proposed in the Trophic Lake Index (TLI; Burns et al. 2000) T Their applicability to ICOLLs was reviewed and recommended by Hamill et al. (2014). These thresholds have been used in ICOLL health assessments and limit setting processes across New Zealand (e.g., lawa.org.nz; Roberts 2020).

Because the thresholds are presented in national policy and have undergone previous review, no change is proposed here, except for the addition of a 'Very poor' band. This extension provides councils with the ability to differentiate between 'at high risk' of a regime shift to 'likely undergone/ or is undergoing' a regime shift to a more persistent poor state. Further the C-band ('Fair') represents a mid-point where an ICOLL is in a moderate state of health; it is also the current national bottom line (i.e., minimum acceptable state). The thresholds proposed for the 'Very poor' band were based on the same literature as the original threshold development (Carlson 1977; Chapra & Dobson 1981; Davies-Colley et al. 1993; Burns et al. 2000) and other supporting literature (e.g., Ferreira et al. 2011; OEH 2016 and references therein).

The ICOLL phytoplankton thresholds were set based on international literature and analysis of New Zealand data (see Burns et al. 2000; Hamill et al. 2014 and references therein) and correspond to phytoplankton biomass at different levels of nutrient (TN and TP) enrichment. The 'Fair' to 'Poor' threshold (i.e., national bottom line or the threshold at which there is high risk of reaching a tipping point) of 12mg/m^3 is set just below a known tipping point (~15mg/m^3) at which submerged macrophytes decrease and there is high risk of the ICOLL shifting from a macrophyte-dominated system to an algal-dominated system (see Wazniak et al. 2007; LTG 2013 and references therein).

All other estuary types:

Bandings for phytoplankton biomass (measured as chlorophyll-*a*) in estuary waters were developed for the Estuary Trophic Index (ETI) Toolbox (Robertson et al. 2016a, b; Zeldis et al. 2017; Zeldis & Plew 2022). The thresholds were largely based on the European Water Framework Directive (WFD) thresholds for Basque Estuaries (Spain) given their similarities to New Zealand systems (Borja et al. 2004; Revilla et al. 2010; Plew et al. 2020). These thresholds have been used in estuary health screening and limit setting processes across New Zealand (Plew & Dudley 2018; Plew et al. 2018; Roberts et al. 2021; Ward & Roberts 2021; Zeldis & Plew 2022).

It is expected that in estuaries with lower salinity there is more freshwater influence (indicating greater influence of catchment-derived nutrient loading) and less dilution by seawater and as such, even under natural conditions, we would expect to see higher concentrations of phytoplankton (Borja et al. 2004). To reflect this the thresholds (for estuary types other than coastal lakes and ICOLLs) are separated into two salinity categories euhaline (>30ppt) and meso/polyhaline (≥5-30ppt), with more stringent thresholds for euhaline systems (Plew et al. 2020). While we propose adopting the ETI thresholds here, like ICOLLs we propose a 5-band system with the addition of a 'Very poor' band (Table A1-2). The mid-point ('Fair') category represents where an estuary is in a moderate state of health. The 'Fair' to 'Poor' threshold represents that there is a high risk of reaching a tipping point where a permanent regime shift to a degraded state may be observed. The 'Poor' to 'Very poor' threshold provides councils with the ability to identify when an estuary at very high risk of a regime shift has likely moved to a more persistently degraded state. The thresholds proposed for the 'Very poor' band were based on the same literature as the original thresholds (Revilla et al. 2010; Ferreira et al. 2011) and are supported by other studies (Bricker et al. 1999; Borja et al. 2004).

The threshold bandings proposed here (Table A1-2) are consistent with NOAA Assessment of Estuarine Trophic Status (ASSETS; Bricker et al. 2003), for which a group of regional experts developed thresholds for chlorophyll-*a* (Sutula 2011): estuaries with annual chlorophyll-*a* less than 5mg/m^3 appear to be unimpacted, and at 20mg/m^{-3} and above effects include decline in seagrass, shift in phytoplankton community structure, high turbidity and low bottom-water oxygen. The WFD uses phytoplankton biomass, taxonomic composition, and abundance and frequency of phytoplankton blooms as the 'biological quality' elements in a framework to categorise waterbodies by ecological condition (Sutula 2011). The WFD uses chlorophyll-*a* thresholds that are similar to ASSETS, with <5mg/m^3 being undisturbed or slightly disturbed, and >30mg/m^3 being highly disturbed or hypereutrophic. This is further supported by guidelines proposed for Queensland and South Australian estuaries in which chlorophyll-*a* was set to 5mg/m^3 (ANZECC 2000) and a study from Queensland estuaries recommended a limit of <15mg/m^3 (90th percentile) to control nuisance problems (Moss 1987). It thus appears that a commonly used threshold for undisturbed systems is about 5mg/m^3.

Phytoplankton breakpoints for estuaries presented here (Table A19-2) are based on international literature because there are limited New Zealand data, particularly for different estuary types that allow ecological condition to be related to phytoplankton biomass. It is therefore recommended (see below) that collation of such data be a priority.

A caveat to note with respect to the thresholds is that deep bays (DSDE), especially those loaded heavily by inflowing catchment nutrients, appear sensitive to low levels of phytoplankton biomass. The main example of this in NZ is the euhaline Firth of Thames. Like other large, deep, long-residence time estuaries that seasonally stratify, (e.g., Chesapeake Bay, USA), the Firth is susceptible to hypoxia and acidification (Zeldis et al. 2022) at only moderate phytoplankton biomass levels (approximately within the 'Good' category of euhaline systems of Table A1-2: Zeldis et al. 2021). This suggests that such systems may require more stringent (i.e., lower) threshold settings than proposed in Table A19-2.

Summary of proposed thresholds:

See Tables A19-1 and A19-2.

Table A19-1: Phytoplankton biomass (as chlorophyll-*a* mg/m^3) thresholds for ICOLLs (Oligohaline <5ppt) taken from the NPSFM (2020). ICOLL ecosystem health graded based on the worst of the two metrics.

Phytoplankton biomass	Ecological Quality Status				
	Very Good	Good	Fair	Poor*	Very Poor
Oligohaline Annual median	≤2	>2 to ≤5	>5 to ≤12	>12 to ≤30	>30
Oligohaline Annual maximum	≤10	>10 to ≤25	>25 to ≤60	>60 to ≤150	>150
Narrative	Ecological communities are healthy and resilient.	Ecological communities are slightly impacted by additional phytoplankton growth arising from nutrient levels that are elevated.	Ecological communities are moderately impacted by phytoplankton biomass elevated well above natural conditions. Reduced water clarity likely to affect habitat available for plants (e.g., seagrass, macrophytes, macroalgae).	Excessive algal growth making ecological communities at high risk of undergoing a regime shift to a persistent, degraded state without macrophyte/seagrass cover, persistent blooms of algae and potential for oxygen depletion.	Likely regime shift, with persistent, degraded state without macrophyte/seagrass cover, persistent blooms of algae and potential for oxygen depletion.

*High risk of reaching a tipping point where wide-spread blooms establish (i.e., regime shift). Equal to the NPSFM (2020) national bottom line.

Table A19-2: Phytoplankton biomass (as chlorophyll-*a* mg/m^3) thresholds for all other estuary types from the ETI (Robertson 2016). Euhaline >30ppt, Meso/polyhaline ≥5-30ppt.

Phytoplankton biomass	Ecological Quality Status				
	Very Good	Good	Fair	Poor*	Very Poor
Euhaline 90th percentile	≤3	>3 to ≤8	>8 to ≤12	>12 to ≤16	>16
Meso/polyhaline 90th percentile	≤5	>5 to ≤12	>12 to ≤16	>16 to ≤32	>32
Narrative	Ecological communities are healthy and resilient.	Ecological communities are slightly impacted by additional phytoplankton growth arising from nutrient levels that are elevated.	Ecological communities are moderately impacted by phytoplankton biomass elevated well above natural conditions. Reduced water clarity likely to affect habitat available for plants (e.g., seagrass, macrophytes, macroalgae).	Excessive algal growth making ecological communities at high risk of undergoing a regime shift to a persistent, degraded state without macrophyte/seagrass cover, persistent blooms of algae and potential for oxygen depletion.	Likely regime shift, with persistent, degraded state without macrophyte/seagrass cover, persistent blooms of algae and potential for oxygen depletion.

*High risk of reaching a tipping point where wide-spread blooms establish (i.e., regime shift). Equal to the NPSFM (2020) national bottom line.

Overall confidence in thresholds/ bands: High.

There is general agreement in the international literature but limited local data to confirm the applicability of these thresholds in New Zealand estuaries.

Recommendation: Phytoplankton biomass

Adopt as preliminary numeric thresholds pending data analysis/review of New Zealand data.

Links to other indicators: As discussed briefly above, other *in situ* water quality measures such as temperature, salinity, dissolved oxygen, pH, turbidity, water clarity, photosynthetically active radiation (PAR) and discrete water quality (e.g., nutrient concentrations, TOC) are possible field-based supporting indicators that can be used to understand both the extent of the problem (e.g., secondary impacts such as lowered dissolved oxygen and lowered pH) in addition to potential drivers (e.g., nutrient concentrations, temperature, salinity stratification, water clarity). Furthermore, complementary stressor indicators include nutrient loads and potential nutrient concentrations, land use types and hydrodynamic characteristics such as flushing time, tidal exchange and dilution. Several of these links have been parameterised within the Bayesian network model of Zeldis & Plew (2022).

Alternative metrics considered: Cyanobacteria biovolume is considered in a separate indicator summary. In addition to phytoplankton biomass, taxonomic composition has been used internationally (see Sutula 2011; Devlin et al. 2014) and in NZ (Safi et al., 2022) to describe phytoplankton responses to eutrophication (including harmful algal bloom species). However, at present there is insufficient local data to propose this as an indicator in NZ.

Additional work recommended:

i. Collate standardised national data (and associated metadata) on phytoplankton biomass (chlorophyll-*a*) across different estuary types.
ii. Collate ancillary data on factors potentially affecting phytoplankton biomass levels, particularly long-term average observed nutrient concentrations, potential nutrient concentrations, and loads.
iii. Additional data collection is likely required across a range of estuary types (including euhaline systems, as noted above) to assess the suitability of the proposed thresholds to New Zealand estuaries. These data should be collected alongside supporting field indicators in addition to nutrient loads to improve local stressor-response relationships (i.e., relationship of phytoplankton levels to eutrophication indices, e.g., hypoxia, acidification, organic deposition.
iv. Develop guidance for sampling phytoplankton in discrete water quality monitoring taking into consideration both well mixed and stratified estuaries.
v. Explore whether current thresholds can be scaled to estuary-wide measures of chlorophyll-*a* through remote sensing and/or predictive modelling.
vi. Explore taxonomic composition of phytoplankton as an indicator of eutrophication and incidence of harmful algal blooms.

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APPENDIX 20. COLLATED RECOMMENDATIONS FOR FURTHER WORK

APPENDIX 1. MACROALGAE

1.1 Opportunistic macroalgal abundance (OMBT-EQR)

i. Collate standardised national data (and associated metadata) on macroalgae extent and OMBT-EQR scores. For example, Salt Ecology have OMBT-EQR data for ~50 estuaries (some over multiple years). It would be useful to combine this with other national datasets (e.g., Cawthron, NIWA and councils) in preparation for more comprehensive analyses.

ii. Additional data collection is required across a range of geographic regions (i.e., particularly estuaries with mangroves) alongside other supporting indicators and nutrient loads to improve local stressor-response relationships.

iii. Assess OMBT-EQR versus total nitrogen concentration (e.g., Plew et al. 2020) for estuaries containing mangroves to determine whether mangroves potentially buffer the effects of nutrients in estuaries and if simple predictive models can be used to predict estuary state (i.e., update Plew et al. 2018).

iv. Further research is required on the effects of macroalgal biomass on New Zealand macrofauna to validate current biomass thresholds. Further understanding is also needed on tipping points for macroalgae collapse at high levels of enrichment (i.e., where decomposition of high biomass blooms lead to severe eutrophic sediment conditions in which macroalgae are no longer able to survive) and responsiveness of the indicator to management interventions (i.e., whether legacy effects from persistent stable beds delay positive outcomes).

v. Explore whether remote sensing methods can be used to assist calculation of the OMBT-EQR, e.g., by improving percent cover estimates, remotely assessing biomass and entrainment, and reducing ground-truthing requirements.

vi. Analyse within and between provider accuracy in mapping of percent cover and measures/estimates of biomass and entrainment.

vii. Explore active management methods that include macroalgal removal, particularly in situations where issues are localised and not yet persistent and self-sustaining.

APPENDIX 2. MANGROVES

2.1 Mangrove forest extent

i. Standardise methodology for broad-scale mangrove habitat mapping using satellite imagery.

ii. Collate standardised national data (and associated metadata) on mangrove extent and explore relationships between changes in mangrove extent and ecological health to refine narrative thresholds, and to develop numeric thresholds.

iii. Provide guidance on how to consistently define reference/baseline conditions.

iv. Explore additional spatial metrics to assess condition via satellite remote sensing, for example global mangrove forest patch characteristics (Hai et al. 2022) that could be applied in Aotearoa.

v. Quantify correlation between sediment erosion rates and annual rates of mangrove expansion to inform the development of numeric thresholds.

vi. Standardise methodology for broad-scale mangrove habitat mapping using satellite imagery.

vii. Collate standardised national data (and associated metadata) on mangrove extent and explore relationships between changes in mangrove extent and ecological health to refine narrative thresholds, and to develop numeric thresholds.

2.2 Change in areal extent of mangrove forest covered by tall and dwarf mangroves (as indicator of mangrove quality)

i. Standardise methodology for broad-scale mangrove habitat mapping, including using satellite imagery to delineate stature.

ii. Collate standardised national data (and associated metadata) on mangrove extent and stature.

iii. Provide guidance on how to consistently define reference/baseline conditions.

iv. Explore additional spatial metrics to assess condition via satellite remote sensing, for example global mangrove forest patch characteristics (Hai et al. 2022) that could be applied in Aotearoa.

v. Explore applicability of comprehensive indices of mangrove forest quality, such as those developed globally, including ecological and environmental characteristics (e.g., macrofaunal communities, turbidity), as well as social attributes (Ibrahim et al. 2019), reflecting mangrove use and economic value.

APPENDIX 3. MUD-ELEVATED (25% MUD CONTENT) SEDIMENT

3.1 Percent of intertidal area with mud-elevated (25% mud content) sediment

i. Standardise sampling methods and reporting metrics, based on the current NEMP revision (Stevens et al. in prep).

ii. Collate standardised national data (and associated metadata) on measured mud-elevated extent.

iii. Test the classification accuracy of subjective assessments of substrate mud content using existing validation data.

iv. Analyse within and between provider mapping accuracy and assess confidence intervals on the assessment of temporal and spatial change.

v. Revise interim thresholds based on iii and iv. to refine percent loss breakpoints.

vi. Undertake further studies to determine the potential historical mud-elevated extent of New Zealand estuaries.

vii. Investigate development of supporting thresholds based on change measured in hectares.

3.2 Change in intertidal mud-elevated (>25% mud) sediment extent from the first accurate baseline

i. Collate standardised national data (and associated metadata) on measured mud-elevated extent.

ii. Analyse within and between provider mapping accuracy and assess confidence intervals on the assessment of temporal and spatial change.

iii. Revise interim thresholds based on i and ii. to refine percent loss breakpoints.

iv. Undertake further studies to determine the potential historical mud-elevated extent of New Zealand estuaries.

v. Investigate development of supporting thresholds based on change measured in hectares.

APPENDIX 4. SALT MARSH

4.1 Percentage of salt marsh in available salt marsh habitat (ASH)

i. Collate standardised national data (and associated metadata) on salt marsh extent.

ii. Assess measured salt marsh extent relative to Available Salt marsh Habitat (ASH).

iii. Revise interim thresholds based on ASH.

iv. Investigate development of a salt marsh multi-metric for New Zealand, similar to that developed by the WFD, which combines measures of current extent, loss from estimated historical extent, and loss from the first accurately measured baseline.

v. Assess the degree of potential salt marsh displacement by mangroves or *Spartina*.

vi. Assess the need to develop separate guidance for estuaries containing mangroves or *Spartina*.

4.2 Change in salt marsh extent from the first accurate baseline

ii. Collate standardised national data (and associated metadata) on measured salt marsh losses.

iii. Assess measured salt marsh losses attributable to natural processes or variation.

iv. Revise interim thresholds based on iii. to refine percent loss breakpoints.

vi. Investigate development of a salt marsh multi-metric for New Zealand, similar to that developed by the WFD, which combines measures of current extent, loss from estimated historical extent, and loss from the first accurately measured baseline.

v. Consider the merit of expressing percent change in salt marsh extent on an annualised basis (dividing by the number of years since the baseline was established) to enable standardised comparison among estuaries.

4.3 Change in salt marsh extent from estimated historical extent

i. Collate standardised national data (and associated metadata) on historical salt marsh extent.

ii. Revise interim thresholds based on ii. to refine percent loss breakpoints.

iii. Investigate development of a salt marsh multi-metric for New Zealand, similar to that developed by the WFD, which combines measures of current extent, loss from estimated historical extent, and loss from the first accurately measured baseline.

iv. Consider the merit of expressing percent change in salt marsh extent on an annualised basis (dividing by the number of years since the baseline was established) to enable standardised comparison among estuaries.

4.4 Salt marsh quality

i. Review sampling methods and reporting metrics to determine whether standardised national data can be compiled in future.

ii. Consider developing visual guides for classifying different states of visually observable degradation related to physical impacts, e.g. grazing, presence of introduced species, to facilitate consistency in reporting.

iii. Consider whether a rapid-screening metric for salt marsh quality could be developed from the above to derive potential narrative thresholds of salt marsh quality.

APPENDIX 5. SEAGRASS

5.1 Percent loss of dominant (>50% cover) intertidal seagrass from the first accurate baseline

i. Evaluate the consistency and accuracy of remote sensing methods for recording seagrass across a range of percent cover to determine minimum consistent data capture and reporting thresholds.

ii. Analyse within- and between-provider mapping accuracy.

iii. Collate standardised national data (and associated metadata) on seagrass extent.

iv. Analyse relationships between seagrass extent and other indicators (e.g., sediment accretion rates, nutrient concentrations, catchment land-use change) to explore links between potential drivers of change and seagrass extent. Refine thresholds as appropriate.

v. Develop standard methods to consistently define reference or baseline conditions.

vi. Assess natural temporal variation and variation attributable to anthropogenic stressors that can be managed.

5.2 Area weighted average percent cover (density) of intertidal seagrass with >10% cover

i. Collate standardised national data (and associated metadata) on seagrass cover.

ii. Analyse relationships between seagrass cover and other indicators (e.g., sediment accretion rates, nutrient concentrations, catchment land-use change) to explore links between potential drivers of change and seagrass cover. Refine thresholds as appropriate.

iii. Develop standard methods to consistently define reference conditions.

iv. Assess likely temporal variation attributable to anthropogenic and natural stressors.

v. Analyse within and between provider mapping accuracy.

5.3 Seagrass quality

i. Collate standardised national data (and associated metadata) on seagrass quality measures to enable potential patterns in seagrass quality response to be identified.
ii. Develop visual guides for classifying different states of visually observable degradation related to nuisance epiphyte or macroalgal cover and fine sediment smothering.
iii. Refine potential narrative thresholds based on field data to determine whether a general screening metric can be developed.

APPENDIX 6. SHELLFISH

6.1 Shellfish bed extent

i. Come to agreement and disseminate agreed operational definitions of 'high-density bed' for a set of readily identifiable estuarine shellfish species.
ii. Utilise existing published methods to rapidly map shellfish bed extent in estuaries.
iii. Define estuary-specific historical baselines for shellfish bed extent.
iv. Develop thresholds for percent change in shellfish bed extent from a recently measured baseline.
v. Consider the merit of expressing percent change in shellfish bed extent on an annualised basis (dividing by the number of years since the baseline was established) to enable standardised comparison among estuaries.

6.2 Shellfish Quality (Health)

i. No additional work is recommended at this stage.

APPENDIX 7. SEDIMENTATION

7.1 Sediment accretion rate (SAR)

i. Greater understanding of the link between SAR and ecological health.

APPENDIX 8. MACROFAUNA

8.1 AZTI's Marine Biotic Index (AMBI)

i. Develop NZ-specific EGs using QA/QC'ed macrofauna datasets, for which associated sediment quality data are available.
ii. Road test the EGs with a group of experts to achieve consensus, and make the agreed EGs available nationally (e.g., include EGs in the C-SIG Coastal Species Resource Tool; see: https://specieskey.atlasmd.com/).
iii. Further evaluate the limitations of AMBI in different estuary types and consider its utility as a tool for assessment of temporal change in stressor affects at discrete sites. Simultaneously, evaluate the RI-AMBI.
iv. Develop a guidance and tools (e.g., R code, desktop app) specific to the NZ-AMBI to enable easy calculation of AMBI and RI-AMBI by councils and science providers.

8.2 National Benthic Health Models (BHM)

i. Continue to trial the BHMs in a wider range of other estuary types across New Zealand, and evaluate the efficacy of the method for tracking temporal change in the effects of sediment mud and metals.
ii. Evaluate the scope to refine the relative ranking thresholds based on the BHM scores where the strongest shifts in macrofauna occur.
iii. Seek to develop 'absolute' thresholds that relate BHM scores to ecological condition, rather than scores relative to other estuaries.

iv. Support proposed work to develop a software package (likely within the software Primer), to enable easy and reliable BHM score calculation by councils and science providers. Simultaneously, it is recommended that training to use any such software is provided, to help ensure consistent application and interpretation.

8.3 Traits Based Index (TBI)

i. Develop guidance (e.g., methods manual, open-source R code, desktop app.) to enable easy and reliable TBI score calculation by councils and science providers.
ii. Calculate the TBI in other estuaries across New Zealand and compare results with those for Auckland and Waikato estuaries to evaluate national applicability.
iii. Determine the sensitivity of the TBI to changes in key environmental drivers; e.g., sediment mud content, nutrient load, and which are likely to be targeted for management.
iv. Evaluate whether proposed TBI thresholds can be further refined to provide greater discrimination of estuarine health.

APPENDIX 9. MICROALGAE

9.1 Sediment microalgae (Chlorophyll-a and phaeopigments)

i. Collection and analysis of existing national data (e.g., from regional authorities) to understand variability in chlorophyll-a and phaeophytin seasonally and nationally and across impacted to non-impacted estuaries.

APPENDIX 10. MUD CONTENT

10.1 Sediment mud content

i. More extensive analyses of collated existing national data, to specifically focus on threshold development, with consideration of factors that may influence ecological community sensitivity to mud, such as estuary typology.
ii. For future monitoring, seek agreement among councils and providers to ensure consistent and comparable analytical methods for sediment mud content. It is assumed that revisions to the NEMP will provide a means of fostering consistency in methods for sample collection.
iii. Consider the most appropriate way(s) to determine baseline state with respect to sediment mud content and investigate the development of related thresholds. As an interim measure, Zaiko et al. (2018) recommended 'bottom-line' guidance that 'sediment mud content at representative sites should not increase from its current extent'.

APPENDIX 11. NUTRIENTS (NITROGEN AND PHOSPHORUS)

11.1 Total Nitrogen (sediment)

i. Collate standardised national data (and associated metadata) on sediment TN including supporting indicators (grainsize, TOC, TP, TS, RPD, macrofauna). For example, Salt Ecology have sediment TN data for ~34 estuaries (some with multiple years). It would be useful to combine this with other national datasets (e.g., Cawthron, NIWA and councils) in preparation for more comprehensive analyses.
ii. Undertake a comprehensive analysis of a national dataset to improve confidence in the preliminary thresholds proposed for TN.
iii. Strategies addressing how site-specific sampling can be scaled to estuary-wide characterisations should be developed (e.g., stratified-random designs). These approaches, with more data collection, can then be used to assess spatial thresholds for sediment TN.

11.2 Total Phosphorus (sediment)

i. Collate standardised national data (and associated metadata) on sediment TP including other indicators (grainsize, TOC, TN, TS, aRPD, macrofauna). For example, Salt Ecology have sediment TP data for ~34 estuaries (some with multiple years). It would be useful to combine this with other national datasets (e.g., Cawthron, NIWA and councils) in preparation for more comprehensive analyses.

ii. Undertake a comprehensive analysis of a national dataset to determine whether TP thresholds are suitable for use in estuaries.
iii. Strategies addressing how site-specific sampling can be scaled to estuary-wide characterisations should be developed (e.g., stratified-random designs).

APPENDIX 12. ORGANIC MATTER

12.1 Total Organic Carbon (%TOC)

i. Supporting data needs include nutrient and sediment loads, and macroalgal, microphytobenthic and RPD indicator monitoring along with %TOC. These should be done across estuary types where appropriate. Sampling strategies addressing how site-specific sampling can be scaled to estuary wide characterisations should be developed (e.g., stratified-random designs).

APPENDIX 13. REDOX POTENTIAL DISCONTINUITY (RPD)

13.1 Redox Potential discontinuity (RPD)

i. Supporting data needs to include nutrient and sediment loads, and macroalgal, microphytobenthic biomass, %TOC and macrofaunal monitoring, along with depth of RPD. Work to determine estuary health responses at depth of RPD values intermediate between ~10 and 40mm (i.e., B-C threshold of Table A13-1) would be useful to firm up thresholds.
ii. Additional work should be done across estuary types where appropriate. Sampling strategies addressing how site-specific sampling for RPD depth can be scaled to estuary wide characterisations should be developed (e.g., stratified-random designs).

APPENDIX 14. SEDIMENT SULPHUR

14.1 Ratio between %TOC and %TS.

i. Sediment TS is not routinely collected in fine-scale monitoring therefore there are limited data available for a national analysis. Additional data collection across different substrate types should be considered before comprehensive data analysis is undertaken. The data analysis should address whether the proposed thresholds for TOC:TS are appropriate and whether it is applicable to all substrate types or restricted to depositional areas.
ii. Strategies addressing how site-specific sampling can be scaled to estuary-wide characterisations should be developed (e.g., stratified-random designs). These approaches, with more data collection, can then be used to assess spatial thresholds for sediment TOC:TS.

14.2 Degree of pyritization (DOP)

Not recommended for further development at this stage.

APPENDIX 15. TRACE METALS

15.1 Trace metal concentration in bed-sediment

i. Elucidate the thresholds at which adverse ecological effects occur, and whether there exist regional differences.
ii. Better understand the current status of trace metals in New Zealand estuaries, and the extent to which differences within and among regions can be related to factors such as catchment land use.
iii. Provide insight into 'reference' conditions for trace metals in New Zealand estuaries.

APPENDIX 16. CYANOBACTERIA

16.1 Planktonic cyanobacteria (human health)

i. Review available international literature to assess the feasibility of developing planktonic cyanobacteria guidelines for estuaries. A project titled "Managing marine harmful algal blooms (HABs) in recreational settings", currently being undertaken by Cawthron Institute, alongside Health New Zealand (Te Whatu Ora) and the New Zealand Ministry of Health (Manatū Hauora), goes some way toward achieving this (Smith 2024 in prep).

ii. Data collection is likely required across a range of estuary types to assess the most common cyanobacteria species and cyanotoxins present before toxicological studies can be undertaken to develop thresholds. This data should be collected alongside in situ water quality indicators in addition to nutrient loads to improve local stressor-response relationships.
iii. Further research is required to assess the use of cyanobacteria as an ecological health indicator (e.g., effects on seagrass, macrofauna, fish, birds etc.).
iv. Explore estuary-wide measures of cyanobacteria (e.g., remote-sensing).

16.2 Benthic cyanobacteria (human health)

i. Review available international literature to assess methods for measuring benthic cyanobacteria in estuaries. Further consideration of the effects of biomass on toxin concentration may also be necessary.
ii. Review available international literature to assess the feasibility of developing benthic cyanobacteria guidelines for estuaries. A project titled "Managing marine harmful algal blooms (HABs) in recreational settings", currently being undertaken by Cawthron Institute, alongside Health New Zealand (Te Whatu Ora) and the New Zealand Ministry of Health (Manatū Hauora), goes some way toward achieving this (Smith 2024 in prep).
iii. Data collection is likely required across a range of estuary types to assess the most common benthic cyanobacteria species and cyanotoxins present. This data should be collected alongside in situ field measures (e.g., water quality, substrate type, sediment quality) in addition to climate variables and nutrient loads to improve local stressor-response relationships.
iv. Further research is required to assess the use of benthic cyanobacteria as an ecological health indicator (e.g., effects on seagrass, macrofauna, fish, birds etc.).
v. Explore estuary-wide measures of benthic cyanobacteria (e.g., remote-sensing).

APPENDIX 17. DISSOLVED OXYGEN (WATER COLUMN)

17.1 Dissolved oxygen (Water column)

i. A protocol should be developed that specifies where, when, and how samples should be collected, e.g., standardised protocols and/or guidance for measurement of DO, including the spatial (across estuary, surface vs. bottom v. integrated) and temporal density of data collection.
ii. Develop an assessment framework that clearly articulates how data would be applied to decide whether the estuary is impaired. Consideration should be given to formalise guidance for monitoring programmes and interpretation of DO data. Part of this guidance should include methodologies to interpret temporal/spatial representation data in the context of 'acute' and 'chronic' threshold limits. Related to this, appropriate averaging periods for 'acute' and 'chronic' criteria should help in establishing defensible thresholds (c.f., Sutula et al. 2012)). From this perspective, New Zealand datasets collected at high frequency (minutes) for long periods (months to years) could be interrogated statistically to determine to the optimum balance of averaging periods for DO criteria, and associated sampling designs for frequency and duration of DO monitoring. An example of such a dataset is that of Zeldis et al. (2022) for Firth of Thames.

APPENDIX 18. NUTRIENTS (WATER COLUMN N AND P)

18.1 Potential nutrient concentrations (nitrogen and phosphorus)

i. Analyse all regional council / research organisation nutrient concentration data sets to:
ii. Compare measured nutrient concentrations with macroalgae EQR and chlorophyll-a thresholds (and their associated potential nutrient concentrations) for estuaries where such indices are available.
iii. Assess if threshold measured nutrient concentrations are evident, either annually, with respect to season, or with respect to estuary type.
iv. As part of this, assess if 'reference' estuaries exist, with low eutrophication indices; these could provide a basis for minimum nutrient concentration threshold(s) (cf. The TRIX trophic index of Maurizio et al. (2007)).
v. Assess if consistently eutrophic estuaries exist; these could provide a basis for 'bottom line' nutrient concentration threshold(s).

vi. For open coastal waters of New Zealand, we recommend further development of regional baseline values using repeated open coastal sampling at sites with minimal anthropogenic influence.

APPENDIX 19. PHYTOPLANKTON

19.1 Phytoplankton biomass

i. Collate standardised national data (and associated metadata) on phytoplankton biomass (chlorophyll-a) across different estuary types.
ii. Collate ancillary data on factors potentially affecting phytoplankton biomass levels, particularly long-term average observed nutrient concentrations, potential nutrient concentrations, and loads.
iii. Additional data collection is likely required across a range of estuary types (including euhaline systems, as noted above) to assess the suitability of the proposed thresholds to New Zealand estuaries. These data should be collected alongside supporting field indicators in addition to nutrient loads to improve local stressor-response relationships (i.e., relationship of phytoplankton levels to eutrophication indices, e.g., hypoxia, acidification, organic deposition.
iv. Develop guidance for sampling phytoplankton in discrete water quality monitoring taking into consideration both well mixed and stratified estuaries.
v. Explore whether current thresholds can be scaled to estuary-wide measures of chlorophyll-a through remote sensing and/or predictive modelling.
vi. Explore taxonomic composition of phytoplankton as an indicator of eutrophication and incidence of harmful algal blooms.

Greenhills Estuary, Tasman

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